# Recommended mitigation bundles for cost analysis of mitigation of sediment and other freshwater contaminants in the Rangitāiki and Kaituna-Pongakawa-Waitahanui Water Management Areas

Prepared for the Bay of Plenty Regional Council

Final report, forming partial delivery for Milestone 1A

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Perrin Ag Consultants Ltd and Manaaki Whenua Landcare Research

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# Contents

# 1 Executive summary

A list of 43 rural land use management and land use change mitigations were evaluated for their effectiveness and cost to the farm or orchard system in order to develop mitigation bundles for use in evaluating the cost of improving water quality in the Kaituna-Pongakawa-Waitahanui and Rangitāiki Water Management Areas (WMAs).

Similar to Vibart et al. (2015) and Daigneault and Elliot (2017), a cumulative three-layer framework was developed to bundle the mitigations. However, in this case, bundles were primarily determined based on cost at the farmgate, filtered for effectiveness at reducing contaminant losses. These mitigation strategy bundles, designed to be applied cumulatively to farm and orchard systems, are:

(i) M1: low barrier to adoption; primarily defined by being of low cost (equivalent to less than 10% of EBIT) with a minimum least low effectiveness;

(ii) M2: moderate barrier to adoption; primarily defined by direct costs and/or lowered revenue equivalent to more than 10% but less than 25% of EBIT and at least medium effectiveness for the targeted contaminant;

(iii) M3: high barrier to adoption, primarily defined by significant reductions in pre-mitigation profitability (>25% EBIT) and high effectiveness at contaminant reduction;

Total land use change mitigations were considered a separate bundle (M4) and excluded from consideration.

These bundles were then further considered for applicability on each of the five major land use categories used in the APSIM and eWater SOURCE catchment model, which will be the basis for the bio-physical analysis undertaken for these two WMAs.

Testing both the definitions of the bundles and farmer familiarity with the individual mitigations themselves at the planned community group meetings will be critically important.

# 2 Overview

In this report, we aim to provide guidance on the suggested bundling of different practices to reduce sediment and other freshwater contaminants from rural land use in the Bay of Plenty Region. Such bundling needs to be structured around both the cost to growers/farmers from implementation and the effectiveness of the mitigation(s) in reducing contaminant load.

Studies looking at the effectiveness and cost of both individual and suites/bundles of on-farm and on-orchard mitigations to improve water quality have been regularly undertaken in the last decade. These have tended to look at the four primary contaminants to water – nitrogen (N), phosphorus (P), sediment and bacteria such as *Escherichia coli (E. coli).* As a result, there is reasonable understanding amongst the scientific and farming community about the relative costs and benefits of various systems and land use changes with regard to mitigating contaminants to water from agricultural land use.

Previous publications that summarise mitigation options for farmers include Low et al (2017), McDowell et al (2013), McKergow et al (2007), Ritchie (2008), Waikato Regional Council (2013) and Wilcock et al (2008). A bundled approach to considering mitigations has previously been considered in New Zealand, including by Vibart et al (2015), Daigneault & Elliot (2017) and Monaghan et al (2016). However, research to increase understanding around the applicability of, and expected effect from, the adoption of individual and bundled practice change within individual regions, freshwater management areas and sub-catchments is ongoing.

Accordingly, in this report we have attempted assess the costs of sediment and other freshwater contaminants' reduction from implementing different mitigations, with a long list of suggested practices used by the BOPRC in canvassing community groups in the targeted WMAs as the starting point. This report then presents a short[ened]-list of mitigation options that are grouped together to form different bundles based on cost. While the mitigation options included in each bundle may or may not be implemented, the aim is to define a range of mitigation options that covers the range of costs likely to be experienced when implementing mitigation options and the range of effectiveness covered by possible mitigation options.

To make such assessment, we have completed a high-level review of the current literature related to on farm land use management practices and supplementary (technological) mitigation options, as well as our own experiences in evaluating cost to farmers and growers from implementing practice change, which has often involved analysis using Farmax[1] and OVERSEER[2] software.

We note that the literature reviewed is not consistent in its estimates or reporting of "cost" to farmers/grower in terms. "Cost" has previously been defined as everything from a relative cost assessment, gross (absolute) cost, cost as a percentage reduction in profit through to a cost per unit of contaminant reduced. With the emphasis in this piece of work being on the cost to farmers and growers, expressing the cost of a mitigation as the equivalent % reduction in annual operating profit (defined here as earnings before interest and tax) is probably most helpful.

[1] http://www.farmax.co.nz/
[2] https://www.overseer.org.nz/

Based on the expected cost of mitigation options identified in the review, the potential mitigations will be structured into suggested low, medium and high cost mitigation bundles for subsequent modelling. Using a framework proposed by Macdonald (2018) (see Section 4 below), proposed mitigations will also be cross-referenced against effectiveness. This will ensure that potentially high cost mitigations with low effectiveness at reducing contaminant load will not be recommended.

## 2.1 Description of contaminants and key pathways to water

### 2.1.1 Nitrogen loss

Nitrogen typically enters waterways as nitrate ($NO_3$-) through drainage, with such losses variable throughout the season based on rainfall, underlying pasture growth and soil moisture conditions. OVERSEER modelling can account for some of these drivers of loss rates. While direct losses are possible through fertiliser or effluent application [via overland flow], the uneven redistribution of N via the livestock urine patch is the primary driver of N loss in pastoral systems. Mineralisation of soil organic matter from cultivation or the excessive application of nitrogen (to ensure N is non-limiting to a developing plant) is a more typical driver of loss in arable and horticultural systems.

Most mitigation practices in relation to reducing N loss to water focus on improving the N conversion efficiency of the agricultural system.

### 2.1.2 Phosphorus loss

While OVERSEER modelling can estimate average P losses from farming activity, the reality is that such losses are neither uniform across the relevant parts of the property, either spatially or temporally. It is recognized that 80% of all P losses from a pastoral farming operation come from 20% of the property (Gburek & Sharpley, 1988), particularly those areas where transport mechanisms (i.e. water flows) and contaminant sources, such as stock camping areas, water trough surrounds, coincide. These have been defined by McDowell & Srinivasan (2009) as critical source areas ("CSAs").

While it is impossible to eliminate the creation of these CSAs within a farming or horticultural environment, strategies to slow the movement of storm water through ephemeral channels (to facilitate sediment deposition) or break the connectivity between ephemerals and these risk areas tend to dominate P loss mitigation.

### 2.1.3 Sediment loss

Sedimentation happens in wetlands, lakes, slow-flowing parts of rivers and estuaries, when the sediment load received from the freshwater catchment exceeds their capacity to flush out the sediment. Sediment loads can be caused by mass movement, gully, sheet and rill, streambank and human induced ground erosions. Sedimentation might increase when there is land without (native and exotic) forestry[3] on steep slopes, land with heavily grazed vegetation, soils with poor infiltration

[3] Including after forestry harvest

and saturated soils. The sedimentation damages fish populations, degrades benthic habitat, and smothers river beds.

### 2.1.4 Bacterial contamination

*E. coli* is used as an indicator of freshwater bacterial contamination from animal faeces and is one of the attributes of the “Human Health” water quality value. The higher *E. coli* indicate an increasing risk of infection in humans who use fresh water for primary and secondary recreation activities. *E. coli* enters streams through a direct deposition of faecal matter of livestock, discharges of dairy effluent into streams, overland flow from excess irrigation water and drainage. The main source of such freshwater contamination is ultimately grazing livestock.

# 3 Assessment of mitigations

Descriptions of sediment and freshwater contaminant reduction and costs of mitigation options are given in Table 1 overleaf based on a review of published research. More detailed description of each mitigation option is given in Appendix 1.

In considering the mitigations in Table 1 below, it is important to recognise that the evaluations of effectiveness (“expected reduction [in losses] from baseline” have been developed from a mixture of empirical research and modelled analysis. The reality is that the impact in real situations could be highly variable depending on individual situations. As such, the information presented should be considered useful for the purposes of relative assessment, rather than absolute accuracy.

All of the mitigations considered would have a high level of applicability to other parts of the Bay of Plenty region.

**Table 1**: Summary of water contaminant mitigation practices to be considered in the Kaituna-Pongakawa-Waitahanui and Rangitāiki water management areas (? = Uncertain)

| Mgmt Area | Mitigations | Expected reduction from baseline: N leaching | P loss | Sediment/ erosion | *E. coli* | Cost (% reduction in EBIT)[4] level | Nominal costs: Initial capital | Operating (recurring) costs[5] | Additional details | References |
|---|---|---|---|---|---|---|---|---|---|---|
| **Land use** | Land use capability (LUC) class 6, 7 and 8 land that is currently in pasture converted into forestry/mānuka and fenced | 4% | 15% | 80% | ? | Medium (steep land) to High (easy contoured land) | $1,000-$2,000/ ha | Just ongoing maintenance | Opportunity cost is 100% of profits from the area occupied by trees, but generates income from trees over time. | Daigneault et al (2017); Doole (2015) |
| | Creation of new wetlands (assumes 1% of farm area) | 40% | 70% | 80% | Up to 50%[6] | High | $8,940/ ha of wetland, including planting and fencing | $300/ wetland | One wetland can cover 400 ha of area | Daigneault & Samarasinghe (2015); Doole (2015); Low et al (2017) |
| | Management of gorse (e.g. replacing with pasture, mānuka or natives) | 80% on areas converted to trees, 50% to dry stock farming | ? | ? | ? | Medium | $1,000-$2,000/ ha (assumes trees) | Just ongoing maintenance dependent on subsequent land use | Opportunity cost is 100% of profits from the area occupied by trees, but generates income from trees over time. | Magesen & Wang (2008) |
| | [Complete] Land use change to a less intensive use (e.g. sheep, deer, horticulture, forestry) | 50% in changing from dairy to dry stock, 80% in converting from grass to trees | ? | ? | ? | High | Variable depending on relative value of stock classes and infrastructure required | $140-$1,000/kg per N loss reduction | The cost levels occur depending on former and current land use practice. Excludes loss of capital value | Perrin Ag (2012) |

[4] Will include the annual opportunity cost of capital associated with capital investment
[5] Can include the annual depreciation cost of capital investment
[6] But recent NIWA work indicates more complexity in this issue

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] level | Nominal costs | | Additional details | References |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment/ erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| **Riparian management** | Effective stock exclusion and planted buffer around water bodies | 15% for dairy; 5% for drystock | 10% for dairy; 5% for drystock | 40% | 25-35% | Medium to high | $255/ha | Just ongoing maintenance | A minimum of $255/ha, subject to the opportunity cost of buffer, its width and range of waterbodies are excluded. | Doole (2015); Dymond et al (2016), Keenan (2013); Monaghan and Quinn (2010) |
| | Stock water reticulation away from surface waterbodies | 15% for dairy; 5% for drystock | 10% for dairy; 5% for drystock | 40% | 25-35% | Medium | $142-$601/ha | $3.13-$12.56/ha | Results in good medium-term payback, but some benefit may be extracted through higher carrying capacity, which may increase N losses | Doole (2015); Journeaux and Van Reenen (2017) |
| **Erosion control** | Swales, soak holes, slag socks, sediment ponds, | None | 0-20% from swales | Swales reduce by 40%; Sediment ponds by 50% | None | Medium to high | $255-$1,300/ha | None | Swales cost $255/ha; sediment ponds cost $750-1,300/ha, | Keenan (2013) |
| | Detainment bunds | None | Variable | Variable | ? | Medium | $300-$500/ha of catchment | Elimination of P fertiliser from ponding areas | Detention bunds appear to be effective at catching particulate P in overland flow, but what this actually equates to on a farm or catchment scale is not fully understood. Not modelled in OVERSEER. | Clarke et al. (2013), http://www.rotorualakes.co.nz/vdb/document/796 |
| | Complete protection of gully heads | None | None | 70-90% | None | High | $1,000-1,650/ha | Just ongoing maintenance | Considering protection using afforestation | Daigneault et al (2017) |

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] level | Nominal costs | | Additional details | References |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment/ erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| Erosion control | Manage risk from contouring/ landscaping | ? | ? | 40% | None | Low | None | $82/ha cropped | Implemented on cropped area | Keenan (2013) |
| Erosion control | Spaced planting of poplars or willows on land use capability class 4-6 (steep erodible) land | None | 20% | 70% | None | Low to Medium | $34/ha | | Costs are annualized | Daigneault and Elliot (2017) |
| Stock management | Appropriate stock type and stocking rates for land characteristics (e.g. sheep on steeper land) | 21% | 2% | None | ? | Low to Medium | 35% reduction in profits per hectare in comparison to baseline practice | None | Reductions in stocking rate of lamb finishing farms with some beef finishing | Doole (2015) |
| Stock management | Change in sheep to cattle ratio by increasing sheep ratio | 19% | 4% | None | ? | Low | Variable, depending on relative value of stock classes | 91% increase in profits per hectare in comparison to baseline practice, but highly dependent on underlying market relativities | Includes hill-country beef farm with no sheep. Mitigation practice is introduction of sheep. Impact on profitability does depend on market. | Doole (2015) |
| Stock management | Rotation, grazing management (e.g. wintering off away from catchment or in less sensitive area within catchment) | 36% for dairy; 16% for S+B | 30% for dairy; 20% for S+B | 40% for dairy; 10% for S+B | 10% for dairy; 10% for S+B | Low | None | $2-$30/head/week, depending on stock class and species | Can be costly, but a regular component of many dairy farm systems due to high rate of return. However, applicability as a mitigation moving forward is uncertain | McDowell et al (2005); McDowell and Houlbrooke (2009) |

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] | Nominal costs | | Additional details | References |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment/ erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| Stock management | Appropriate location of feeding and stock drinking water through sites away from waterways | None | Variable | Variable | Variable | Medium | Variable | Ongoing maintenance | Extent of contaminant reduction depends on the extent of hydraulic connectivity from these CSAs | |
| | Responsible break-feeding practices | None | Up to 80% | Up to 80% | ? | Low | None | 2.5% reduction in crop areas | Should be no significant cost associated with this change in management approach. | Orchison et al (2013) |
| | Low leaching animal varieties | 9% | None | None | None | Medium | Variable | Variable | | Perrin Ag (2013) |
| | Dung beetles | ? | 70-100% via overland flow | 70-100% | 35% | Low | $6,000 per farm for colony establishment (for 150ha) | None | Preliminary trial NZ trial work is encouraging and in line with other global research. Additional NZ work is currently being undertaken. | Brown et al (2010), Dymond et al (2016), Forgie et al (2018), Paynter et al (2018), Slade et al (2016) |
| | Barns for intensive systems or in sensitive environments | 15%- - 17% | 15% | None | 10% | High | $1,000-$2000/ cow | $171/ha | Less than half case study farms in Journeaux & Newman generated a return that exceeded their cost of capital. Utilising a barn to reduce N losses is unlikely to be profitable | Greenhalgh (2009); McDowell (2014); Perrin Ag (2013); Journeaux & Newman (2015); Daigneault et al. (2017), Perrin Ag (2018) |

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] | Nominal costs | | Additional details | References |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment/ erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| Feed and crop management | Low nitrogen-leaching pasture/fodder crop/imported feed varieties | 33% | 6% increase | None | None | Low | None | $87-$391/ha reduction in profits depending on reduction of maize | Represents hill-country bee-breeding farm without sheep and the use of maize-silage crop for dairy support | Doole (2015 |
| | No tillage/low impact cultivation (e.g. along contours, appropriate for season, strip tillage, direct drilling) | 10% | 50% | 25% | None | Low | None | $171/ha | Expected reduction of 10% in EBIT from arable cropping | Daigneault and Elliot (2017) |
| | Winter forage crop management | 25% | Up to 80% | Up to 80% | Mode rate | Variable | None | Possibly reductions in costs | This is a potential combination of grazing practices, crop establishment and cover crop usage. | Carlson et al (2013), Lucci (2013), Orchison et al (2013) |
| | Grass buffer strips (2-metre) around cropping paddocks | 10-20% | 15-30% | 65% | 80-95% | Low | None | $175/ha to be mitigated | Price is dependent on area, buffer width and vegetation used | Barber (2014); Low et al (2017); Wilcock et al, (2009) |
| | Cover crops between cultivation cycles | 70-80% if planted in March; 25% if planted in June | None | None | None | Low | None | $80/ha for cropped area | | Low et al (2017) |
| | Earth decanting bunds for intensive cultivation | None | None | 87.5% | None | Low | None | $130/ha | Recommended capacity is 0.5% (50m/ha) for catchments less than 5ha, and 1% (100m/ha for catchments over 5ha | Barber (2014), Low et al (2017), Doole (2015) |
| | Alum applied to pasture or forage crops | None | 30% at grazed cropland; 5-30% at pasture | None | None | High | None | On grazed land $160-$260/kg of P conserved; On grazed cropland $150-$500/kg of P conserved | | McDowell (2010) |

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] | Nominal costs | | Additional details | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment/ erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| Access/crossing infrastructure | Access crossings, bridges, culverts over all waterways regularly crossed by stock | None | 95% | 99% | Variabl e | High | ? | ? | Can be a significant cost depending on the size of the catchment the waterway drains. | Low et al. (2017) |
| | Appropriate gate, track and race placement, design and maintenance (e.g. diverting effluent away from waterways, slope access tracks away from drains to reduce sediment loss and avoid water flowing across disturbed area) | None | Variable, but based on the work of McDowell & Srinivasan (2009), could contribute significantly to reductions in losses if these form critical source areas. | | | Low to medium | ? | ? | Maintaining water tables and laneway camber is cheap to achieve but shifting gateways out of flow paths can be costly if an existing race network also needs to be altered. At a whole farm level, contaminant reduction can be significant (up to 80% if all managed effectively) | McDowell & Srinivasan, 2009 |
| Fertiliser management | Paddock/block-level fertiliser planning/nutrient budget based on soil tests and crop needs | 10% | 10% | None | None | Low | None | $500 per year | Gains likely to be in association with other practices highlighted by appropriate nutrient budgeting | |
| | Maintaining optimal soil phosphate levels | None | 18% | None | None | Low | None | Potentially as high as $200/ha/ year savings while mining excessive soil P levels | Extend of gain will depend on level of above optimal soil enrichment | Perrin Ag (2017c) |
| | Use of low solubility P fertiliser | None | 6% | None | None | Low | Some initial capital application might be required to buffer lag of availability | None | The value of P in RPR tends to be lower than in superphosphate, but sulphur will generally also need to be added as well. The availability of the P from RPR will be limited initially, so best used in conjunction with mining of soil Olsen P levels | |

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] | Nominal costs | | Additional details | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment /erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| Fertiliser Management | Efficient fertiliser use (e.g. not coinciding with rainfall, temperatures below 7 degrees Celsius, appropriate fertiliser types and timing of application, Geographical Positioning System [GPS]-based application). | Low (3%) | Variable | None | None | Low | None | Proof of placement technology will incur ongoing costs (est. $2,000 per annum), but savings in N fertiliser use expected | Costs based on fertiliser application level | Grafton et al (2011, 2013), Perrin Ag (2017b) |
| | Reducing fertiliser N use | 15%-33% | None | None | None | Medium | May result in reduction in stock numbers if being used to support capital livestock | Net benefit- $350/year/kg N loss reduction | The extent of any profitability change tends to relate to the cost of any feed purchased in to replace the N boosted pasture or the amount of production forgone by the loss of the feed. | AgFirst (2009), Perrin Ag (2012) |
| | Use of plant growth regulators (Gibberellic acid) | 4-29% | None | None | None | Low | None | $38/ha, but will expect some N savings | Application level is 20kg/ha | Ghani et al. (2014), Bryant et al. (2016) |
| Irrigation | Efficient irrigation application based on soil moisture deficit monitoring, awareness of soil type/infiltration rate and assessment of crop needs and expected rainfall | 10% | None | None | None | Low | Cost of mid-range tensiometer could be as little as $1,100 | $58/ha of annualized costs | | McDowell et al (2013), Strong (2001) |

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] | Nominal costs | | Additional details | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment/ erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| Effluent management | Solid separation | Variable | Variable | None | Variable | Medium to high | Costs vary depending on whether passive or mechanical separation is chosen | Ongoing maintenance and cost of spreading solids ($2,000-$3,000 p.a.) | Weeping walls, screw press and fixed screens are all examples of this technology. Cost of investment and ongoing operation can vary hugely. | Longhurst et al 2017 |
| | Closed loop effluent recycling | ? | ? | ? | ? | Medium | $397,000 (based on stated payback of 7.5 years and a suggested $53,000 annual gap between annual costs of pond system versus the FORSI system) | $18,000 per annum | Still require solids separation (via a screen) and disposal of solids to land. No trial work available, but concept has long term potential for farms constrained by soil moisture levels for land-based liquid effluent disposal | https://www.forsi.co.nz/wp-content/uploads/forsi-effluent-recycling-system-2017.pdf |
| | Farm Dairy Effluent ponds: sufficient holding capacity to comply with soil moisture application standards and fully lined | ?, but as much as 5% | 10-30% | None | Up to 25% | Medium | $30,000-$100,000 depending on size of farm | $30/kg of P conserved | High capital cost | Dymond et al (2016), McDowell (2010), Low et al. (2017), Perrin Ag (2018) |

| Mgmt Area | Mitigations | Expected reduction from baseline | | | | Cost (% reduction in EBIT)[4] | Nominal costs | | Additional details | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N leaching | P loss | Sediment / erosion | *E. coli* | | Initial capital | Operating (recurring) costs[5] | | |
| Effluent Management | Maize on the effluent block | Variable | None | None | None | Low | None | $140/ha benefit assuming half of N fertiliser could come from effluent | Should allow a reduction in base N fertiliser requirements | FAR (2008), Johnstone et al (2010). |
| Effluent Management | Efficient application that complies with soil moisture standards and crop needs, more than 20 metres away from all waterbodies | Variable | Variable | None | Variable | Low to medium | Limited | None | $500 for basic soil moisture probe, but on high risk soils more investment may be required | Perrin Ag (2018) |
| Effluent Management | Increase application area to reduce application concentration | Variable | Variable | None | Variable | Medium to high | C. $705/ha | Ongoing maintenance | Depends on spatial layout of the farm and existing effluent areas | Perrin Ag (2018) |
| Denitrification | Use of nitrification inhibitors | 10% | None | None | None | Medium | None | Prev. $97/ha applied | Products currently banned for use in NZ | Di & Cameron (2007) |
| Denitrification | Denitrification technology (i.e. Spikey) | 10% | None | None | None | Medium | Investment in equipment | Potentially increased pasture production could offset increased costs, but limited field trials | Moderate capital investment, returns potentially good, but field trials still ongoing | Bates & Bishop (2016) |
| Denitrification | Denitrification beds | 25% | None | None | None | High | High (not costed) | $137/ha of annualised cost | High capital cost plus. Loss of some fertiliser value from dairy effluent | Schipper et al (2010); McDowell (2013) |

# 4 Proposed mitigation bundles

In contrast to Vibart et al. (2015) and Monaghan et al. (2016), in this study the mitigation practices that are summarised in Table 1 have been bundled based on their cost level (expressed as a reduction in pre-mitigation farm profit as measured by EBIT), but first having been filtered based on their effectiveness as proposed by Macdonald (pers. comm, 2018). This framework is presented in Figure 1 below.

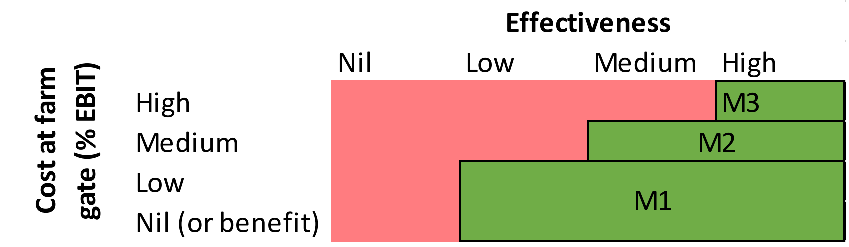


**Figure 1:** Bundling framework as suggested by Macdonald (pers. comm 2018)

For clarification, the "cost" of mitigation should include the opportunity cost of any capital employed and the loss of value (depreciation) over time, even though the former isn't captured in EBIT. These total mitigations are simply being considered in relation to amount of pre-tax profit that might be consumed as a result of its implementation.

The bundles are therefore broadly defined as:

(iv) M1: low barrier to adoption; primarily defined by being of low cost (equivalent to less than 10% of EBIT) with a minimum least low effectiveness;

(v) M2: moderate barrier to adoption; primarily defined by direct costs and/or lowered revenue equivalent to more than 10% but less than 25% of EBIT and at least medium effectiveness for the targeted contaminant;

(i) M3: high barrier to adoption, primarily defined by significant reductions in pre-mitigation profitability (>25% EBIT) and high effectiveness at contaminant reduction;

The mitigation bundles are designed to be applied cumulatively to farm and orchard systems i.e. M2 mitigations are applied only after applicable M1 mitigations have been implemented on farm.

This framework potentially includes two additional bundles, which have not been listed in the following tables:

(ii) M0: existing [best] management practice already assumed to be largely in place within farm systems (such as stock exclusion of dairy cattle from waterways) with essentially no cost to adoption.

(iii) M4: total land use changes

Based on the above, the proposed mitigation bundles M1 to M3 generated from this analysis are presented in Table **2** through Table **4** overleaf.

In reaching these final bundles, it is important to highlight several practices from the long list of specific mitigations have been excluded from the current bundles due to a current shortage of trial data of their impact on contaminant load to water in the NZ context and low current extent of adoption. However, these mitigations have some promise with regards to cost-effectively lowering the loss of N, P, sediment and/or bacteria to water from our farm and orchard systems. These specifically included:

- the “Spikey’ technology;
- introduction of dung beetles to pastoral systems.

**Table 2:** Summary of the proposed M1 mitigation bundles to be considered (as applicable) in the Kaituna-Pongakawa-Waitahanui and Rangitāiki WMAs

<table>
<tr><th rowspan="2">Mitigation bundle</th><th colspan="5">Land use type</th></tr>
<tr><th>Dairy pastoral</th><th>Non-dairy pastoral</th><th>Arable</th><th>Horticulture</th><th>Forestry</th></tr>
<tr><td>M1</td><td>▪ Placement of feeding equipment<br>▪ Timing of effluent application in line with soil moisture levels (assumes sufficient storage)<br>▪ Reduced tillage practices<br>▪ Improved nutrient budgeting and maintenance of optimal Olsen P<br>▪ Laneway run-off diversion<br>▪ Grow maize on effluent blocks (if already growing maize)<br>▪ Elimination of summer cropping<br>▪ Reductions in seasonal stocking rate<br>▪ Efficient fertiliser use technology<br>▪ Efficient irrigation practices (soil moisture monitoring)<br>▪ Use of plant growth regulators [to replace N]<br>▪ Adoption of low N leaching forages<br>▪ Relocation of troughs<br>▪ Slow release phosphorus fertiliser RPR<br>▪ Reduce autumn N application - replace with appropriate low(er) N feed<br>▪ 3m average vegetated and managed buffer around rivers, streams, lakes and wetlands subject to the Dairy Accord; 1m around drains; 5m average buffer on slopes between 8 and 16 degrees, 10m average buffer on slopes above 16 degrees</td><td>▪ Improved nutrient budgeting and maintenance of optimal Olsen P<br>▪ Efficient fertiliser use technology<br>▪ Stock class management within landscape<br>▪ Adopt M1 arable cultivation practices for winter cropping<br>▪ Laneway run-off diversion<br>▪ Relocation of troughs<br>▪ Appropriate gate, track and race placement, design (where possible)<br>▪ Targeted space planting of poles<br>▪ Slow release phosphorus fertiliser RPR<br>▪ Adoption of low N leaching forages<br>▪ Full stock exclusion from all waterbodies greater than 1m wide at any point adjacent to farm (including drains) and wetlands. 2m average vegetated and managed buffer around rivers, streams, lakes and wetlands; 1m around drains; 3m average buffer on slopes greater than 8 degrees; 5m average buffer on slopes greater than 16 degrees.</td><td>▪ Grass or planted buffer strips<br>▪ Complete protection of existing wetlands<br>▪ Maintain optimal Olsen P<br>▪ Efficient fertiliser use and technology<br>▪ Cover crops between cultivation cycles<br>▪ Manage risk from contouring<br>▪ Reduced tillage practices</td><td>▪ Complete protection of existing wetlands<br>▪ Maintain optimal Olsen P<br>▪ Laneway run-off diversion<br>▪ Efficient fertiliser use and technology<br>▪ Efficient irrigation practices (soil moisture monitoring, not following fertiliser application)<br>▪ Grass swards under canopy, minimise bare ground and vegetated buffers around waterways.</td><td>▪ Management of gorse<br>▪ Complete protection of existing wetlands</td></tr>
</table>

**Table 3:** Summary of the proposed M2 mitigation bundles to be considered in the Kaituna-Pongakawa-Waitahanui and Rangitāiki WMAs

| Mitigation bundle | Land use type | | | | |
|---|---|---|---|---|---|
| | Dairy pastoral | Non-dairy pastoral | Arable | Horticulture | Forestry |
| M2 | ▪ Increase effluent application area<br>▪ Develop a detention bund<br>▪ Controlled grazing with stand-off pads (16 hours per day on pad in autumn), if they already have a stand-off pad<br>▪ Installing variable rate irrigators on existing pivot irrigators<br>▪ Reduce imported autumn supplement fed by 20%<br>▪ Reducing fertiliser N use (to 100kg N/ha)<br>▪ Full stock exclusion from permanently flowing waterbodies less than 1m wide (REC Order 2 and above) and average 2m vegetated and managed buffer; 3m average buffer on slopes between 8 and 16 degrees, 7m average buffer on slopes above 16 degrees | ▪ Eliminate N that supports capital livestock<br>▪ Detention bunds<br>▪ Complete protection of gully heads<br>▪ Management of gorse<br>▪ Whole paddock space planting of poles<br>▪ Full stock exclusion from permanently flowing waterbodies less than 1m wide (REC Order 2 and above) and 1m average vegetated and managed buffer; 2m average buffer on slopes greater than 8 degrees, 3m average buffer on slopes greater than 16 degrees [with associated stock water reticulation, if any].<br>▪ Convert steep land (e.g. LUC class 7-8, >26 degrees) into forestry/mānuka and fenced<br>▪ Changing stock ratios to reflect lower N leaching potential | ▪ Use of silt fencing<br>▪ Complete protection of gully heads -N/A<br>▪ Reducing fertiliser N use<br>▪ Strip tillage | ▪ Detention bunds in gullies (where they exist in amongst orchard properties) | |

**Table 4:** Summary of the proposed M3 mitigation bundles to be considered in the Kaituna-Pongakawa-Waitahanui and Rangitāiki WMAs

| Mitigation bundle | Land use type | | | | |
|---|---|---|---|---|---|
| | Dairy pastoral | Non-dairy pastoral | Arable | Horticulture | Forestry |
| M3 | ▪ Afforestation of erosion prone land (e.g. >26 degrees)<br>▪ Stock excluded from REC Order 1 watercourses less than 1m wide and 1m wide average vegetated buffer<br>▪ Impervious effluent storage and sufficient capacity to comply with soil moisture guidelines and low rate effluent application<br>▪ Restricted grazing in covered stand-off pad, with use extended to winter as well<br>▪ Put in standoff pad if they haven't got one and use for 16 hours per day in autumn<br>▪ Switching from manual (e.g. K-line) to pivot irrigators with variable rate irrigators – irrigated dairy farms with manual irrigation systems only<br>▪ Creation of new wetlands<br>▪ Reducing stocking rates down by 0.3 cows/ha | ▪ Full stock exclusion from REC Order 1 watercourses less than 1m wide and 1m wide average vegetated buffer.<br>▪ Creation of new wetlands<br>▪ Eliminate N that supports trading livestock<br>▪ Reducing stocking rates | ▪ Creation of new wetlands<br>▪ Sediment traps | | ▪ Creation of new wetlands |

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# 6 Appendix 1

## 6.1 Land use

### 6.1.1 Land use capability (LUC) class 6, 7 and 8 land that is currently in pasture converted into forestry/mānuka and fenced

In areas where potential pasture production is low (<4t DM/ha), conversion from pastoral farming to forestry is likely to have minimal impact on farm profitability when considered on the basis of long term pricing for timber and animal products. Costs are mainly related to tree plantation establishment and harvesting, and opportunity cost of alternative land use. For instance, Perrin Ag (2013) found that when afforestation of steep hill country was modelled on case study farms in the Upper Waikato, there was limited (if any) reduction of long term enterprise operating profit. However, the precise forestry regime, harvest requirements and location relative to ports and/or mills can have significant impacts on forest profitability. We note also that the recent National Environmental Standards for Plantation Forestry place limits on the afforestation of land deemed to be of very high erosion susceptibility (https://www.teururakau.govt.nz/growing-and-harvesting/forestry/national-environmental-standards-for-plantation-forestry/erosion-susceptibility-classification/).

The economics of plantation mānuka for honey production are questionable given current establishment costs, yields and price and the suitability of targeted lands for the cost-effective harvest of the biomass needed for oil extraction is likely to be low.

### 6.1.2 Wetland and ephemeral flow path management and protection

Stock exclusion from wetlands is recognised as having positive impacts on downstream water quality. A study of a Waikato hill country seepage wetland by Hughes et al (2013) found that cattle actually spent little time grazing in the shallow wetland and the direct effects of their grazing were minor, fluxes of cattle derived pollutants and damage to wetland margins and vegetation were detected. However, deeper wetlands tend to be avoided by livestock and don't spend sufficient time in them to have a notable effect on contaminant load or sediment disturbance.

On balance, given the loss of productivity from excluding livestock from wetlands is likely to be low and the concern about the long-term effect on water quality from stock access and exclusion is a sensible practice and likely to be achievable with limited cost.

The actual development of new artificial wetlands can be extremely expensive and as a result are often better considered at a whole-of-catchment scale. The review by Low et al (2017) suggested the cost could be between $550 and $7,500/ha, depending on the extent of nutrient and sediment capture desired and the nature of the existing flow in planned wetland area. In contrast, the study by Daigneault and Samarasinghe (2015) estimated that each new wetland can cost $100,000 that covers 400 ha of area. The capacity of new wetlands to take up nutrient losses from the receiving catchment is significant, although this can take a number of years to do so and such features will eventually reach equilibrium. Also, there are high positive impact of wetlands in reducing *E. coli* (50%) and sediment losses (80%) (Low et al., 2013; Daigneault and Samarasinghe, 2015). However, 2014 research by NIWA (https://www.niwa.co.nz/freshwater-and-estuaries/freshwater-and-estuaries-update/freshwater-update-63-november-2014/surprising-net-export-of-e-coli-from) found higher concentrations of *E. coli* in the outflow than in the inflow during monthly monitoring of farm drainage to a wetland during the wetter months of the year. This is a surprising finding given that constructed wetlands treating domestic sewage usually achieved net removal of *E. coli*. It was suspected that this observed increase in *E. coli* was probably due partly to wildlife deposition, but genetic and other evidence suggests that the main source is growth of this bacterium as environmental 'naturalised' populations within the wetland. This potentially raises interesting questions regarding the microbial ecology of *E. coli*, its use as a faecal indicator and interpretation of *E. coli*-based water quality in relation to waterborne disease and the human health risk downstream from wetlands and potentially other nutrient and/or organic-rich vegetative environments.

#### 6.1.3 Management of gorse (e.g. replacing with pasture, mānuka or natives)

From a fundamental point of view, the eradication of gorse and conversion to alternative ground covers is likely to result in a reduction in N loss to water. Magesan & Wang (2008) calculated nitrogen losses to water from mature gorse stands in the Rotorua catchment at 36kg N/ha and 40kg N/ha, which would be equivalent to losses from either intensive dairy support activity or extensive dairy farm systems in the same area. However, there is insufficient information in the literature on the effect of gorse on P losses, sediment and *E. coli*.

#### 6.1.4 Land use change to a less intensive use (e.g. sheep, deer, horticulture, forestry)

Land use change to less intensive activities can substantially change the nutrient leaching, erosion and *E. coli levels*. However, currently, such practice can have limited appeal for land owners. This is typically a result of the following factors:

- Cost of transition can be high i.e. cost of orchard development ($220,000/ha for kiwifruit pergolas and shelter), deer fencing (>$20/m) and handling facilities;
- Barriers to entry to the supply chain of lower intensity alternatives with profitable returns i.e. licences for crop varietals (G3 kiwifruit licence), supplier shares (i.e. Dairy Goat Co-op milk supply rights), limited markets for supply (sheep milk);
- Likely loss of capital value with "permanent" land use change including potentially low salvage value of prior investment (i.e. dairy land being planted in radiata pine);
- Perceived or real loss of profitability and annual cashflow, particularly where existing businesses are moderately or highly geared (pasture land converting to forestry);
- A desire to prevent the "stranding of assets" that have not yet reached the end of their economic life i.e. milking parlours, feed pads etc.
- Inadequate land owner knowledge of the alternative land uses;
- Personal preference.

### 6.2 Riparian management

#### 6.2.1 Effective stock exclusion and planted buffers around drains, rivers, streams and lakes

Effective stock exclusion and riparian fencing with planted buffer includes vegetation around rivers, streams and lakes. Meta-analysis by Zhang et al (2010) found that buffer width alone accounted for

37%, 44% and 35% of the variance in removal efficacy for sediment, N & P respectively. A summary of the existing literature by Doole (2015) also suggested that the width of the buffer does have an impact on the extent of N loss reduction, but whether this is due to a greater interception area or a reduction in pastoral area (with a commensurate reduction in stocking rate) is unclear. We also note that much of the literature reviewed by Zhang considered N losses in overland flow or run-off, which in NZ pastoral systems is unlikely to be the primary pathway of nonpoint-source N loss to water.

This mitigation option focuses on preventing livestock from direct deposition of manure into these waters or direct stream bank erosion using the planted buffer. This management option will have a substantial reduction in sedimentation and *E. coli*., while to a lesser extent in reduction of N leaching and P losses. There is a concern that nutrient cycling within the riparian areas can act as an indirect source of N and P loss if planted vegetation is not regularly cut and removed (Collier et al, 2013). According to Doole (2015), use of 5-metre pastoral buffer strip can reduce actual N leaching of about 15% and 5% for dairy and dry stock farms respectively, assuming livestock had access to waterways previously.

For P loss reduction the levels are even more modest than for N leaching mitigation and is about 10% and 5% for dairy and dry stock farms (Doole, 2015). In addition, based on estimates of Keenan (2013), Daigneault et al. (2017a) showed that it is possible to reduce 40% of sediment with grass buffer strips. However, Zhang et al (2010) found that buffers composed of trees have higher N and P removal efficacy than buffers composed of grasses or mixtures of grasses and trees. The cost of establishing riparian vegetation strip is around $255/ha for horticulture (Keenan, 2013), but this will vary depending on the choice of any planted vegetation. BOPRC advise that a native sedge vegetation riparian planting strip could be established at an average cost of $20 per lineal metre of waterway planted, assuming both sides of the waterway were planted), with annual weed control costs of $130/ha retired (De Monchy 2018, pers. comm).

To date, most of the regulation and voluntary practice change around riparian management has been centred on high order water bodies and lowland drains. However, McDowell et al (2017) found that 77% of national contaminant load was coming from lower-order streams that are not currently required to be fenced. With P being the primary nutrient entering water ways from overland flow and direct [stock] deposition, the fencing of low-order streams in areas of high P load may be extremely effective in reducing pollution.

As regards to the relative cost and challenge to adoption, Vibart et al (2015) considered excluding dairy cattle from waterways to fall into an M1 bundle, sheep & beef cattle into M2 and utilising a buffer strip (7m) within M3.

### 6.2.2 Stock water reticulation in lieu of using surface waterbodies

The replacement of natural water sources with reticulated supply for livestock has the potential to improve the profitability of the pastoral operations where it is implemented, although the installation of reticulated supply is likely to require additional co-investment. Journeaux & van Reenan (2017) found in a study of 11 farmers that stock water reticulation can result in the significant internal rate of return of 53% on average. Such mitigation option can reduce *E. coli* and sediment by about 30% and 40% respectively, and with contribution on N leaching and P loss of about 10% depending on livestock type. However, stocking rate tended to increase with the introduction of reticulated stock water in the case study farms, which may in practice, lead to limited (if any) reductions of N loss to water.

## 6.3 Erosion control

### 6.3.1 Swales, soak holes, slag socks, sediment ponds, detention bunds/dams

Sedimentation (or erosion) can be controlled using swales, soak holes, slag socks, sediment ponds, detention bunds/dams. Swales are broad grass strips (like riparian grass buffer strips) used to treat sedimentation. Such practice can reduce sedimentation by 40%, in contrast to the baseline land use practice such as horticulture and pasture grazing but is highly slope dependent. The cost of such practice is about $255/ha (Keenan, 2013).

A constructed soak hole can act as a sediment trap, where sediment is collected and left to discharge to a controlled outlet or soak into the ground.

Slag socks are installed sock technologies/materials that intercept and address sedimentation of clay particles. Sediment retention ponds are constructed ponds to trap sediment at bottom of sub-catchment to tackle surface erosion and are suitable for all farm land use types. The sediment ponds can reduce erosion by 50% in comparison to farming practices, and cost of such mitigation option ranges between $750 and $1300/ha of catchment (Keenan, 2013). Detention bunds/dams or debris dams are effective in trapping erosion and associated P from water leaving pastoral farmland during rainfall and runoff events, and their effectiveness depends on influent load in the ephemeral stream . Detainment bunds temporarily pond ephemeral water (via controlled outflow) behind an earth bund (about 1.5 m high) for settling sediment and associated nutrients to onto the pasture and become part of the soil matrix (Clarke et al., 2013). Clarke et al. (2013) observed the largest retention of sediment and P was 2.7 t and 6.8 kg of P respectively in just one ponding event, but what this equated to on a whole far, scale wasn't apparent. Average P retention in Hauraki Stream catchment is 0.2 kg of P per ponding event that could save $28,000 for lake restoration costs over 20 years (Clarke et al., 2013).

### 6.3.2 Complete protection of gully heads

Once gullies have begun to form they must be treated as soon as possible to reduce negative consequences. To control gullies, building detention dams or bunds and revegetation such as afforestation and space-planting should be undertaken. Afforestation plantations can reduce erosion by 90% from the baseline if trees are not harvested (reduce erosion by 80% if trees are harvested) and can cost farmers $1000/ha (Daigneault et al, 2017). Space planting assumes that areas are planted and all tree plantations are maintained. Such land use practice can reduce sedimentation by 70% and costs $1650/ha (Daigneault et al, 2017). Typically dams are used in combination with tree plantations to control the runoff into gullies to trap sediment within gully systems.

### 6.3.3 Manage risk from contouring/landscaping

Tillage practices and cultivation on slope ridges can increase erosion. Contour strip cropping can be used and includes strip of pasture or small grain alternation with a strip of row crops. Ridges in contour strip cropping reduce the possibility of erosion. Contour strip cropping can reduce soil erosion by as much as 50% as comparing to farming up and down hills (USDA, 2013).

Cover crops are cultivated often solely to manage erosion. Planting cover crops can lead to the seasonal reduction in surface erosion in contour farming by planting legumes, cereal rye, clover and other crops in horticultural farms. According to Keenan (2013), erosion reduction effectiveness of cover crops is 40% from baseline erosion, which can cost $82/ha in an arable situation.

### 6.3.4 Spaced planting of poplars or willows on land use capability class 4-6 (steep erodible) land

While the space-planting poles on erosion prone hill country has long been accepted as an effective means of reducing erosion (Hicks 1995), the economic imperative for it is not great. Analysis by Parminter et al (2001) concluded that the productivity gain from soil retention was typically less than the suppression effect from shading on pasture dry matter production and that only on highly erodible soils and where farmers were happy with low returns on the investment from planting was the cost-benefit positive for the landowner. This analysis excluded the potential public good benefit from reducing soil erosion.

## 6.4 Stock management

### 6.4.1 Appropriate stock type and stocking rates for land characteristics (e.g. sheep on steeper land)

Treading damage to soils from livestock is recognised to have the potential to increase both the risk of surface run-off and the loss of sediment, phosphorus and nitrogen in any run-off. This risk is heightened in periods of high soil moisture, which in New Zealand typically coincides with the winter period. Nguyen et al (1998) concluded that intensive winter grazing on hill country pasture is potentially a major source of contaminant runoff to receiving waters. This is more likely to occur with [older] cattle than with sheep, but the lower pasture covers potentially achievable under sheep grazing regimes (albeit not desirable from an animal performance perspective) can expose soil to greater erosion risk. Limiting/excluding cattle older than 18 months from steeper hill slopes during winter is a recommended practice.

The risk of soil erosion from deer pacing fence lines on fragile soils can be significant but can be successfully managed by a combination of sensible fencing solutions (including remedial options for existing farms) and stock management practices (New Zealand Deer Farmers' Association 2012). However, the introduction/expansion of deer onto properties with more fragile soils (i.e. pumice) does need to be considered carefully.

The impact of stocking rate and stock type on N loss to water is reasonably well understood, with the urine patch the primary driver of N loss to water in pastoral grazing systems. As a result of urinary dynamics cattle will have a higher N loss signature than deer or sheep, and female stock a greater N loss signature than males. All things being equal, higher stocking rates will generate higher N loss to water as a result of higher quantities of N cycling through the farm system and more N therefore subject to the inefficient return via the urine patch. According to Doole (2015) appropriate stock type and stocking rates have lower P loss (2%) than N leaching (21%) reduction but can lead to profit reduction of 35% per hectare in comparison to the baseline practice. Temporal dynamics are increasingly recognised as being important, with late summer/autumn urine patches to pasture potentially having more impact that those deposited in the late winter, even with higher underlying soil drainage.

### 6.4.2 Rotation, grazing management (e.g. wintering off away from catchment or in less sensitive area within catchment)

The grazing of stock off-farm as a management practice has typically been limited to dairy farm operations, where either:

(i) a reduction in dry period feed demand is a cost-effective solution to shift feed into the early spring period to support the higher feed demands associated with lactation; or
(ii) the removal of replacement heifer feed demand allows an increase in the stocking rate of cows in-milk, with an increase in the marginal return per kg DM consumed.

The improvement in system N conversion efficiency from both strategies, as well as the reduction in urinary N deposition at a period of high drainage and low pasture growth from these management practices has also typically resulted in a reduction in direct farm N losses to water. In addition, there is high conversion efficiency for P loss, *E. coli* contaminant and erosion reduction, depending on livestock type, from rotation and grazing management. For instance, implementation of such mitigation options at dairy farm can reduce 30%, 40% and 10% of P loss, sediment and *E. coli* with a $9-$30/head/week (McDowell et al., 2005; McDowell and Houlbrooke, 2009).

However, the "exporting" of N and P loss, *E. coli* and sediment from one catchment to another as a mitigation strategy is potentially only a short-term solution, as the importance of water quality in receiving water bodies across New Zealand is of increasing importance.

### 6.4.3 Appropriate location of feeding and stock drinking water trough sites away from waterways

The importance of reducing the hydraulic connectivity of critical source areas from flow paths and waterways has been highlighted by McDowell & Srinivasan (2009). However, to reduce the cost of installation the location of stock facilities (primarily troughs) have often been placed adjacent to stock access ways, which can commonly be in flow paths. The cost of mitigation will depend on the distance required for relocation and whether the reticulation system has sufficient pressure to deliver water to the new location.

### 6.4.4 Responsible break-feeding practices

Research conducted by Orchiston et al (2013) demonstrated that break feeding [winter] forage crops with a view to managing overland flow dynamics within the crop paddock (cows entering at top end of the paddock, strip grazed moving in a downhill direction, protection of critical source areas from grazing, back-fencing every 4-5 days) resulted in a considerable reduction in the yields of sediment and nutrients carried in the flow. The cost of achieving such reductions was assessed as low (including a loss of 2.5% of potential crop yield through loss of area cropped).

### 6.4.5 Low leaching animal varieties

The relative profitability of the sheep, cattle and deer enterprises has a significant impact on the likely profitability of using livestock system change to reduce nutrient losses. While increasing the sheep/deer to cattle ratio tends to lower nitrogen losses, depending on their positions within their respective commodity cycles, implementing such a change might not lead to an increase in profitability if the lamb price is low in comparison to the beef price. Changes in livestock policies, particularly where breeding stock are involved, often have significant lag periods before increases in profitability are achieved and are not easily reversed once implemented. Altering specie ratios may also present challenges for the management of pasture quality and parasite burden.

### 6.4.6 Dung beetles

Initial NZ research (Forgie et al 2014) suggested that dung beetle activity in New Zealand pastures will result in reduced surface run-off, which is in line with the global research in this area (Brown et al 2010, Doube 2008). Given the strong association of P losses with sediment loss, the observed reduction in sediment loss of between 73-100% where dung beetles were present (Forgie et al 2018) would be expected to result in a similarly high rate of reduction in P losses. Dung beetles would also be expected to significantly reduce the loss of *E. coli* and other pathogens to water, with research in both NZ (Paynter et al 2018) and offshore demonstrating this. At a catchment scale, Dymond et al (2016) estimated *E. coli* contamination to water could be reduced by approximately 35% through the introduction of dung beetles. Other positive ecosystem benefits appear to be generated by dung beetles in pastoral grazing systems, such as reductions in greenhouse gas emissions (Slade et al 2016) and reductions in nematodes (Forgie et al 2014).

### 6.4.7 Stand-off pads or barns in dairy farm systems

Feed pads have limited impact on reducing contaminant loads to water given:

(i) the short period of time they tend to be in use; and
(ii) that the benefits from potential improvement in feed utilisation is typically captured by increased milk production, not reduced feed use, so the quantum of nutrients cycling through the farm system increases.

The use of stand-off pads in conjunction with duration-controlled[7] grazing throughout the season has, based on empirical trial work, the potential to significantly reduce the loss of N in drainage to water (in the order of 30%-40%). P loss reduction is lower than N leaching and is close to 15% reduction, while *E. coli* mitigation is about 10% lower than the current/baseline dairy farm practice (McDowell, 2014; Perrin Ag, 2013; Journeaux and Newman, 2015; Daigneault et al. 2017). However, this may come at the cost of lowered pasture production due to the changes in both the timing and form of the application of nutrients from animal excreta to the pasture (Christensen et al 2011).

Journeaux & Newman (2015) concluded, based on an analysis of 14 case study dairy farms that, in general, "inclusion of a barn without intensification of the farming system will result in a reduction in nitrogen losses, but at a (potentially significant) cost… [and] that intensifying the farm system to make the barn profitable often results in a rapid erosion of the environmental benefits". A 2013 analysis of a dairy support operation in the Taupo-Ohakuri catchment, part of the Upper Waikato Drystock Nutrient Study (Perrin Ag 2013), assessed that installing a wintering facility resulted in a

[7] Where cows graze for only 4 hours each morning and evening to consume their desired daily pasture intake and are then removed from the pasture for rumination. This differs from restricted grazing, where cows are totally withheld from the pasture during a given period (say autumn & winter) and pasture is harvested and fed to the cows on a pad or barn facility.

reduction in EBIT of ($113)/ha (23%) for a 17% reduction in N loss. At the same time, in average terms the annual operating costs are about $171/ha (Greenhalgh, 2009; Daigneault et al. 2017). A significant increase in the rate charged for contract winter grazing was required to offset the loss in profitability.

Capital costs to farmers will tend to be less for stand-off pads than that for barns, but the costs can vary widely and can be between $1,000 and 2,000 (Greenhalgh, 2009; Daigneault et al. 2017).

## 6.5 Pasture/crop management

### 6.5.1 Low nitrogen-leaching pasture/fodder crop/imported feed varieties

There are a number of alternative forage species that early research indicates have the potential to lower farm N loss to water, albeit such impacts are not well captured in OVERSEER.

Lucci et al (2015) found evidence that suggested chicory planted after a winter brassica crop recovered greater amounts of winter deposited N than a conventional ryegrass white clover sward, but this is yet to be captured in OVERSEER. Analysis by Perrin Ag (2017) indicted replacing summer brassica crops with chicory had a positive impact on farm profit, but the impact on N loss reduction as expressed in OVERSEER was limited to differences in cultivation, not crop variety.

Modelling by Khaembah et al (2014) suggested that diverse pasture mixes (containing at least 50% of alternative species such as plantain and chicory) could result in reductions in urinary N concentration and hence N leaching), but the economic impact was not determined. Subsequently, Edwards et al (2015) observed a 20% reduction in cow urinary N concentration for cows grazing a diverse pasture sward compared to those on conventional ryegrass/white clover. In similar research, Box et al (2016) found cows grazing a monoculture of plantain had reductions in urinary N of up to 56% from that of cows grazing conventional pasture. Again, insufficient data exists to include such impacts within the OVERSEER model, but the impact on productivity through the introduction of high herb content swards is unlikely to be significant, particularly if winter active varieties are selected. Doole (2015) found that substitute of maize-silage crop with low nitrogen imported feed can reduce N leaching 33% than the current feed given to livestock. However, such imported feed increased P loss by 6% and resulted in profit reduction of $87 and $391/ha depending on reduction of maize.

The Forages for Reduced Nitrogen Leaching (FRNL) project (Dairy NZ 2017) has found that leaching from a urine patch was 25-35% lower under Italian ryegrass-based pastures than under other types of pastures due to cool-season N uptake of Italian ryegrass.

### 6.5.2 No tillage/low impact cultivation (e.g. along contours, appropriate for season, strip tillage, direct drilling)

It is generally accepted that the establishment of crops or forages using conventional “full” cultivation methods result in greater rates of mineralisation of N in soil organic matter than no-till alternatives. However, the impact that this has on actual N loss on soil drainage can be variable. Carran (1990) found that a similar amount of nitrate was present in the sub-soil in mid-winter after establishment of spring sown wheat crops out of established pasture irrespective of tillage method.

However, research to date in the FRNL project found that compared with conventional tillage, direct drilling autumn-sown forage crops reduced the compaction that results from winter grazing, leading to as much as a 20% improvement in the yield of a subsequent cereal [catch] crop, which in turn increases N uptake from the soil. According to Daigneault and Elliot (2017), eliminating crop disturbance from tilling can also reduce P loss and sediment along with N leaching but reduce EBIT of arable crops by 10%.

In practice, there is little difference in the cost of establishment of crops using no-till techniques, with greater weed and pest control often required. However, irrespective of the impact on freshwater and water contaminants reduction, direct drilling or strip tillage will lower the risk of run-off and soil loss and represent a useful practice change on farm.

### 6.5.3 Winter forage crop management

Lucci et al (2013) assessed that the major risk of N losses associated with winter forage crops was associated with the risk of redistribution of N in the crop via the urine returned to the soil via grazing animals. Their research on crop establishment on pumice soils demonstrated no loss of yields associated with direct drilling compared with conventional cultivation (which would typically be expected to lead to greater mineralisation) and the potential for forage brassicas to remove high levels of mineral N from the soil during growth. Their research also suggested that total DM yields did not increase with fertiliser N applications in excess of 200kg N/ha.

Research by Carlson et al (2013) also indicated the N losses from grazed winter forage brassicas might be reduced through later season (i.e. late July), rather than earlier season grazing (June), further complemented by ensuring the subsequent crop had the potential to uptake significant amounts of mineral N still in the soil.

### 6.5.4 Grass buffer strips (2-metre) around cropping paddocks

The appropriateness of grass buffer strips of this width is essentially limited in application where there is little risk of surface run-off and they are essentially in place to deliver livestock exclusion from flow paths or stream channels (McKergow et al, 2007). In a cropping context, such width strips are best used for the exclusion of stock from critical source areas whilst grazing forage crops (see Responsible break-feeding practices above). Grassed swales used for controlling overland flow through ephemeral flow paths amongst arable cropping activity should be at a minimum 3m wide shaped into a flat shallow saucer about 0.3m deep (Barber 2014). Grass buffer strips are particularly effective in reducing sediment loss and *E. coli* (Wilcock et al., 2009; Barber 2014; Low et al., 2017).

### 6.5.5 Cover crops between cultivation cycles

Cover crops are usually grown to be ploughed into the soil, but not harvested or grazed, in order to improve soil quality. Cover crops stabilise soil, accumulate nutrients left from previous land uses, improve drainage and soil structure, and can fix nitrogen (for some cover crops). Such cropping practices are suitable for all farm land use practices (Low et al, 2017). The N leaching reduction from cover ranges depending on crop and season and can be about 70-80% reduction from the baseline for cover crop sown in March, and about 25% reduction for cover crop sown in June. The cost of cover crop cultivation is approximately $80/ha, depending on cover crop. However, this land use has some limitations as it might lead to substantial reduction in N leaching for some crops, e.g. barely, while have meagre effect on the whole farm outcomes (Low et al, 2017).

### 6.5.6 Earth decanting bunds for intensive cultivation

An earth decanting bund for intensive cultivation is a temporary berm of compacted soil to create a damming area where ponding can occur (Low et al., 2017). Earth decanting is established along the flat contours at the bottom of paddocks. The paddock can hold the runoff to drop out the sediment by moving the headland further up the paddock (Low et al., 2017). According to Doole (2015) the efficacy in sediment reduction of earth decanting bunds in the Lower Waikato region is 87.5% and its cost is $130/ha.

### 6.5.7 Alum applied to pasture or forage crops

Another option to mitigate P loss is to decrease the source by adding P-sorbing agents such as aluminium sulphate (alum). In cases when alum can bind to the soil before being washed off, it can be effective to decrease P loss. Application of alum to grazed cropland can reduce P loss by 30%, compared to untreated land use and can cost between $160 and $260/kg of P conserved (McDowell, 2010). Alum use on pasture can be effective to reduce P loss by 5 to 30% than under the baseline land use practices, and costs range from $150 to $500 /kg of P conserved (McDowell, 2010). The cost-effectiveness will be influenced by the availability of a ready source of cheap materials. Alum for P loss reduction might be obtained as a by-product from the fertiliser industries.

## 6.6 Access/crossing infrastructure

### 6.6.1 Access crossings, bridges, culverts over all waterways regularly crossed by stock

Surface runoff from farming is a great source of P, sediment load and *E. coli* loss to waterways is considered even to have higher pollution than runoff from pasture (Low et al., 2017). Management requires good track design, bunding of culverts and bridges. Implementation of such mitigation options can help to decrease total P loss in runoff by 95% and suspend sediment by 99% (Low et al., 2017).

### 6.6.2 Appropriate gate, track and race placement, design and maintenance (e.g. diverting effluent away from waterways, slope access tracks away from drains to reduce sediment loss and avoid water flowing across disturbed area)

This essentially comprises the management of critical source areas (with hydraulic connectivity) discussed by McDowell & Srinivasan in 2009.

## 6.7 Fertiliser management

### 6.7.1 Paddock/block-level fertiliser planning/nutrient budget based on soil tests and crop needs

The value of whole farm paddock soil testing is questionable. Withnall (2015) suggests that dairy farms utilising this technique are reducing the range in soil fertility status over their farm (i.e. applying less nutrients to areas of high fertility and more nutrients to areas of low fertility), potentially implying that the incidence of [P] fertility above optimal levels is lowered. However, Edmeades (2011) notes the inherent variability in the soil test results for typically tested nutrients and fertility measures, highlighting the reality that a soil Olsen P measure of 20ppm and 30ppm could both be 25ppm. He suggests that taking soil tests (20 cores from a transect) from blocks of similar soil group, slope, land use, and past management history still represents the best process and cost-efficient method for identifying soil nutrient status.

### 6.7.2 Maintaining optimal soil phosphate levels

Lowering soil Olsen P status provides one of the most powerful mitigations as regards reducing P loss that is quantifiable in OVERSEER. For example, Morton and Roberts (1999) state that near maximum pasture production is achieved at soil Olsen P levels of 38 on pumice soils. However, on rolling contour, soil Olsen P levels of this nature massively increase the risk and extent of P loss. Given both the typical utilization of pasture grazed in situ on dry stock properties and the economic returns from dry stock farming activities, it is questionable as to whether there is an economic return from maintaining soil P reserves at these levels.

Econometric analysis presented by Edmeades in 2008 indicated that the economically optimal soil Olsen P level at a superphosphate price of $400/t can vary between 10 and 24 depending on the level of underlying farm profitability (as expressed in terms of gross margin).

### 6.7.3 Efficient fertiliser use (e.g. not coinciding with rainfall, temperatures below 7 degrees Celsius, appropriate fertiliser types and timing of application, Geographical Positioning System[GPS]-based application).

Analysis of Grafton et al (2011, 2013) infers that at an application rate of 100kg/ha of urea (46%), lowering the coefficient of variance (CV) of spread from 40% to 20% improves the observed DM response rate in pasture from N fertiliser from 10:1 to 11.2:1. This relationship was the basis for the assumption that N fertiliser application can be reduced to 89.2% of pre-precision technology levels without reducing DM production, cow intakes and milk production. Analysis by Perrin Ag (2017b) indicated that for farms of a suitable scale, use of precision fertiliser spreading technology was likely to increase profitability while reducing N losses.

Grafton et al also comment that reduction in CV of spread for superphosphate would reduce risk of accidental discharge into sensitive (i.e. riparian, drainage) areas etc. However, this is not able to be modelled in OVERSEER, nor is there sufficient research to establish whether phosphate fertiliser applications could be reduced as a result of this technology without compromising existing soil P reserves (as measured by Olsen P).

However, adoption of what is generally considered best practice in relation to the application of fertiliser would be expected to reduce the risk of direct nutrient loss to water. Such practices would include applications being undertaken in accordance with the Spreadmark Code of Practice, P fertiliser not be applied if the three-day weather forecast indicates there is likely to be heavy rainfall, avoiding P applications to ephemeral flow paths and during the months of May through August and considering withholding P fertiliser from all significant stock camping areas. Such practices are already encouraged in the guidance documents for the preparation of nutrient management plans required by farmers in the Rotorua Catchment under BOPRC Plan Change 10 and the Farm Environment Plans under the WRC Plan Change 1

### 6.7.4 Reducing N fertiliser use

The use of nitrogenous fertiliser, even when applied in line with best management practices has a contributory impact on increasing nitrogen losses from the farm system. This occurs through both increasing the quantity of N cycling through the farm system and typically allowing higher stock intensities to be farmed, normally through the higher risk winter leaching period. The elimination of N in dairy systems might be managed through the importing of additional feed or the use of gibberellin (see 6.7.5 below). However, in dry stock systems where the returns per kg DM eaten are typically lower than the cost per kg DM of imported feed, it is typically more profitable to lower feed demand (i.e. reduce stock numbers) than increase feed supply (i.e. purchase more feed).

Analysis in the Upper Waikato Drystock Nutrient Study (Perrin Ag, 2013) found that the cessation of fertiliser nitrogen usage, typically accompanied by a reduction in stocking rate, generally led to a reduction in system N losses with no reduction in EBIT. This was typically due to the marginal cost of the N fertiliser exceeding the return from the feed reduced.

### 6.7.5 Use of plant growth regulators (Gibberellic acid)

Gibberellic acid ($GA_3$) is a plant hormone that when applied to grasses and cereals typically results in the elongation of leaf, sheath and stem (a dry matter response), providing the plant has already experienced sufficient vernalisation (chilling) (Bryant 2014). GA is a growth promoter and won't work in the total absence of plant available N in the soil.

Ghani et al (2014) found that the %N in herbage of pastures treated with GA were significantly lower than those untreated which would reduce urinary-N excretion under grazing. Subsequent modelling suggested whole farm annual N losses could be reduced by 4-29%, although some of these reductions would be associated with the replacement of N fertiliser applications with GA (i.e. same DM production for less N applied). Bryant et al 2016 also concluded that using GA to increase DM yield with reduced herbage protein concentration may have reduced environmental impact through reducing N intake of livestock.

Unpublished PhD research from Woods (2017) indicated that in a lysimeter trial the application of GA had no direct impact on reducing N leaching [through promoting plant uptake of urinary N that would have otherwise leached] which suggests that any whole system N loss reduction from the use of GA is associated with the substitution of N fertiliser and an improvement in whole system N use efficiency. However, Bates & Bishop (2016) propose that this lack of N loss reduction was due to the GA being applied to pasture of insufficient mass to promote a response or that conditions were too cold to get any growth at all (Bates et al 2017).

In conjunction with the urease inhibitor NPBT, GA and (if required) and dissolved organic carbon (marketed as ORUN®) is being promoted as a means to increase the lateral movement of urine patches (the NBPT) and then utilise the N in the urine patch before it leaves the rootzone (via the GA), with Bates & Bishop (2016) suggesting targeted application to the actual urine patch is the preferred method.

## 6.8 Irrigation management

### 6.8.1 Efficient irrigation application based on soil moisture deficit monitoring, awareness of soil type/infiltration rate and assessment of crop needs and expected rainfall

Metering the rate and total volume of irrigation water can help to adjust the irrigation application levels and avoid overuse of irrigation water that can increase the leaching of nutrients and bacterial contaminants. Applying water as dictated by soil water needs can have a similar impact. Also, technology can help to avoid poor timing of required irrigation for crops and thus improve crop growth.

## 6.9 Effluent management

### 6.9.1 Solid separation

Separation of the solid fraction from effluent is a mechanism to lower application depths for the liquid fraction of farm dairy effluent. This can allow this liquid fraction to be applied in conjunction with conventional irrigation. This is of significant advantage where effluent volumes are likely to be significant (such as from housing, pads) or contain greater volumes of coarse fibrous material.

Separation of solids may also allow more targeted application of the nutrient in dairy effluent, as total %N is highly associated with the dry matter fraction of dairy effluent (Longhurst et al 2017).

The ability to lower the application rate will be beneficial on higher risk soils [that can't sustain higher application rates in achieving appropriate depths] or where targeted application of the nutrients in solids (such as in cropping programmes) may be more manageable than significant land-based slurry application.

### 6.9.2 Farm Dairy Effluent ponds: sufficient holding capacity to comply with soil moisture application standards and fully lined

If farms have insufficient effluent storage they will be forced to irrigate when soils are actively draining, creating direct losses of nutrients and *E. coli*. While most regional authorities require that effluent is not applied in such conditions, the reality is that many farmers with permitted or consented effluent management facilities are unable to operate with full compliance all of the time.

It is also noted that Houlbrooke et al 2014 identified the losses from old [unlined] two-pond systems that discharge to water as the single largest effluent risk to surface waters, which reinforces the move to eliminate these systems by regional authorities, where they still exist.

Dymond et al (2016) also calculated that where they didn't previously exist, the creation of [lined] storage ponds that allowed irrigation of dairy effluent to land to be time to not coincide with high risk of overland flow would reduce *E. coli* losses to water in the order of 25%.

### 6.9.3 Maize on the effluent block

The main water quality benefit from growing maize for silage on pastoral areas receiving dairy effluent is a reduction in the quantity of fertiliser nutrient required to be applied in the first and potentially second year's crops, which reduces the risk of direct losses to water and lowers the introduction of mobile nutrients into the farm system. There is an expected improvement in farm profitability from doing so as well (FAR 2008, Johnstone et al 2010).

### 6.9.4 Efficient effluent application that complies with soil moisture standards and crop needs, more than 20 metres away from all waterbodies

The depth of applied effluent (measured in mm) should always be less than the soil moisture deficit at the time of application. If effluent irrigation occurs on soils that are too wet, then run off to surface water bodies or drainage below the root zone will occur, with valuable nutrients and also bacteria being lost from the farm and contaminating the environment (Dairy NZ 2014).

Deferred irrigation and low application irrigation systems (e.g. irrigation sprinklers) are effective options to reduce contamination related with land uses. The nutrient losses resulting from a single poorly managed irrigation event is estimated in the order of 12 kg N/ha and 2 kg P/ha, approximately one third of the average total whole farm N losses and three times the annual average pastoral P loss (McDowell, 2010). The potential to decrease nutrient losses with better irrigation techniques is great. Such irrigation techniques can be established based on the agro-ecological conditions such as soil types and climate as well irrigation requirement of crops. Deferred irrigation and low application irrigation systems are not only environmentally beneficial, but also can be cost effective.

### 6.9.5 Increase application area to reduce application concentration

Using N from the fertiliser effluent system to replace N fertiliser is a good mechanism for improving N conversion efficiency on a farm, which will typically result in lower N losses to water. Roach et al (2001) found that nitrate leaching increases significantly when pond FDE is applied at rates above 200 kg N/ha/year and that lowering the application rate to target 100kg FDE N/ha/year (increasing the application area) would deliver maintenance potassium requirements at the same time. The cost-benefit of this will depend on the fertiliser benefit of the additional K and the cost of expansion.

## 6.10 Nitrate inhibition

### 6.10.1 Denitrification technology (e.g. Spikey)

The use of dicyandiamide (DCD) as a means to limit N losses from grazed winter forage crops was successfully demonstrated by Shepherd et al (2012), but due to the presence of DCD found in milk products in 2013, this product is not currently available for use in NZ farming systems. When its use (as described by Shepherd et al) was previously modelled by Perrin Ag (2013) for the Waikato Regional Council, it did introduce a cost to the farm system that wasn't able to be recouped through productivity gains.

However, the "Spikey" technology developed by Pastoral Robotics Ltd (Bates & Bishop 2016), with the ability to detect individual urine patches and then apply an alternative treatment to prevent the rapid conversion of urea to nitrate (see 6.7.5 above) may be as equally effective as blanket DCD application, were it still a viable tool.

### 6.10.2 Denitrification beds

Denitrification beds have application when dealing with point source discharge, like effluent from a farm dairy parlour or a tile drain. Essentially lined containers filled with organic carbon (typically wood chip or coarse sawdust), the wood chips act as an energy source for denitrifying bacteria that convert $NO_3^-$ to N gases. While initial trial work in NZ found a denitrification bed removed the entire N load from dairy effluent (Cameron et al 2010), the applicability of this technology on farm at this juncture is uncertain, given the economic value to the farm system of recycling the N fraction of FDE as a fertiliser and the need to still dispose [to land] of the treated FDE, which will still be high in other nutrients, such as K and P.