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Report for Environment Bay of Plenty

August 2010

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R.W. McDowell

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## Abstract

The loss of phosphorus (P) from land to water is detrimental to surface water quality in many parts of New Zealand and Australia. Farming, especially pasture-based dairying, can be a source of that P. Strategies to mitigate the P loss from farms vary in their effectiveness and costs due to differences in farm management systems, topography, stream density and climate. A range of fully-costed strategies are required to ensure that the best mix of appropriate mitigation measures can be found for each farm. We present a summary of the efficacy and cost of a range of mitigation options available now, or under development, to lessen P loss from grazed grassland farms in the Lake Rotorua catchment. Mitigations are classified as those that; intervene in on-farm management (e.g., optimum soil test P, low solubility P fertiliser, effluent spreading), intervene as amendments (e.g. alum,) and intervene at edge-of-field (in-stream sorbents, buffer strips, constructed wetlands, and dams for water recycling). We considered components of cost-effectiveness such as material, labour and opportunity costs to assess the best options for farms in the Lake Rotorua catchment. Although many P mitigation options are available, an analysis of strategies that target on-farm management options such as dairy effluent and fertilizer management were the most cost-effective way of decreasing P losses (cost beneficial to $65/kg P conserved) although their overall impact is limited. Field amendments that may capture more P than preventing P loss by on-farm management were not as cost-effective ($25-500 /kg P conserved), but better than edge-of-field strategies, such as wetlands ($250->500 /kg P conserved). However, it is important to note that these latter measures often have other benefits, such as N and sediment removal, modification of peak flows, improved biodiversity and mahinga kai areas.

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# 1. Introduction

Practical and effective strategies are required to mitigate the loss of phosphorus from land to surface waters in the Lake Rotorua catchment. Previous work by Rutherford and Timpany (2008) and J. McIntosh (*pers. comm.*) has highlighted both an increase in P concentrations from the 1970's to the 1990's (Table 1) and an enriched concentration of particulate P in the catchment's streams (Figure 1). As part of Environment Bay of Plenty's (EBOP) 5 year P mitigation project, a scoping study will be conducted that:

1) reviews potential P mitigation strategies and their suitability for the Lake Rotorua catchment, and;

2) determines areas, termed critical source areas, that contribute the majority of P loss to surface waters but occupy a minority of the catchment area.

The operational phase of the P mitigation project then:
3) applies the most appropriate mitigation strategy to critical source areas and monitors the effect on P loads in surface waters of the Lake Rotorua catchment.

As part of the scoping study, the following report briefly outlines potential mitigation strategies and their suitability (emphasizing any potential issues and target P fraction) for use in the Lake Rotorua catchment. Phosphorus fractions are defined as dissolved or soluble (P that passes through a 0.45 μm filter) and particulate P (PP; i.e. > 0.45 μm). No mention is made of the relative effectiveness of each mitigation strategy to particular streams. This requires an in-depth analysis of potential flow pathways and critical source areas (areas that account for the majority of P loss) beyond the scope of the current report but follows in the next stage (2) of EBOP's P mitigation project.

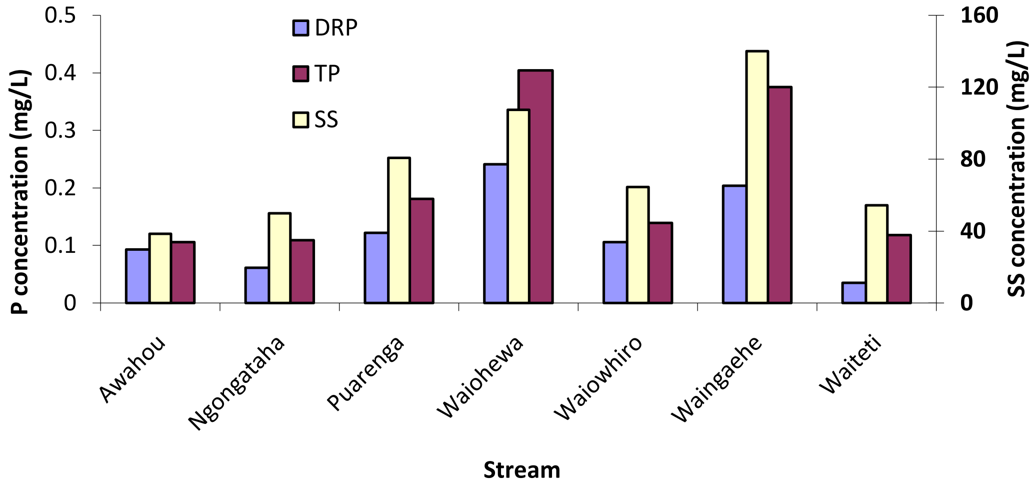


**Figure 1.** Mean concentrations for dissolved reactive P (DRP), total P (TP) and suspended sediment (SS) in 1993-94 for streams draining into Lake Rotorua (J. McIntosh, *pers. comm.*).

**Table 1.** Mean concentrations of total P (g $m^{-3}$) in the 1970's and 1990's (and difference) in streams draining the Lake Rotorua catchment (J. McIntosh, *pers. comm.*).

| Stream | 1970's | 1990's | Difference |
|---|---|---|---|
| Waingaehe | 0.129 | 0.194 | 0.065 |
| Waiohewa | 0.089 | 0.153 | 0.064 |
| Waiowhero | 0.058 | 0.101 | 0.043 |
| Puarenga | 0.083 | 0.098 | 0.015 |
| Ngongotaha | 0.056 | 0.063 | 0.007 |
| Awahou | 0.072 | 0.078 | 0.006 |
| Hamurana | 0.086 | 0.083 | -0.003 |
| Utuhina | 0.076 | 0.070 | -0.006 |
| Waiteti | 0.056 | 0.042 | -0.014 |

## 2. Cost effectiveness

In addition to reviewing the literature, relevant to pastoral farms, an estimate was made of the cost effectiveness of each mitigation strategy using the BMP toolbox (Monaghan,

2009) (Table 2). The percentage effectiveness is taken from the literature outlined in this review and another by McKergow et al. (2007a). For simplicity, strategies are classified as those that intervene in on-farm management (e.g., optimum soil test P, low solubility P fertilizer, effluent spreading), intervene as amendments (such as alum,) and intervene at edge-of-field (in-stream sorbents, buffer strips, constructed wetlands, and dams for water recycling).

**Table 2**. Summary of efficacy and cost of P mitigation strategies

| Strategy | | Effectiveness (%) | Cost (NZD $/kg P conserved) |
|---|---|---|---|
| Optimum soil test P | management | 5-20[1] | highly cost-effective[1] |
| Low solubility P fertilizer | management | 0-20 | 0-30 |
| Stream fencing | management | 10-30 | 5-65 |
| Greater effluent pond storage | management | 10-30 | 30 |
| Low rate effluent application to land | management | 10-30 | 45 |
| Tile drain amendments | amendment | 50 | 25-100 |
| Restricted grazing of cropland | amendment | 30-50 | 150-250 |
| Alum to pasture | amendment | 5-30 | 150->500 |
| Alum to grazed cropland | amendment | 30 | 160-260 |
| Grass buffer strips | edge of field | 0-20 | >250 |
| Sorbents in and near streams | edge of field | 20 | 350 |
| Retention dams / water recycling[2] | edge of field | 10-80 | >500 |
| Constructed wetlands[3] | edge of field | -426-77 | >500 |
| Natural seepage wetlands[3] | edge of field | <10% | >500 |

[1] depends on existing soil test P concentration, but no cost if already in excess of optimum.

[2] upper bound only applicable to retention dams combined with water recycling

[3] potential for wetlands to act as a source of P renders upper estimates for cost infinite.

# 3. Strategies to mitigate P loss to water

## 3.1 Management mitigations

### 3.1.1 Optimum soil test P concentration

*Target:* dissolved and particulate P

*Background:* Of the many methods available to minimise P losses to waterways from New Zealand pastures, the best approach applicable in the Lake Rotorua catchment is to ensure that soil Olsen P is maintained within the range of concentrations considered optimal for pasture production and not excessive for any given soil type. Since the magnitude of P losses from soil via overland or subsurface flow is proportional to soil P concentration (McDowell et al., 2003a; Gillingham and Gray, 2006), having an Olsen P concentration above optimum represents an unnecessary source of P loss and an unnecessary waste of the P inputs (e.g. fertiliser, effluent or dung). To ensure P is not accumulating in soils, a nutrient budget should be used to account for all P sources and to estimate P fertiliser requirements to maintain optimum soil P status (e.g. Wheeler et al., 2006). However, maintaining optimal soil Olsen P does not totally prevent P losses from occurring; moreover, some soils can lose a lot of P at optimal Olsen P concentrations for pasture production (e.g., soils with little Al and Fe oxides such as Podzols; McDowell and Condron, 2004). Furthermore, if a soil is already P-enriched then it can take many years for Olsen P to decline unless soil is cultivated, perhaps during regrassing, to remove surface enrichment and redistribute P within the plough layer (Sharpley et al., 2003). However, while cultivation could decrease P loss, N losses may increase due to mineralisation.

*Potential efficacy:* In pastures grazed by dairy cattle, losses of P from soil account for about 30-50% of total paddock losses (McDowell et al., 2007). Where soils are potentially erosion prone, soil-P losses may account for more (i.e. nearer 50%) paddock P losses compared to stable soils. The soils in the Lake Rotorua catchment vary in their ability to retain P. Redding et al. (2006) examined 28 sites for anion storage capacity (ASC) and Olsen P concentration and found that where ASC was low, enrichment of deeper soil layers (as indicated by Olsen P concentrations) had occurred. The same trend between Olsen P and ASC is relevant for surface runoff losses (McDowell and Condron, 2004). Given the high potential for erosion in BOP catchments, and the direct relationship between soil P and sediment P concentrations in streams (McDowell et al., 2004), ensuring that Olsen P is as low as agronomically possible is a key mitigation strategy.

Environment Bay of Plenty recently (July 2010) faciliated discussion on optimal Olsen P concentrations for soils within the Lake Rotorua catchment with local agriculture and environmental consultants (V Fulton, D Edmeades, S Park, L Matheson, D McNae and M MacIntosh). Consensus was reached on three conclusions:

1. Although there are a range of soil classes in the Rotorua area including Pumice, Podzols and Recent, they all have similar agronomic optimal Olsen P concentration ranges.
2. For dairy farms in the catchment, the economic optimal Olsen P concentration range for typical flat to rolling dairy land is 40-45 mg/L and for steeper areas 35-40 mg/L.
3. For drystock farms the economic optimal Olsen P concentration range varied considerably depending on production targets but generally fitted in the range from 15-30 mg/L.

### 3.1.2 Low solubility P fertiliser

*Target:* dissolved P (and via soil enrichment - particulate P)

*Background:* McDowell *et al*. (2007) estimated that losses from superphosphate fertiliser was about 10% of total farm losses under best practice, and which required superphosphate to be applied during a period when surface runoff was unlikely. This is because orthophosphate contained within superphosphate is highly water soluble and while it is sitting on the soil surface diffusing into the soil solution, and eventually the soil matrix, there is a high risk of P loss should surface runoff occur. Several authors have defined this period to last from 7 to 60 days (with a mean of 21-days) depending on the soil type and climatic conditions (e.g., McDowell *et al*., 2003b; Nash *et al*., 2004; Sharpley and Syers, 1979). McDowell and Catto (2005) demonstrated that the availability of P to surface runoff from fertilisers was directly related to the fertiliser's water soluble P concentration. Among P fertilisers, reactive phosphate rock (RPR; phosphate rocks containing calcite and dolomite) has little water soluble P, but a similar or greater total P concentration to single superphosphate (Zapata and Roy, 2004). In areas where there is much surface runoff, low water solubility may prevent the potential sudden loss to water evident soon after applying highly water soluble P fertilisers like superphosphate. The use of RPR instead of superphosphate was shown to decrease P losses from hump and hollow land grazed by dairy on the West Coast (McDowell, 2010) and from a 12 ha catchment in the Hawke's Bay (McDowell et al., 2010).

*Potential efficacy:* On an agronomic basis, in areas where annual rainfall is > 800 mm and soil pH is < 6, RPR can produce pasture as well as superphosphate when maintaining or decreasing soil test P concentrations (Sinclair *et al*., 1990). However, there is a lead in time where plant available P increases by a third per annum so that RPR performs the same as superphosphate by year 4. This, coupled with restrictions where the product could be used may deter its use. Although only a small component of total P loss, the cost benefit analysis for low water soluble P fertilisers is favourable compared to other mitigation strategies (Table 2).

### 3.1.3 Stream fencing

*Target:* dissolved and particulate P

*Background:* Livestock access to streams causes damage to the stream bank and bed and allows for the direct deposition of excreta into stream water. McKergow and Hudson (2007) classified key factors in influencing the amount of damage as 1) livestock management in riparian areas (mob stocking or strip grazing) and 2) soil type, topography and climate. They also concluded that damage and loss of aquatic habitat was likely at stocking rates above 4 stock units/ha (sheep or cattle). In addition to anecdotal evidence, overseas data has shown fencing-off streams from dairy cattle can decrease P loads in streams. For example, James et al. (2007) reported a 32% decrease when fencing occurred in the Cannonsville catchment in New York State. Line (2002) also showed that the dominant form of P lost from a grazed catchment changed from particulate-P to dissolved-P once fencing had occurred. Additional New Zealand data has reported decreases of up to 90% when red deer were excluded from streams (McDowell, 2008).

In an effort to increase the adoption of fencing as a mitigation strategy, Fonterra, the Ministry for the Environment, the Ministry of Agriculture and Forestry and Local Government New Zealand (on behalf of Regional Councils) signed the ‘Dairying and Clean Streams Accord’ in 2003. One of the stated performance targets within the Accord was that “dairy cattle would be excluded from 50% of streams, river and lakes by 2007, and 90% by 2012”. Reporting on the accord to date shows there has been a progressive increase in the percentage of farms compliant with the scheme’s targets: 78% of cattle are now excluded from waterways on farms; 99% of farms now have a system in place to manage nutrient inputs and outputs; and only 2% of Accord-defined crossings remain unbridged or unculverted (Figure 2; Ministry of Agriculture and Forestry, 2009). The importance of bridges and culvert are covered later. While targets are being met, others have questioned both the method of reporting and auditing and whether or not the accord has had any effect on water quality (Deans and Hackwell, 2008). Indeed, in 4 out of 5 predominantly dairying catchments that have been monitored and best management practices encouraged since 2001, surface water quality indicators such as nutrient and faecal indicator bacteria concentrations have neither improved nor deteriorated (Wilcock et al., 2007). However, this must be seen against a background of increasing land use intensification and milk production within the catchments.

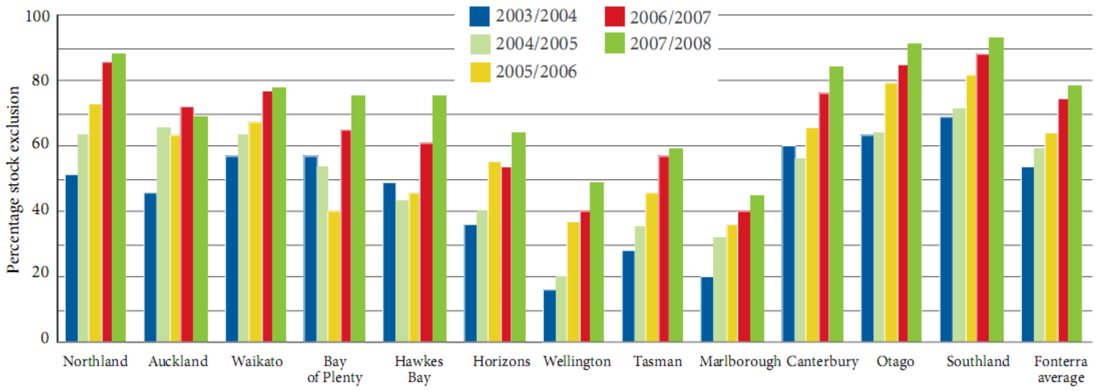


**Figure 2**. Relative percentages of surveyed farms within regions with Accord-type (wider than a stride and deeper than a red band) waterways fenced-off from stock between 2003 and 2008 (Ministry of Agriculture and Forestry, 2009)

In an effort to improve the water quality of areas too costly to fence or those that do not fit within the rules of the accord, strategies are being trialled to deter stock from using streams as water sources or for recreation. Provision of water troughs, shade and shelter are often thought to benefit water quality, but data has been mixed. For example, studies in the US have indicated that the time spent near a stream and stream loads of dissolved and total P were decreased when water troughs and shade was provided (Byers et al., 2005), but work in New Zealand indicates that the provision of a water trough did not deter cattle from using stream or riparian areas in the Waikato (Bagshaw et al., 2008).

While protection of Accord-type (wider than a stride and deeper than a red band) waterways are close to 100% achieved in the Lake Rotorua catchment, a mid-summer survey in 2007/2008 of one quadrant of the lake's catchment (AG Gowing survey - J. Paterson pers. comms.) found 10.4km of small unprotected permanently flowing waterways (narrower than a stride and shallower than a red band).

*Potential efficacy:* If the accord reaches 100% it is still questionable whether or not the decreases seen in the US (e.g. James et al., 2007) or when red deer are excluded from wallowing systems will be achieved. This is due to the exclusion in the accord from fencing of any stream “wider than a stride and deeper than a red band” i.e. most first order and ephemeral channels (both excluded in the studies of McDowell, 2008 and James et al., 2007). Fencing these areas varies from a cost of $5/kg P for a single wire to $75/kg P for a post and batten fence, but may pose a challenge in rolling country.

### 3.1.4 Greater effluent pond storage and low rate effluent application

*Target:* dissolved and particulate P

*Background:* Recent research has identified that the risk of waterway contaminantion via land application of FDE (dairy shed effluent, or farm dairy effluent) is greater on soils with: a high degree of preferential flow; artificial drainage or a coarse structure; either an infiltration or drainage impediment; or when applied to soils on rolling/sloping country (Houlbrooke et al., 2006, Monaghan and Smith, 2004). Accordingly, more recent research efforts have focussed on improved methods of scheduling and applying effluent to these sensitive soil types. Best management practices such as deferred irrigation (pond storage of FDE during periods when soil moisture is close to or at field capacity) and low application rate irrigation systems (movable irrigation sprinklers that apply FDE at rates < 10 mm/hr) have proven effective at decreasing or eliminating the direct contribution of FDE to contaminant losses associated with land application (Houlbrooke et al., 2006, Houlbrooke et al., 2004, Monaghan and Smith 2004). Research conducted in the Manawatu on a soil with inherent high risk for direct FDE loss (mole and pipe drained Pallic Tokomaru silt loam) demonstrated the potential importance for avoiding wet applications of FDE (Houlbrooke et al., 2008). Figure 3 compares the FDE and system (rest of the farm) loss of N and P from a ‘one-off’ poorly timed event under “standard practice” with wet soil conditions, compared with measured annual losses of N and P when using a deferred irrigation strategy. These nutrient losses from a single, badly-managed irrigation (12 kg N/ha and 2 kg P/ha) event are considerable, particularly when compared to measured annual losses under dairy farming without any influence of FDE application (31.4 kg N/ha/yr and 0.65 kg P/ha/yr). The uptake of best management practices such as deferred irrigation and low FDE application rate tools has been high in the dairy regions of Southland and Otago where many soil and landscape conditions have made it difficult to keep effluent nutrients in the root zone using traditional effluent management practices. Uptake of deferred irrigation in the Lake Rotorua catchment is low, probably less than 75% of diary farms have storage (J Paterson pers. comm.) and non-compliance with FDE consent conditions was close to 50% during a spring compliance check made in August 2009 (S. Pickles pers. comm.).

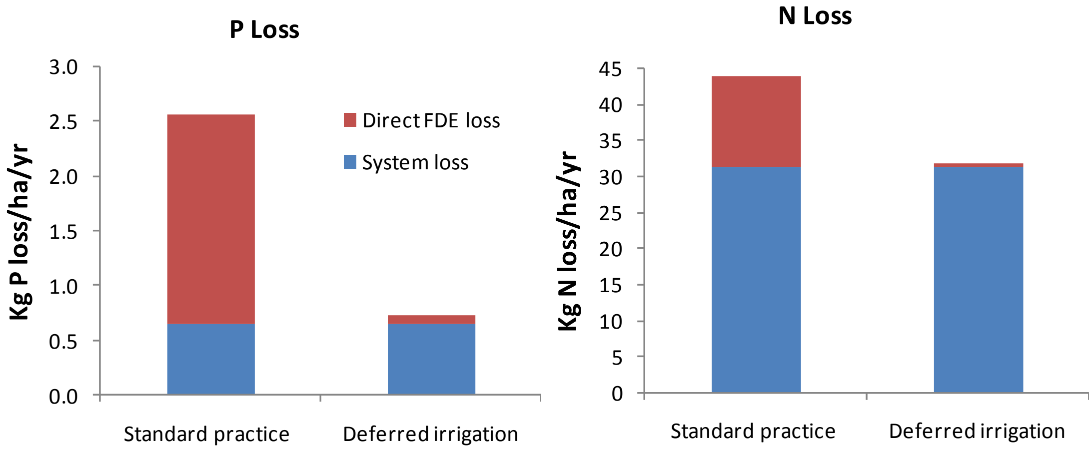


**Figure 3**. Comparison of FDE and system N and P loss from a single poorly timed application of FDE under "standard practice" and under deferred irrigation.

*Potential efficacy*: The potential to decrease P losses with better effluent utilisation and application is great. Methods to do this are not only environmentally beneficial, but also cost effective. Recent publications to aid the calculation of effluent storage and the application of FDE are available (Horne et al., 2010; Houlbrooke and Monaghan, 2010). This calculator model is now being used in the Bay of Plenty by consents staff and consultants to support decisions on FDE storage and application (EBOP, 2010).

## 3.2 Amendments

### 3.2.1 Tile drain amendments

*Target:* particulate and dissolved P (depending on soil P and effluent applications)

*Background:* Several studies have shown that soils rich in P have elevated P concentrations in drainage water (e.g., Sharpley et al., 1994; Sims et al., 1998). Furthermore if a soil has a large number of macropores then these can act as a direct conduit for P loss to tile drains and drainage water. This is thought to be a major mechanism behind the direct loss in drainage of dairy shed effluent irrigated onto pastures, and can be especially problematic when the soil is either already wet and the likelihood of the soil matrix retaining added water and P is limited (Monaghan et al., 2002) or the soil is prone to cracking when dry. Consequently, some P loss in tile-drained land is inevitable and means that a cheap and effective method of mopping-up P in drainage water is desirable.

Over the past two decades, by-product materials rich in P-sorptive Ca, Al and Fe have been identified as decreasing P loss from soils with varied success (Vlahos et al., 1989; O'Reilly and Sims, 1995; Haustein et al., 2000; Stout et al., 2000). These include, but are not limited to, zeolites, aluminium sulphate, water treatment residuals, and fluidized bed bottom-ash and fly ash from coal fired power plants (e.g., Reichert and Norton, 1994; Sakadevan and Bavor, 1998; Moreno et al., 2001; Callahan et al., 2002). McDowell et al. (2008) tested a number of materials (fly ash, melter and basic steel slag) according to (1) cost of the material (2) toxicity to the environment, and (3) efficacy of P sequestering. The idea was to use them as a backfill for tile drains, intercepting P en route. A field site was chosen and compared a mixture of steel melter slag (90%) and basic slag against greywacke both as backfill. Mean dissolved reactive P (DRP) and total P (TP) concentrations from greywacke backfilled drains were 0.33 and 1.20 mg $L^{-1}$, respectively. In contrast, slag backfilled drains had DRP and TP concentrations significantly ($P < 0.05$) less at 0.09 and 0.36 mg $L^{-1}$, respectively. Loads of DRP and TP in greywacke drains (0.45 and 1.92, respectively) were also significantly greater ($P < 0.05$) than those from slag drains (0.18 and 0.85, respectively). Backfilling with slag decreased P loss in drainage compared with greywacke backfill. Data from a farm where melter slag was used as a backfill, suggested that slag would have a life expectancy of about 25 years. Similar work by Hanly et al. (2008) tested the efficacy of 1-4 mm diameter volcanic tephra as a fill for mole channels and as a backfill. During the winter drainage season, P losses were decreased from a paddock grazed by dairy cattle by 45 and 47%, respectively compared to a standard mole and pipe drainage system.

*Potential efficacy:* The cost of this mitigation is prohibitive to widespread use due to shipping costs from the source in South Auckland. Other untested materials or local origin material may work. However, where P-sorptive materials like steel slag is not available, some P loss could be prevented by installing drains about 10 cm from the bottom of the fill, to allow particle settling, decreasing the potential for drain blockage and increasing effective life span. It should also be noted that the main impediment for the use of this technology in the Lake Rotorua catchment is the relatively small area of poorly drained soil and hence presence of artificial drains (J. Paterson *pers. comm.*).

Tile drainage may be a solution to the issue of ephemeral channels if drainage is accompanied by cultivation to redistribute P in the plough layer and break the continuity between topsoil and drain via macropores.

### 3.2.2 Restricted grazing of cropland

*Target:* mostly particulate P

*Background:* The only data on P losses associated with grazed cropland comes from work done in North and South Otago on winter forage crops (McDowell et al., 2003c; 2005; McDowell and Houlbrooke, 2009). This demonstrated a number of key points:

1. The loss of P was much greater from forage cropland than pasture;
2. Losses were dependant on soil moisture status and soil type – wetter soils lost more P, but if near their plastic limit, hoof prints prevent the loss of particulate P via settling out of sediment during runoff;
3. Losses were greatly influenced by dung deposition independent of soil moisture, and therefore a factor of stocking rate and grazing duration.

The mitigation strategy was to allow cattle (and sheep) to graze for 3-4 hours and ingest their maintenance feed requirement. This had the effect of decreasing treading damage and the number of dung deposits on the soil surface. The strategy decreased annual P losses in both dissolved and particulate forms from the irrigated site by about a third (Figure 4).

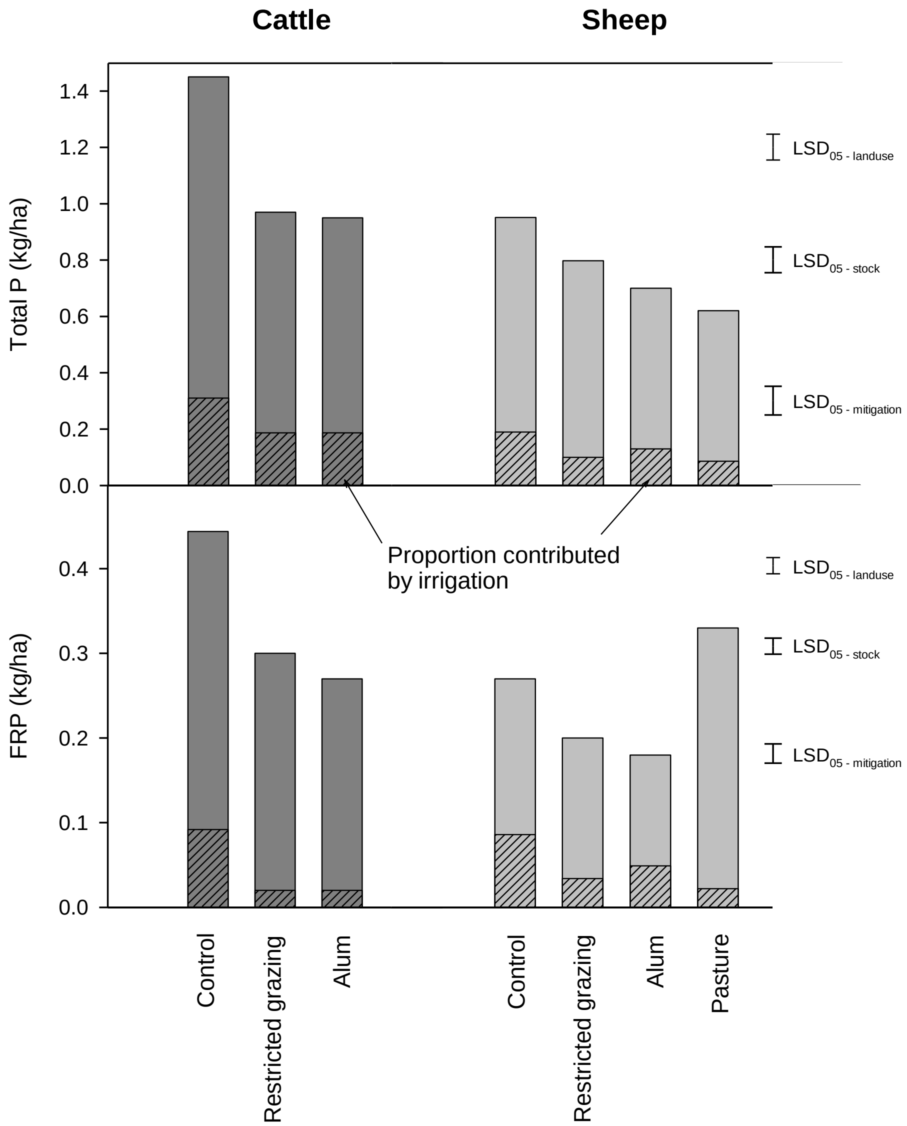


**Figure 4**. Annual P losses associated with the grazing of winter forage crops from an irrigated (P losses due to irrigation-induced runoff represented by hatched bars) site in North Otago. Data from McDowell and Houlbrooke (2009).

*Potential efficacy:* Although data applies only for grazing of winter forage crops, the key points should also apply to summer forage crops i.e. decreasing time and dung deposits. The need to have a feed or stand off area to accompany restricted grazing of forage crops in winter will not be required in summer where cows can be fed pasture. This should increase the cost-effectiveness of the mitigation for summer crops relative to the cost for winter crops indicated in Table 2.

### 3.2.3 Application of alum to pasture and cropland

*Target:* dissolved P

*Background:* One possibility to mitigate P loss is to decrease the source by adding P-sorbing agents. These agents increase the sorption capacity of the soil and decrease solubility in dung. Of several agents that have been trialled, such as poly-DADMAC (Churchman, 2002), gypsum (Cox *et al*., 2005), ferrisulphate (Schoumans *et al*., 1995), polyacrylamide (Mason *et al*., 2005), one of the most studied, effective, cheapest and readily available is aluminium sulphate (alum). Alum has been used around the world to flocculate P out of water columns (e.g., adding to lakes and reservoirs), and in the US to decrease the water solubility of P in manures added to land (Smith *et al*., 2001). Moore and Edwards (2007) monitored two catchments since 1995 and found that in one with poultry litter applied, much more P (1.5 kg P/ha/yr) was lost than from an adjacent catchment with applied poultry litter amended with alum (0.45 kg P/ha/yr). Further work has indicated that pasture growth was unaffected by alum application (Warren *et al*., 2006), and that ingestion at the rates used (10-40 kg Al/ha/yr) would be unlikely to impair animal performance (Mora *et al*., 2006; Moore and Edwards, 2007).

Work in New Zealand has focused on the use of alum following the grazing of a winter forage crop (McDowell and Houlbrooke, 2009) and application to grazed pasture on the West Coast (McDowell, 2010). Annual losses associated with application after grazing a winter forage crop were equivalent to those lost via restricted grazing, a third less than unrestricted grazing of the same site (Figure 4). However, a similar application of 20 kg Al $ha^{-1}$ to pasture on the West Coast did not significantly decrease P losses. This most likely due to Al being washed-off in surface runoff (annual rainfall 4-5 m and runoff of 1-2.2 m) before it could bind to the soil.

*Potential efficacy:* In situations where alum is allowed to bind to the soil before being washed off it has proven to be effective at decreasing P losses. The data in Table 2 was calculated using forage crop data. However, additional unpublished data (R. McDowell) has compared the application of alum to 5 different soils, including a Pumice soil. Data indicated that P losses were decreased by about 30-50% compared to untreated soil for at least 40 days after application (no data was available again until 110 days after application). Although this suggests that the strategy may be of use in the Rotorua lakes region the cost-effectiveness will be dictated by a ready source of cheap material. This may become available as a by-product of the fertiliser industry.

### 3.3 Edge of field mitigations

#### 3.3.1 Buffer strips

*Target:* particulate-P

*Background:* Grass buffer strips work to decrease P loss in surface runoff by a combination of filtering by vegetation, deposition onto filter strip soil and improving infiltration (Dillaha et al., 1989). The upslope edge of the strip functions to decrease the speed of surface runoff decreasing its ability to transport particulate-P. If the speed of surface runoff is decreased enough then deposition will occur. Furthermore, the upslope edge is also where most large particles and associated particulate-P are filtered. However, the main mechanism of a buffer strip is to increase the likelihood of infiltration, allowing deposition of particulate material on the soil surface or vegetation and transporting dissolved materials into the soil (Dosskey, 2001).

Grass buffer strips do have major flaws; 1) the strip can quickly become clogged with sediment, 2) strips only work in areas where upslope surface runoff is generated by infiltration-excess, they do not work in areas where surface runoff occurs due to saturation-excess conditions in, for example, near stream areas (the benefit is gained from fencing-off, not the strip itself), and 3) strips only function well under sheet flow, whereas most surface runoff tends to converge into small channels that can bypass or inundate strips (Verstraeten et al., 2006).

*Potential efficacy:* Riparian and grass buffer strips have been widely employed and trialled in the Waikato and Rotorua Lakes catchment and have proven effective at mitigating P losses. For example, Smith (1989) established buffer strips on a Waikato drystock farm and noted a 40-50% decrease in flow-weighted P concentration, but concluded that the effect was due to a decreased quantity of P in the strips due to fencing off. To improve their performance, Redding et al. (2008) applied water treatment residual (largely alum) and polyacrylamide and increased P retention within a 2.1 m wide strip by up to 40%. However, channelisation occurs readily in the catchment and can quickly lead to erosion. To avoid this, Longhurst (2009) trialled filter strips with spreaders and sediment traps in an ephemeral channel and noted a small (5-20%) decrease in P losses. McKergow et al. (2007b) trialled the installation of grass buffer strips within, not at the edge of, paddocks. Two surface runoff events were monitored from fenced-off areas and exhibited a 40% decrease in P losses compared to a grazed control. Trials with a number of less palatable grass species to create more persistent swards for use in ephemeral areas in Rotorua (Guinto 2008) did not identify grass species that can significantly resist grazing by dairy cows. The concept also has practical impediments such as a loss of production and the cost associated with creating the strips will limit their adoption.

### 3.3.2 Sorbents in and near streams

*Target:* Particulate- and dissolved-P

*Background*: Management practices to decrease P in stream flow are limited. Many techniques such as dosing the stream with materials such as alum ($Al_2(SO_4)_3$) or altered bentonite clays (e.g., Phoslock®) to sorb P rely on P attached to the sorbent/carrier remaining on the stream bed or for the material to cap the bed and block P dissolution from sediment (Cooke et al., 1993; Wrigely, 2001; Robb et al., 2003; Steinman et al., 2004). This may not occur, as materials can be lost downstream during high flow events unless the material is very heavy. Furthermore, the input and deposition of new P-rich sediment negates the cap’s effectiveness (Moore et al., 1998). Following a survey of 57 P sorbing materials, Drizo et al. (1999) and later McDowell (2004) chose steel melter slag. The material was non-toxic (neutral pH), exhibited good stability (i.e. did not breakdown readily in turbulent conditions), and was heavy enough to remain stationary should the packaging material break. The material has been used successfully in wastewater filter beds (Shilton et al., 2006) and has been tested as backfill for tile drains (McDowell et al., 2008).

Encasing steel melter slag in a mesh, McDowell et al. (2007) installed “P socks” on the stream bed of the Mangakino stream in the Lake Rerewhakaaitu catchment (Figure 5). 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 removal strategy at low flows, relatively little P was retained at flow rates > 20 litres $s^{-1}$ (just above baseflow). When a cost-benefit analysis was done, the technology proved to be expensive compared to alum dosing (Paul et al., 2008) due to transport costs. Hence, this approach may be limited to small, slow-flowing waterways.

Near stream areas and areas connected to the stream are important sources of P loss to most waterways. Work by Hively et al. (2005) and Lucci et al. (2010) found that the potential for P loss from areas such as gateways, lanes and around barns and troughs was much greater than from the rest of a grazed paddock. McDowell and Srinivasan (2009) also confirmed that when connected to a stream these areas are important sources of P year round. Runoff from such areas usually occurs via infiltration-excess conditions and as such represents the dominant source of P during summer-autumn months when waterways are at most risk from algal growth.

*Potential efficacy*: The importance of tracks and lanes was emphasized within the study of McDowell et al. (2007) who traced 90% of the P loss back to runoff from a crossing where daily traffic to and from the milking parlour resulted in regular dung deposition i.e. a critical source area. The installation of slag on the side of the lane (Figure 5) resulted in >90% of PP prevented from entering a tributary of the Mangakino stream over a period of 1 year (Table 3). Some of this decrease would have been due to the different contributing areas (600 vs. 1000 $m^2$) yielding different runoff volumes, but concentrations mirrored loads implying most of the difference was caused by treatment not design effects.  The cost-effectiveness of using a material such as steel melter slag in strategic locations is much better than installation of P socks.

**Figure 5**. Picture showing the P socks installed on the bed of the Mangakino stream (left) and one half of the race with altered steel melter slag and tipping bucket installed (right). Note the Mangakino stream tributary flows through a culvert located below the tipping bucket.

**Table 3**. Loads of runoff, P fractions, sediment and *E. coli* in the slag (1000 $m^2$) and control (600 $m^2$) races and the percentage mitigation (i.e. the fraction of load from altered steel melter slag vs. control races).

| Parameter | Mean load for altered steel melter slag | Mean load for control | Percent decrease |
|---|---|---|---|
| Runoff (L) | 15665 | 35950 | 56 |
| DRP (g) | 1.5 | 19.8 | 93 |
| DOP (g) | 2.4 | 2.6 | 8 |
| PP (g) | 3.6 | 136.9 | 97 |
| TP (g) | 7.4 | 159.3 | 95 |
| SS (kg) | 2.0 | 173.0 | 99 |
| *E. coli* (cfu) | $2.22 \times 10^8$ | $1.54 \times 10^9$ | 86 |

### 3.3.3 Retention dams

*Target:* particulate-P and some dissolved-P

*Background:* Sediment traps are useful for the retention of coarse sized sediment. Hudson (2002) concluded that as a "rule or thumb" a coarse sediment trap should be 1.5 times the channel width, 10 times the trap width – long and excavated 1.5m below the existing channel bed. If cleaned twice a year, depending on sediment load, this would remove up to 90% of fine sand. For systems receiving silt-sized particles the guidelines is that the ratio of volume to inflowing water should be between 0.1 and 1 with a residence time of at least 1 day (Griffin, 1979). Little data is available in New Zealand on the performance of coarse sediment traps for retaining P. However, McDowell et al. (2006) noted a 10% decrease in total P in water exiting a trap compared to sediment rich inflowing water from a stream draining red deer wallows. The removal rate is perhaps lower than expected due to the greater enrichment of P in fine as opposed to coarse sediment (Stone and Murdoch, 1989). While potentially beneficial for sediment and P removal, it was noted by Maxted et al. (2005) that the inclusion of 6 small ponds in a stream in the Auckland region caused an increase in particulate-P (exported as algae) and dissolved oxygen levels that were detrimental to aquatic life.

An alternative strategy may involve coupling a retention dam with the reuse of water on the farm. Examples of this strategy are plentiful in Australia and often result in farms that have nearly zero P loss. For example, Barlow et al. (2005) studied P loss from an irrigated farm in south-eastern Australia and although P losses were up to 23 kg

P/ha/y from field plots effective reuse of this water via a retention dam decreased P loss by 98%.

As part of a reuse system, or feeding a retention dam, artificial drains have been found to be both a source and sink of P. For example, Nguyen et al. (2002) found that when a solution containing P was injected into a farm drain, the P concentration decrease by 56% over a distance of 150m. However, Barlow et al. (2003) found that P increase by 4.4 mg P/L down a pasture-based drain, but decreased by 1.2 mg P/L down an earthern drain. Management such as drain clearing can cause significant, but temporary, increases in P concentrations (Smith et al., 2006). Until further work is done, manipulation of a drainage network cannot be relied upon as a strategy to mitigate P loss.

*Potential efficacy:* The installation of both retention dams and water reuse systems require considerable capital and maintenance which decrease their cost-effectiveness (Table 2). However, if a water harvesting system coupled with a reuse system can be installed on farm it has potential to mitigate most P loss.

### 3.3.4 Natural and Constructed Wetlands

*Target:* particulate-P

*Background:* Wetlands, depending on factors such as loading rates and layout, can be sinks and sources of P (Reddy et al., 1999). The retention of particulate-P associated with sediment is usually large, especially if the input is sediment rich (e.g. from cropland). However, over time the ability of wetlands to retain particulate-P decreases as the wetland becomes choked with sediment. Compared to particulate-P, the retention of dissolved P by wetlands tends to be poor, often requiring large areas (to maximise residence times) and a P sorptive substrate. However, processes that retain particulate-P and dissolved-P can be antagonistic if for example, P-rich sediment is retained within the wetland, it will act to desorb and be a source of dissolved P. Several examples exist whereby concentrations of dissolved P (and total P over time) are greater exiting than entering a wetland (Tanner et al., 2005; Sukias et al., 2006).

Compared to natural wetlands, constructed wetlands can be designed to remove P from waterways by: 1) decreasing flow rates and increasing contact with vegetation – thereby encouraging sedimentation; 2) improving contact between inflowing water, sediment and biofilms to encourage P uptake; and 3) creating anoxic and aerobic zones to encourage bacterial processing. However, three wetlands that have been constructed to date (1 each in the Waikato, Northland and Southland) exhibit little to no uptake of P due to sediment poor inflow and largely anoxic conditions (Sukias et al., 2006). The inclusion of a P-sorptive material, insensitive to changes in redox conditions may aid in P removal. Ballantine and Tanner (2010) reviewed a number of materials for inclusion in constructed wetlands ranging from tephra to sand, but concluded that porous materials shell-sand or materials enriched with Al or Fe like melter slag were the best candidates. Monitoring over 10 years at the Waiuku wastewater treatment plant has seen < 70% of P retained by melter slag during the first 5 years of operation (Shilton et al., 2006). Tanner (2001) notes that while the regular harvesting and removal of emergent plants may increase P removal, the P is usually taken from sediment and unless the biomass has an economic value, harvesting is not a cost-effective strategy.

An extension of constructed wetlands has seen the inclusion of scrubbers to periodically remove the algal turf (periphyton) and associated P. Craggs et al. (1996) noted removal rates of 0.44 g P/m$^2$/d from wastewater. However, removal rates of 0.12 g P/m$^2$/d have been noted for less enriched agricultural runoff (Adey et al., 1993). The diversion of stream water through a watercress bed was found to remove 33% of DRP inflow (0.016 g P/m$^2$/d) at low flow and 16% of DRP inflow (0.074 g P/m$^2$/d) at high flow (probably due to sediment trapping) (Sukias and McKergow, 2010). Another adaptation has seen the inclusion of floating wetlands (emerging wetland plants grown hydroponically on floating mats) remove 20-51% of DRP from artificial urban stormwater compared to unplanted mats (Headley and Tanner, 2007).

*Potential efficacy:* A constructed wetland has been installed in the stream feeding Lake Okaro (Tanner et al., 2007). The likely lifespan that the wetland will continue to remove P is estimated to be > 15 years, beyond which it may act as a source. Effectiveness results have initially been encouraging (Hudson et al., 2009) with 47% of TN and 50% of TP removed. However, little data is available to determine its long-term effectiveness particularly with P removal as the wetland matures and co-benefits for biodiversity and mahinga kai area has not yet been assessed. Due to the large area required, the cost-effectiveness of a constructed wetland is poor (Table 2) compared to other, more targeted, mitigation strategies.

## 4. Conclusions

Mitigations that are either farm management strategies, are amendments such as alum, or are those that act after P has left the field, vary in effectiveness. After also considering components of cost-effectiveness such as material, labour and opportunity costs, strategies that target on-farm management options such as best practice with effluent and fertilizer management were the most cost-effective way of decreasing P losses although their overall impact is limited. Field amendments that may capture more P than preventing P loss by on-farm management were not as cost-effective (25-500 $/kg P conserved), but better than edge-of-field strategies, such as wetlands (250->500 $/kg P conserved). However, it is important to note that while decisions can be made on a purely $/kg P conserved, farm and environmental decisions should cast a wider net and that mitigation strategies may have more relevance in some areas (such as critical source areas) or have multiple benefits. For example, wetlands have other benefits, such as N removal and/or modification of peak flows. The cost-effectiveness of some of these mitigation strategies will also be improved by targeting their application to certain areas in the catchment. These areas, termed critical source areas, account for the majority of P loss but occupy only a small part of the farm or catchment. They will be the subject of further work to incorporate P mitigation within a wider farm systems approach to improve their mapping and the selection of an appropriate mitigation strategy.

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