

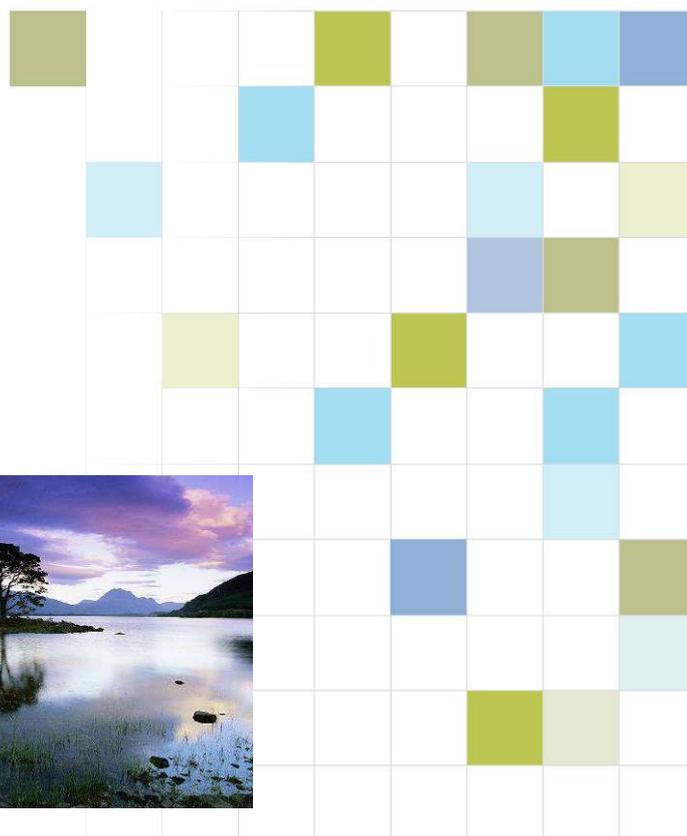
A Commentary on Agricultural Sources, Transport and Impact of Phosphorus in Surface Waters: Knowledge Gaps and Frequently Asked Questions



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A Commentary on Agricultural Sources, Transport and Impact of Phosphorus in Surface Waters: Knowledge Gaps and Frequently Asked Questions

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December 2008

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1. Abstract

National research has shown that both nitrogen and phosphorus contribute to eutrophication, but is limited by phosphorus in more cases than nitrogen. Following concerns about incomplete understanding about phosphorus, a workshop was convened to present the current state of knowledge to regional councils on the sources, transport and impact of phosphorus on surface waters in New Zealand. The workshop highlighted that knowledge about sources of phosphorus (soil, fertiliser, dung and effluent, grazing and treading, lanes and stock camp areas, direct stock access to streams) and their relative importance is progressing. There is also considerable research on transport processes and some in-stream processes, but there remain significant knowledge gaps regarding the specific linkages to impacts, in particular the timing and sources of phosphorus to in-stream plants at low flows when excessive growths are typically a problem. Research should focus on understanding these linkages between P loss and periphyton growth and on effective management strategies and policy to mitigate the deleterious effects of P losses and eutrophication.

Keywords phosphorus; agriculture; best management practise; eutrophication; grazing; lake; periphyton; stream

2. Introduction

Concern about eutrophication has focused much national attention on increasing diffuse nutrient pollution arising from land use intensification over the past decade (PCE, 2004). Recent studies have shown both nitrogen and phosphorus contribute and limit the growth of periphyton (Wilcock et al., 2007). Under ideal growing conditions (e.g., light, temperature and flow), nutrient limitation can change between nitrogen and phosphorus over short time (e.g., hours) and spatial (e.g., km) scales. However, a recent analysis of New Zealand rivers and streams indicated that on average, more sites were limited by phosphorus than nitrogen (McDowell and Larned, 2008; appendix I). For lakes, a similar picture is emerging, but should be interpreted with caution as this also includes an interaction with depth (Ministry for the Environment 2007; Fig. 1).

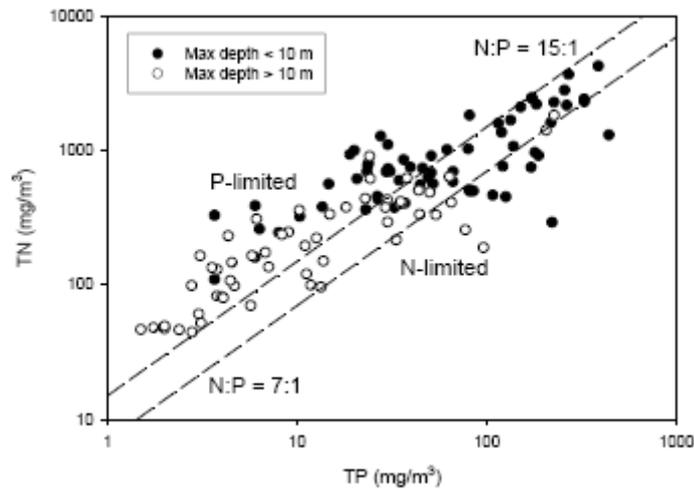


Fig. 1 Plot of total N (TN) against total P (TP) for samples taken within lakes shallower or deeper than 10 m. Data points above the 15:1 line (N:P mg/m³) are P-limited, while data below the 7:1 line (N:P mg/m³) are N-limited and those between the two lines are Co-limited. Plot courtesy of the Ministry for the Environment (2007).

There is also much uncertainty amongst scientists and staff from regulatory and industry groups concerning the sources, transport and impact of P. To inform these groups and present recent P research in New Zealand, a workshop was held in July 2008. The intent of the workshop was to inform the workshop participants of the state of knowledge on P stocks, transport and ecological effects. The workshop was structured in two parts - the first was a New Zealand-centric overview of sources, transport processes and impacts. This involved the presentations listed in Appendix II by R McDowell (AgResearch – Importance of Phosphorus to NZ Surface Water Quality), L Condon (Lincoln University – Cycling and efficiency of phosphorus transformations in New Zealand soils), D Nash (Victorian Department of Primary Industries - Components of a Water Quality Problem), S Larned (NIWA - Nutrient Dynamics and Nutrient-Limitation in the Selwyn River Catchment) and K McArthur et al. (Horizons Regional Council - Phosphorus Research: the Horizons Story). The second part was a discussion based on questions submitted by Scientists from Regional Councils and Crown Research Institutes, the Fertiliser Industry and various Pastoral sectors before the workshop. The present report is a synopsis of both parts (see also appendix II). Although drafted by the authors, this paper has been circulated widely to incorporate current opinions and knowledge.

3. Sources

The sources of P in most catchments are non-point and agricultural in origin (Foy & Withers 1995). Information and understanding about the locations, contributions and management of point sources is much further developed than for non-point sources. However, considerable progress has been made recently in determining the relative importance of a number of P sources within farms. The following section is a summary of research results concerning the potential contributions of each source to total P losses from farmland. Interactions between each source and the hydrological processes that affect P transport are discussed in the following section.

3.1 Soil

Soil is not only a medium that supplies plants with P, it is also the principle source of P to overland and sub-surface flowpaths and to surface water bodies. Rates of soil P loss depend upon flow rates and supply rates (i.e., detachment rates for particulate P and dissolution rates for dissolved P). Dissolved P is generally defined as the inorganic (termed reactive in reference to colorimetric analyses; largely orthophosphate) and organic P that passes through a 0.45 µm filter; the fraction greater than 0.45 µm is defined as particulate P.

When exposed to flow rates of low kinetic energy, most soil P is lost in dissolved form (Sharpley et al. 1992). When this happens, P losses are largely controlled by rates of P dissolution from soil to soil solution. Dissolution rates depend on soil characteristics (i.e., P sorption strength) and surface area (McDowell & Sharpley 2003a). In New Zealand, sorption strength can be approximated by soil P retention. McDowell & Condon (2004) produced empirical equations to estimate DRP in sub-surface and overland flow:

$$1 \quad \text{DRP (overland flow; mg/L)} = 0.024 (\text{Olsen P [mg/kg]/P retention [\%]}) + 0.024$$

$$2 \quad \text{DRP (sub-surface flow; mg/L)} = 0.069 (\text{Olsen P [mg/kg]/P retention [\%]}) + 0.007$$

The equations use the quotient of Olsen P and P retention to account for different soil sorption capacities. The quotient also accounts for the exponential increase in P loss with Olsen P concentration in a specific soil. Some (e.g., Heckrath et al. 1995) have approximated this relationship to a split line model that defines an Olsen P concentration beyond which P loss increases at a greater rate than if below, but this is just a mathematical construct.

Equations 1 and 2 are applicable for cases of unlimited P supply. However, in very large or long flow events, soil P supplies can limit loss rates. This is caused by desorption of all available P from the surface of aggregates and the limited resupply of P via diffusion from the inside of aggregates or smaller pores in the soil matrix. The result is a dilution of P in solution. Such supply-limited P loss occurs more often during overland flow where soil-water contact periods are short, compared with sub-surface flow (Koopmans et al. 2002).

3.2 Fertiliser

Fertiliser is applied to soils to replenish available P, which is commonly measured as Olsen P. However, it takes time after each application for fertiliser-P to dissolve into the soil solution and then be sorbed onto the soil surface. During this time, some dissolved fertiliser-P may be lost from soil in overland or subsurface flow. These losses can be decreased through simple management practices like scheduling fertiliser applications outside of periods of frequent rain or on saturated soils. However, even when this is done correctly it is estimated that the contribution from fertilisers like superphosphate comprise about 10% of annual farm P losses from grazed paddocks (McDowell et al. 2007a).

Another strategy for minimising fertiliser P losses is to use relatively water-insoluble P fertilisers, such as reactive phosphate rock (RPR), which is < 1% water soluble. In contrast, superphosphate is > 90% water soluble. Results of a rainfall simulation study indicated that losses from RPR were 90% lower than losses from superphosphate (McDowell et al. 2003a). Catchment-scale and long-term comparisons of RPR and superphosphate have not been made, particularly to determine whether low-concentration, long-term P losses released from RPR are greater than high-concentration, short-term P losses from superphosphate. However, this scenario seems unlikely, given the differences in P concentrations in overland flow, observed following rain events within a week of applications of superphosphate (up to 80 mg/L), and RPR (< 0.1 mg/L) (McDowell et al., 2003a).

3.3 Dung and effluent

In a grazed pasture the quantity of dung and P returned to the soil varies with animal type and diet. For instance, sheep defecate about 19 times a day (range 7-26; Haynes

& Williams 1993) while cattle defecate about 12 times a day, but deposit larger pats. There are also differences between animal types and climate conditions in rates of dung decomposition and incorporation into soil. For instance, Rowarth et al. (1985) found that sheep dung on flat land had decomposed completely within 28 days in winter, but lasted for > 75 days in summer.

When climate, soil and overland flow conditions were held constant, potential rates of P loss were greatest for cattle dung, followed by sheep and deer dung (McDowell 2006). Further work has indicated that dung is the source of 20-30% of the P lost from a typical dairy farm paddock (McDowell et al., 2007a).

Phosphorus losses from effluent to water occur via pond discharge or sub-surface or overland flow of effluent applied to land. Pond discharges generally lose more P to waterways, compared to land applications which retain P on-land due to greater contact and sorption with the soil (Houlbrooke et al. 2004a). There are several factors that influence P-loss rates after effluent applications to land. Houlbrooke et al. (2004a) concluded that wet land increases P losses following dairy shed effluent applications. These losses can be exacerbated by artificial drainage or macropores and cracks that occur in some dry and fine-textured soils. To minimise P losses from land application, it is suggested that effluent applications are limited to periods when there is a soil water deficit (Houlbrooke et al. 2004a, b). Clearly, this requires enough effluent storage capacity to cease applications during periods of high soil moisture. Low-rate effluent applicators can significantly decrease P losses compared to travelling irrigators by reducing effluent ponding and subsequent overland or subsurface flow, or by increasing contact time between the effluent and soil, which maximises P sorption (Houlbrooke et al. 2004b).

3.4 Grazing and treading

Grazing animals affect P losses by smearing and compacting soils and by exposing plant cell vacuoles and cytoplasm (Drewry 2006; McDowell et al. 2007a). The latter action has only been recently considered as a source of P lost from grazed paddocks. The P within plant cells tends to be highly available to aquatic organisms, as it consists largely of orthophosphate and polyphosphate. McDowell et al. (2007) estimated that this source could account for 10-20% of P losses from a paddock grazed by dairy cattle.

Treading can cause soil compaction, which affects flow pathways. In extreme cases, treading decreases soil infiltration rates and causes infiltration-excess (or Hortonian) overland flow. A more common effect of treading is to compact soil, which decreases the porosity and the capacity for water storage. Compacted soils become saturated quickly and are subject to saturation-excess overland flow (McDowell et al. 2003b). The susceptibility of soil to treading damage depends on animal and soil types and soil moisture conditions (Climo & Richardson 1984). Heavy animals such as cattle generally cause more treading damage, but smaller animals can cause severe local effects (e.g., soil trampling by deer along fence lines). Poorly-structured soils, including soils with low organic matter content, tend to compact more easily than well-structured soils (Greenwood & McKenzie 2001).

Agricultural land in the southern South Island of New Zealand is particularly susceptible to P-losses due to animal treading. Animals in this area are often wintered on forage crops. The Pallic soils commonly found in this region have high structural vulnerability, and high rates of P loss occur by overland flow when these soils are wet and compacted. Winter P losses from southern New Zealand farms can equal or exceed P losses for the rest of the year (McDowell & Houlbrooke 2008).

3.5 Other P sources

In addition to pastures, other farm areas can be important sources of P. These include lanes, races, holding pens, and other areas where animals spend time and deposit excreta. For instance, McDowell (2007a) found that the majority of P losses from a small sub-catchment of Lake Rerewhakaaitu occurred during surface runoff events from a lane that was used each day by cows going to and from the milking shed. Troughs and gateways can also produce disproportionately large losses of P compared to paddocks, due to dung deposition, treading, and decreased infiltration (Hively et al. 2005; Lucci et al., 2008). The relative impact of different P source areas depends on hydrological connectivity between source areas and surface waters. Phosphorus losses from non-pasture areas can be minimized by judicious placement within the hillslope away from waterways and sensible engineering of heavily trafficked lanes and races.

Phosphorus losses from non-pastoral agriculture areas (e.g., crop and horticultural fields) have not been studied as intensively as P losses from pastoral systems due to their smaller area and, in some cases, drier locations. For instance, most of the arable cropland in New Zealand is in areas with little potential for P losses (i.e., flat land with

relatively low rainfall). One exception is forage crops grown for grazing animals (see above). Compared to arable crops and pastoral land uses, P inputs to vegetable crops tend to be high and these crops are often grown in areas with high susceptibility to P loss (e.g., humid sub-tropical regions). In these situations, P losses may be linked to high fertiliser application rates. Cropped soils tend to have good infiltration rates, and P losses occur primarily along sub-surface flow paths. High fertiliser use on vegetable crops also increases the risk of P loss via overland flow in periodically saturated areas. Best management practices have been developed recently to minimize losses of P and other nutrients from vegetable cropping land (Hartz 2006).

3.6 Relative losses

In a recent review of pastoral systems, McDowell and Wilcock (2008) compared contaminant losses to water from different farming systems. For P losses, there were no significant differences between dairy, deer and mixed (largely sheep and beef) farms, but these had greater P loss rate than sheep farms or native forests (Fig. 2). Most of the P sources common to the grazed lands have been covered above. However, there are also some system-specific issues that affect P losses. For instance, cattle and deer stand or wallow in waterways for drinking and thermoregulation, but sheep remain on waterway margins while drinking. Excluding cattle and deer from waterways with fences can lead to substantial decreases in P losses. Fencing resulted in a 30% decrease in P losses in a catchment in the northeast United States (James et al. 2007), and a 90% decrease in two deer-farming catchments in Otago (McDowell 2008a). In addition to wallowing, deer cause high P losses via soil erosion near fence lines. Deer pace along fence lines when they are stressed. Pacing exacerbates soil compaction and erosion (Pollard & Stevens 2002). The relative impact of fence-line pacing on P losses depends on the time deer spend in paddocks and the erodibility of paddock soils. For instance, Thorrold & Trollove (1996) found that up to 20 tonnes soil/ha were eroded from a Pumice soil in one year, while the annual loss of Pallic soil from an Otago deer farm was 1 tonne/ha (McDowell & Paton 2004). These two erosion rates correspond to P loss rates of about 14 and 1 kg P/ha/yr (assuming topsoils with P concentrations of ~ 1000 mg/kg, and bulk densities of 0.7 and 1.0 g cm⁻³ for Pumice and Pallic soils, respectively).

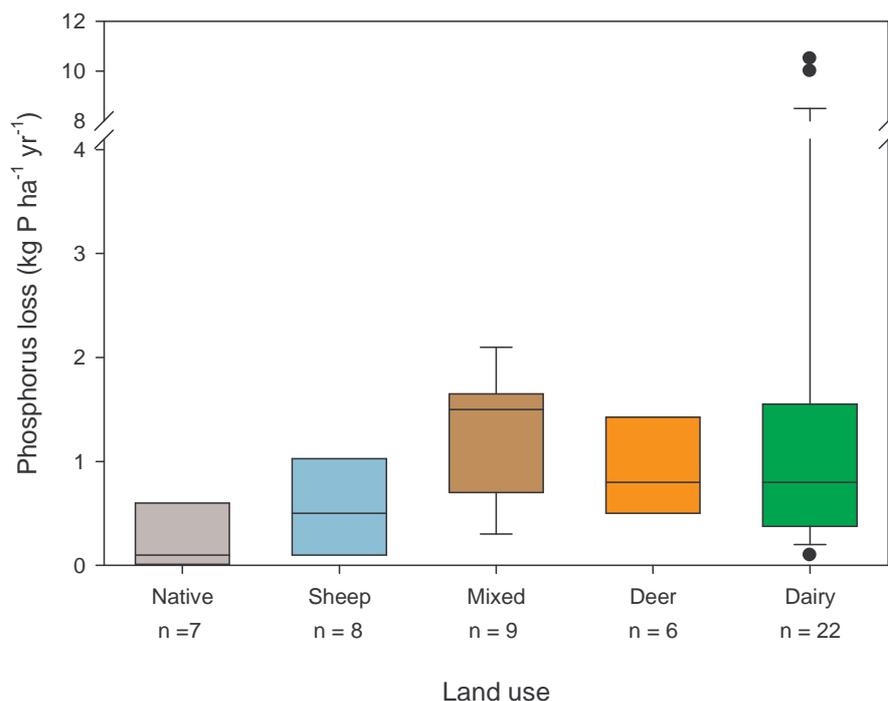


Fig. 2 Box plots of catchment scale phosphorus losses from different land uses. The box plots give the median, 10th, 25th, 75th, and 90th percentiles and outliers (adapted from McDowell & Wilcock 2008).

4. Transport processes

The presence of P sources on a farm does not mean that P will be lost from the farm and reach surface water. In order for this to occur, P needs to be transported. Mechanisms of transport depend on the availability of hydrological pathways, which are in turn controlled by climate, catchment characteristics and land management. In some cases, P is lost via wind erosion, but hydrological pathways generally dominate (Parfitt et al., 2008). To give some indication of how hydrological processes regulate P losses, Table 1 lists P loss rates for different agricultural systems and the predominant pathways.

Table 1 Phosphorus losses from different farming systems around New Zealand in relation to rainfall and the loss pathway.

Site	Loss (kg P/ha/y)	Rainfall (mm)	Loss pathway	Reference
<i>Dairy</i>				
Inchbonnie, West Coast	6.0	4500	Overland flow from hump and hollow land	McDowell (2008b)
Inchbonnie, West Coast	1.6	4500	Sub-surface flow (leachate) from hump and hollow land	McDowell (2008b)
Massey, Manawatu	0.7 - 2.6	1200	Sub-surface flow (mole-pipe drainage) from effluent block	Houlbrooke et al. (2008)
Windsor, Otago	1.2	670	Overland flow from winter forage crop plots	McDowell & Houlbrooke (2008)
Lincoln, Canterbury	0.3	700	Sub-surface flow (leachate)	Toor et al. (2004)
Rerewhaakaitu, Bay of Plenty	1.9	1500	Overland flow from lane	McDowell (2007a)
<i>Deer</i>				
Lumsden, Southland	1.6	800	Overland flow from winter forage crop	McDowell & Stevens (2008)
Telford, Otago	3.9	850	Stream flow from catchment with wallow at outlet	McDowell (2007b)
<i>Sheep and beef</i>				
Ballantrae, Manawatu	1.5	1050	Stream flow from catchment with direct stock access to stream	Lambert et al. (1985)
<i>Sheep</i>				
Winchmore, Canterbury	0.7 - 12.6	650	Overland flow as outwash from border dyke irrigation	McDowell & Rowley (2008)

4.1 Overland flow

Overseas literature indicates that most soil P is lost via overland flow (e.g., Sharpley 1998). In areas with continental climates, most P is lost in summer in response to short high-intensity rainfall events, while snow prevents water movement in winter. In New Zealand's maritime climate, most P losses occur in winter (Gillingham & Thorrold 2000). However, there is still potential for appreciable P loss in summer in warm, humid areas of New Zealand, primarily in the North Island.

Overland flow can occur in response to infiltration-excess or saturation-excess conditions. Infiltration-excess overland flow (or Hortonian overland flow) occurs when rainfall intensity exceeds the infiltration capacity of the soil. In contrast, when rain falls on saturated soil, water cannot infiltrate and overland flow occurs via saturation-excess conditions. Areas within a catchment that are susceptible to saturation-excess conditions are usually depressions, including stream channels. This means that saturation-excess overland flow can occur on stream banks and hillslopes, and can expand and contract in response to changes in drainage and rainfall rates. Contributing areas that expand and contract in response to soil moisture are termed variable source areas (Ward 1984). In contrast, infiltration-excess overland flow is dominant in areas like lanes and tracks, and in areas that are subject to excessive animal treading such as troughs, gateways and wet paddocks with compaction prone soils (e.g., winter forage crops; McDowell et al. 2003b).

Catchment characteristics can alter how, when and where overland flow occurs. Abrupt breaks in the landscape such as changes in soil or bedrock composition can promote or diminish overland flow by altering soil moisture. It is rare for overland flow to occur as sheet flow except during very intense rainfall. Due to flow convergence, overland flow often becomes channelized, which increases its kinetic energy and erosive force. Consequently, P transported in channelized flow is unlikely to be effectively attenuated by buffer strips (Verstraeten et al. 2006).

Contrary to popular belief, P in overland flow is not necessarily dominated by particulate P. The composition of P in overland flow varies with flow mechanism and land use. For example, saturation-excess overland flow generally has relatively more dissolved P than infiltration-excess overland flow (Kleinman et al. 2006). Overland flow on flat land (e.g., border dyke outwash) is likely to contain more dissolved P than on steep land (Nash & Murdoch 1997). For a given flow mechanism and slope, cattle and

deer tend to generate high particulate P losses and while sheep tend to generate high dissolved P losses (Lambert et al. 1984; McDowell & Paton 2004).

4.2 Sub-surface flow

The general consensus is that P losses via sub-surface routes are lower than in overland flow because the longer residence time in sub-surface flow leads to increased P-sorption to soil (Sharpley 1998). There are exceptions to this pattern, however. In soils with low P retention (e.g., podzol, organic and sand soils) there is little sorption during sub-surface flow. In addition, high sub-surface P loss can occur when water flow-soil interactions are circumvented by artificial drainage. Monaghan et al. (2000) found that subsurface P losses from an imperfectly drained Pallic soil with artificial drainage were equivalent to, or greater than, P losses via overland flow. Sub-surface P transport is also enhanced if there are macropores that provide preferential flow paths with minimal soil interactions (Thomas et al. 1997).

4.3 In-stream processes

The biomass of periphyton and macrophytes in streams and lakes is related in part to the bioavailability and concentration of P. In turn, bioavailability depends on physical characteristics of the stream or lake (e.g., water residence time, dissolved oxygen levels) and on the chemical forms of P that are present. Orthophosphate, which dominates the dissolved inorganic P pool, is 100% bioavailable. Some dissolved organic forms of P (e.g., amino acids) are made bioavailable by extracellular enzymes (e.g., orthophosphate monoesterase, orthophosphate diesterase and phytase) released from plants and microbes (Quiquampoix & Mousain 2005). In fast-flowing streams, processes that convert particulate P to dissolved P (e.g., desorption, excretion) may increase dissolved P concentrations downstream, but measurements of total P (TP) do not accurately indicate bioavailability. Therefore, periphyton or macrophyte biomass in streams is best predicted from dissolved P concentration (Biggs 2000b). In slow-flowing streams and lakes, longer residence times mean that both particulate, via exocellular phosphatase enzymes, and dissolved P are potentially available to local periphyton and macrophytes, and TP may be a good predictor of biomass (Chapra 1997).

Total P transport in streams usually reaches annual maximum rates during floods, and a large proportion of the P transported during floods is in particulate form. The source

of particulate P in transport may be resuspension of stream bed sediments, erosion of stream banks, subsoil or topsoil transported in overland and sub-surface flow, or by wind. The relative proportion of each source depends on catchment factors like slope, elevation, and soil type, and interactions between these factors and climate (McDowell & Wilcock 2007). In most intensively farmed catchments, much of the particulate P transported by streams is from topsoil, but in cases of recent gully formation, including channel rejuvenation, subsoil is a major source (Olley et al. 1993). Once in a stream, particulate P settles and is resuspended repeatedly, at frequencies that are related in part to particle size and density. (McDowell & Sharpley 2002).

Periphyton growth in many streams can be limited by flow, substrate, light availability or low temperature, regardless of the availability of P. During low flows, sediment deposited during storms can influence dissolved P concentrations in the water column via adsorption to and desorption from particles (Dorioz et al. 1998; McDowell et al., 2002; Parfitt et al., 2007). As with settlement, desorption is partly determined by particle size. Although more P is held by fine particles than by coarse particles, desorption from fine particles is slower than from coarse particles (Stone and Murdoch 1989). In slow-flowing streams and lakes where stratification and anaerobic conditions develop, fine particles can release P into the water column at high rates (Boström et al. 1982).

The relative proportion of fine to coarse-sized particles and P enrichment also varies with geology. For instance, although most soils used in New Zealand for agricultural production are silt loams, those derived from volcanic materials tend to contain more P than those derived from greywacke (Parfitt et al. 2008).

5. Impacts

The principal ecological risks associated with P enrichment of streams and lakes are proliferations of algae or macrophytes. Such proliferations are a common symptom of eutrophication, which refers to nutrient enrichment at levels greater than desirable or natural. However, it should be noted that it is important to manage both N and P inputs as nutrient limitation varies greatly spatially (e.g., from one tributary to another) or temporally (e.g., season), and in some areas N-limitation may be more prevalent than P-limitation (e.g., Taupo and the upper Waikato river). While the availability of P does not always regulate or limit the growth of algae or macrophytes, experimental assays in New Zealand streams and lakes indicates that P-limitation is frequent and

widespread (Francoeur et al. 1999, Downs et al. 2008). When P supplies to P-limited water bodies are elevated, algal and macrophyte growth rates may therefore increase. If increases in growth rates are relatively small, herbivores (primarily fish and invertebrates) may consume the additional production. If increases in growth rates are large or herbivores are relatively ineffective, algal or macrophyte biomass will accumulate. Proliferations of algae or macrophytes have negative effects on many ecological properties and processes, including decreased biodiversity through habitat alteration, changes in algal or macrophyte composition, the decrease of dissolved oxygen due to decomposition, increased production of algal toxins (e.g., microcystins, anatoxins), elevated pH, and decreased clarity (Smith et al. 1999). In addition to ecological effects, algal and macrophyte proliferations can have negative economic and aesthetic effects, including water taste and odour problems, water intake clogging, impairment of angling, decreased landscape and property values, toxicity to humans and livestock, and fish kills that impact commercial and recreational fisheries.

Despite the severe effects of eutrophication, and the highly degraded states of some P-enriched streams and lakes, a decrease of P loading has resulted in recovery of ecological health at many sites (e.g., Barbiero et al. 2002, Jeppessen et al. 2005). Some cases of successful recovery have occurred at sites where P loading was due to point source inputs (e.g., sewage outfalls), and was relatively easy to remove or mitigate (e.g., Beklioglu et al. 1999). In contrast, many streams and lakes in agricultural landscapes are primarily affected by non-point source P inputs. This appears to be the case for many areas of New Zealand (Parfitt et al. 2008). The effort and expense required to reduce or eliminate non-point source P inputs is generally greater than for point sources, and may require large-scale nutrient management and/or intensive in-situ treatment such as alum addition (Welch and Cooke 1999).

6. Proceedings of a workshop on the role of phosphorus in surface water quality

The workshop was held at the end of July 2008 in response to several questions regarding the role of P in surface water quality. Before the workshop, people were asked to list questions for the attendants, namely: scientists, regional council scientists and some industry representatives. The following represents an edited version of the responses. It was decided to order the questions consistent with the preceding mini-review. Hence, they are categorised into those pertinent to sources, transport mechanisms and impacts. However, while there may be some overlap we

endeavoured to minimise it. For each question a concise answer is listed and a more in depth commentary given if appropriate.

6.1 Sources

Q. Do we know which water bodies are P-limited and what are the P concentrations responsible?

A. Data for this is given in the accompanying paper (McDowell & Larned, 2008), which lists the general state of P limitation. An extension of this analysis is given below for each region (Fig. 3). It should be noted that in some areas like Canterbury the bias towards P-limitation is clear. However, in others – generally in the North Island there is more of a mixture of N-, Co- and P-limitation. Several reasons for this exist such as P-rich geology in Lake Taupo and the upper Manawatu River.

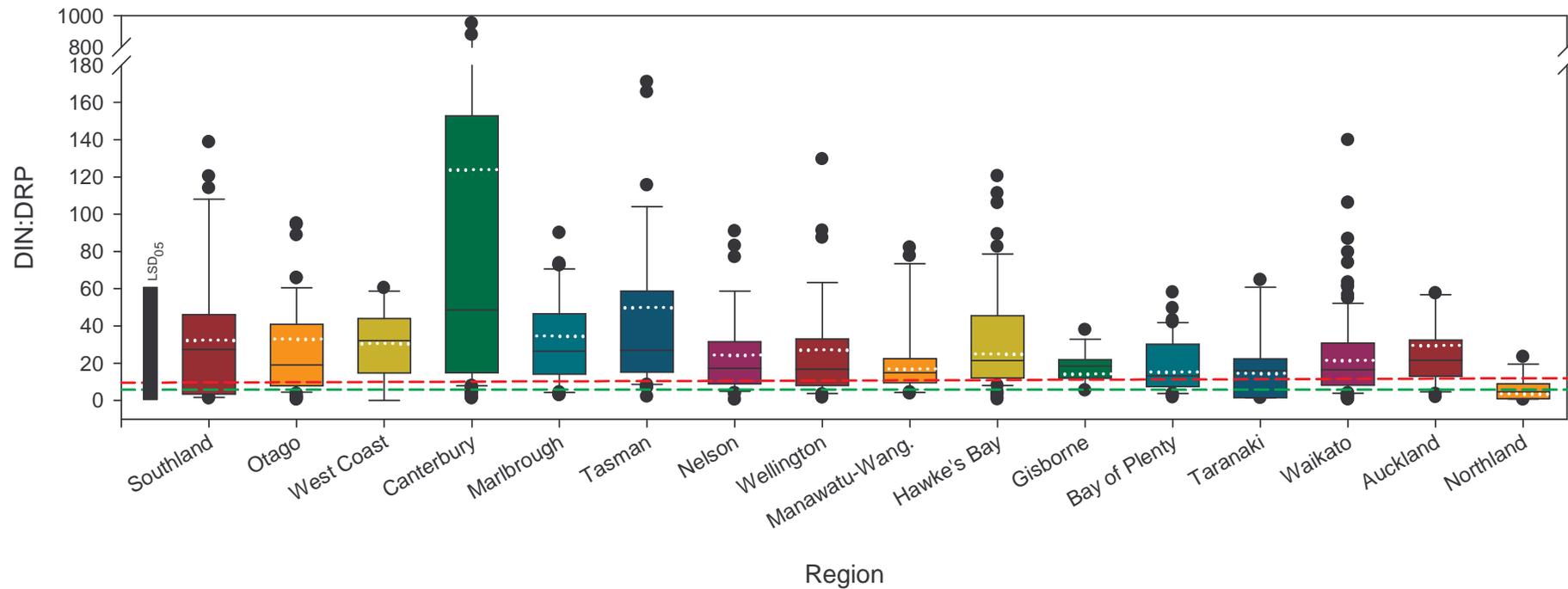


Fig. 3 Box plots showing the median (solid line), mean (dotted line), 25th and 75th percentiles (box edges), the 10th and 90th percentiles (whiskers) and outliers for the ratios of the concentration of dissolved inorganic N to P in each region. Ratios above the top dashed line indicate potential P-limited periphyton growth, ratios below the lower dashed line indicate potential N-limited periphyton growth, and ratios between the dashed lines indicate potential co-limitation.

Q. What proportion of P loss come from diffuse agricultural sources and point sources? Furthermore, where are the hotspots in diffuse agricultural systems?

A. It was felt by the regional council scientists that they had a good handle on the point sources within their region. However, the national picture is unclear, although an audit of inputs and outputs for the 2001/02 growing season lead Parfitt et al. (2008) to conclude that the majority of P was associated with agricultural activities.

Commentary: In general, point sources are associated with urban development. In New Zealand most urban development is located near the coast and is relatively sparse compared to other well developed nations. This infers that the majority of P inputs to our surface waters will be associated with diffuse agricultural inputs. An easy method of confirming the potential for point sources in a catchment is to plot load against flow. By extrapolating a linear regression line back to no flow, an indication is gained of the P coming from flow independent point sources.

The delineation of hot spots of P loss within an agricultural system has seen recent attention. Work in the US describes hot spots as critical source areas: in other words areas in the catchment where there is an enriched and readily available source of P with an easily accessed transport mechanism to get it to a stream (Gburek et al. 2000; Sharpley et al. 2001). Their approach has been to classify fields (or paddocks) according to P inputs and P concentration and its potential for overland flow and connectivity to a stream. Unfortunately, this approach has limited spatial utility. In New Zealand, recent work has categorised P losses to come from infiltration-excess and saturation-excess areas (MS Srinivasan *pers. comm.*). Generally, if a connection to a stream exists then infiltration-excess areas can represent the majority of P loss as they are usually areas like lanes and troughs where animals and dung deposition concentrate. However, if there is little opportunity for areas like these to reach the stream, or if soil hydrology or chemical condition (e.g., a sandy soil with little P retention) are very conducive to P loss, then saturation-excess areas can account for much P loss. Work to define rules when and where each process is dominant in a catchment is in its infancy (e.g., Srinivasan & McDowell 2007).

Q. What are the short- and long-term P losses from single superphosphate and reactive phosphate rock (RPR) and what is their respective environmental impact?

A. The jury is out on long-term P losses from RPR compared to superphosphate, but short-term losses are much less for RPR.

Commentary: Although data from the Winchmore long-term fertiliser trial indicates that water soluble P in the RPR plots is greater than the plots with an equivalent rate of P applied as superphosphate (McDowell et al. 2003a), catchment data at Waipawa (Table 2) indicates that short-term losses soon after application tend to dominate losses during the rest of the year (from fertiliser only). However, this trial applied fertiliser (RPR or superphosphate) in July when a proportion of fertiliser will inevitably be applied to ephemeral stream channels during topdressing (Blennerhassett et al. 2007). The same losses or indeed a difference may not occur if applied in summer when the channel is dry.

Table 2 Annual rainfall and runoff (mm) and loads (g/ha) of dissolved reactive P and total P for both catchments at Waipawa, Hawke’s Bay (data from Blennerhassett et al. 2007 and McDowell unpublished).

Year	Super	Rainfall	Runoff		DRP		TP	
			North	South	North	South	North	South
2004	South	727	164	166	33	49	276	370
2005	North	732	175	211	206	72	533	427
2006	South	958	255	314	115	152	316	654
2007	Both	745	122	152	155	106	226	199

Q. Are there any products that can decrease P losses?

A. Yes.

Commentary: McDowell & Catto (2005) and Entry & Sojka (2008) showed that the risk of short-term losses from fertiliser application was related to water solubility. Products like RPR and serpentine superphosphate are two examples of fertiliser products with lower water solubility than superphosphate. However, in addition to these products, recent work has looked at increasing the soil sorption capacity. Although in its infancy this research has shown that increasing sorption capacity can decrease the losses of dissolved P in overland and sub-surface flow (e.g., McDowell 2003; Redding et al. 2008). However, the wide spread use of any material will depend on the economics of mitigation potential. In other words other strategies focusing on other parts of the farm may give better mitigation potential per dollar spent.

Q. What are the influences of in-stream P-buffering relative to geology?

A. During the workshop areas such as the Manawatu were highlighted vis-à-vis to geology and the potential for enhanced P losses. Here the catchment contains P-rich rocks that contain apatite and calcareous minerals with P inclusions. This makes the probability of decreasing P concentrations below that buffered by the P-rich rocks improbable (Parfitt et al., 2007). Similarly, Lake Taupo is enriched, relative to N, with P due to the weathering of P from the hydroxyapatite containing mineral, ignimbrite (Timperley, 1983).

Commentary: Geology can influence P losses and cycling within the stream, but must be viewed in conjunction with other sources. For example, if a stream receives much effluent either from pond discharge or overland flow then this can mask any geological effect. However, when we refer to geological influence in slow flowing streams or rivers where deposition is possible we are really referring to parent material for soil type. In these cases it is the erosion of a specific soil, either top or sub-soil, that will define P in-stream cycling. However, in faster flowing streams or rivers bedrock will have the major influence on the uptake and release (spiralling) of P. Munn & Meyer (1990) found that a stream with granite bedrock had an uptake length of 85 m, while a stream with volcanic bedrock had an uptake length of 697 m. The P-enriched volcanic sediments had a five times greater ambient DRP concentration leading the authors to conclude that uptake and retention were higher in the granitic stream due to “strong biotic control of P uptake coupled with high P demand, result in relatively short P uptake lengths”.

Q. What are the best practical methods to measure and quantify P losses from bed sediments during low flows?

A. The potential for P to influence P in stream flow depends on flow rate. At high flow, P in the water column may be supplemented by resuspended sediment and dissolved P from the bed. However, much of this particulate-P will be unavailable and deposit downstream, but may become available there. At lower flow rates, without appreciable bed resuspension, dissolved P is controlled by the sorption and desorption characteristics of the bed sediment and dissolved oxygen (Reddy et al. 1999). A few studies have shown that dissolved P concentrations at baseflow is related to the equilibrium P concentration of the sediment at zero net sorption and desorption (EPC_0), and that EPC_0 is related to particle size of the sediment (tends to increase

with particle size; Haggard et al. 2007; McDowell et al. 2001). The method of determining EPC_0 is to shake known amounts of sediment with solutions of different P concentrations. Plotting the P in solution (x-axis) against P sorbed (Q , mg kg^{-1}), not only allows for the fitting of a P sorption isotherm to determine P sorption strength (k) and capacity (Q_{max}) but also allows EPC_0 to be determined as the point where the fitted line crosses the x-axis at zero sorption /desorption (Fig. 4).

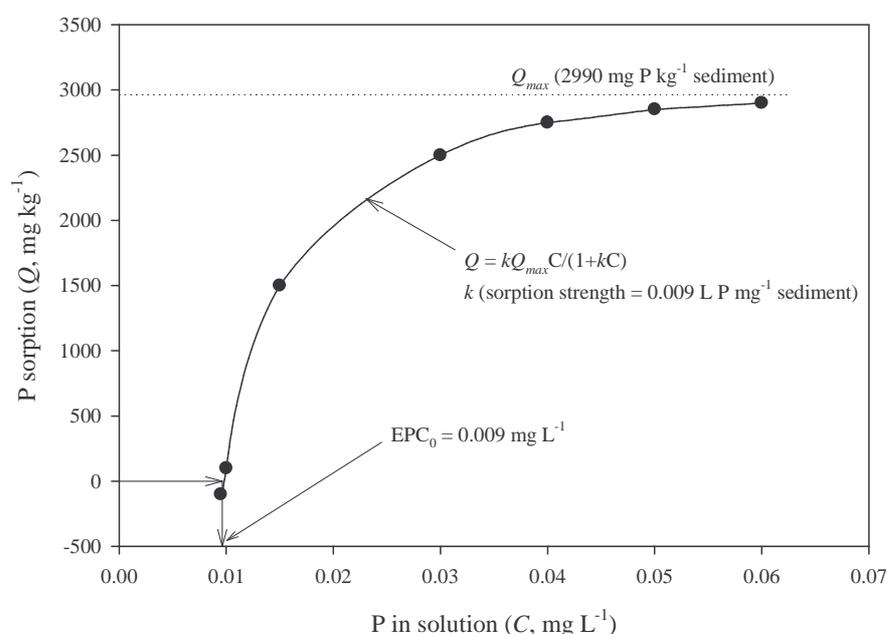


Fig. 4 Plot of a P sorption against P in solution after equilibration with a range of different P concentration solutions indicating how to determine sorption capacity and strength and the equilibrium P concentration at zero net sorption and desorption for a theoretical sediment.

6.2 Transport mechanisms

Q. How much P is lost via surface or sub-surface routes?

A. This depends on the degree of artificial drainage, natural accelerated drainage (i.e. macropores), the soil type, and the split between deep drainage and streamflow via the hyporheic zone. In most catchments, most P is lost via overland flow despite this often comprising the minority of streamflow. However, in artificially drained soils the majority of P can be lost in sub-surface flow. Although the vast majority of water will travel via the drainage network, the quantity of P carried by drainage water depends on the degree of macropores and the use of the land. For instance, if the soil is well

developed and has few macropores then there is a lot of opportunity for uptake as water drains through the soil (Thomas et al. 1997), whereas if the soil has a large proportion of macropores, then there is opportunity for the pore surfaces to become saturated with P enhancing P loss. Soil P saturation is only accelerated if the paddock receives regular P applications, especially if in a liquid or readily transportable form like effluent (McDowell et al. 2005).

Q. What is the likely contribution from sub-surface flow from soils in relation to Olsen P and P retention?

A. As P travels through soil, the degree of uptake is proportional to the P sorption capacity of the soil and the current soil P concentration or degree of saturation (McDowell & Sharpley 2003b). Phosphorus losses increase if the soil has a low capacity to sorb P or as the soil becomes P saturated. An approximation of soil P sorption capacity and degree of soil P saturation is gained by the quotient of Olsen P and P retention, as mentioned in the soil section of the mini-review, and can be used to estimate P concentrations in sub-surface flow. However, this only accounts for dissolved reactive P. Dissolved organic P (DOP) losses can also be bioavailable but the proportion of DRP:DOP tends to be less with increasing Olsen P (Heath, 2005; Whitton et al., 2005). The exception can be in effluent applied soils where DOP losses are facilitated by organic materials in the effluent negating sorption to the soil. Effluent movement through macropores can also result in a lot of particulate P being lost as colloids (McDowell et al. 2005).

Q. What happens to P once it gets into a waterway? What are the primary storage compartments? What influences the movement between compartments?

A. Once in a waterway, P can either be maintained in the water column and travel downstream to a lake or estuary, or cycle through biotic or abiotic systems. We have mentioned how P is cycled through abiotic sediments above, the only addition to the processes mentioned are that in P-enriched streams, P may be stored as precipitate, although this is usually associated only with point source discharges (House 1990). Cycling through biotic systems depends on the ecosystem composition (i.e. the balance between periphyton/phytoplankton and macrophytes, land use, flow conditions and sediments). What is not commonly considered is that about 30% of P in sediments is housed within biomass (microbial and to a lesser extent bio-films; Khoshmanesh et al. 1999). This portion can be altered with sudden desiccation and

rewetting - leading to death and a flush of P or increased retention if repeated wetting and drying cycles occur.

Q. Where does periphyton get its P from at baseflow?

A. The immediate source of P for periphyton is the layer of water surrounding living cells. This is true regardless of the flow level. Molecular transporters at cell surfaces move dissolved compounds containing P into cells, where the compounds are catabolised and synthesised into biomolecules such as proteins and nucleic acids. Unlike vascular macrophytes, periphytic algae have no roots, so streambed sediments are not an immediate nutrient source. Ultimately, streambed sediments, suspended sediments, and the water flowing over and through periphyton are P sources. As noted above, dissolved organic and inorganic P must be released from sediment particles before being assimilated by periphyton. During baseflow periods, suspended sediment concentrations may be relatively low, and stream bed sediments more important than during floods. If baseflow is narrowly defined as groundwater discharge to stream channels, with no runoff contribution, then diffusion from bed sediments is likely to be a dominant nutrient source during baseflow periods.

The relative importance of different P sources for periphyton, and the rates at which dissolved P is released from stream bed or suspended sediments are difficult to determine, for two reasons. First, a practical tracer for dissolved P is not available. The best P tracer is radioactive ^{33}P , but releasing ^{33}P in the environment is rarely permitted. Second, P release rates from sediments are difficult to measure accurately because areas of sediment and volumes of water must be isolated while maintaining realistic hydrodynamic conditions. As an alternative to empirical measurements, the relative importance of different P sources have been estimated by comparing loading rates of major sources (e.g., diffuse groundwater and runoff, point sources, soil erosion), and assuming identical relationships between loading and periphyton uptake for each source (Edwards & Whithers 2007). This mass-balance approach is unlikely to be accurate, because stream bed sediments may be both a source and a sink for dissolved P.

Q. How does particle size/settling in stream pools + P sorptive characteristics support periphyton growth?

A. As noted above, particle size affects both settling and resuspension rates, and rates of dissolved P absorption and desorption. Presumably, P-limited periphyton growth is maximised when rates of dissolved P regeneration (desorption + diffusion)

are maximal. For a given concentration and specific density of suspended particles, larger particles presumably settle faster and desorb P faster than small particles. However, large particles probably form sediment deposits with higher porosity and higher dissolved oxygen concentrations than small particles. Phosphorus desorption and diffusion from sediments is usually inversely proportional to dissolved oxygen concentrations. These observations suggest that particle sizes and settling rates alone are unlikely to control either P regeneration or periphyton growth. Instead, P regeneration is probably strongly influenced by particulate P concentrations, water column dissolved P concentrations, and sediment oxygen concentrations.

6.3 Impact

Q. What load or concentration of P loss from farming systems starts adversely affecting P-limited water bodies?

A. In theory, algae and macrophytes that are limited by P (or another nutrient), respond to increases in P availability rapidly when ambient concentrations are low, and more slowly when ambient concentrations are high. As P concentrations increase, the response per increment increase decreases until further increases in P elicit no response. At this “saturating” P concentration, another factor becomes limiting. Graphs of algal and macrophyte growth or productivity versus P concentration often take the form of rectangular hyperbola (rapid initial growth, eventually reaching an asymptote). Two important messages about eutrophication are conveyed by these productivity-concentration curves. First, very small increases in P losses from land (e.g., $\mu\text{g L}^{-1}$) can lead to relatively large growth responses by algae and macrophytes, particularly when ambient P concentrations are low. In other words, a small increase in P loading to an oligotrophic water body can have a proportionally larger effect than the same increase in loading to a eutrophic water body. Second, when ambient P concentrations are high (e.g., mg L^{-1}), modest decreases in loading may not result in detectable reductions in growth or biomass, because growth is in the asymptotic portion of the curve. In other words, it is much harder (and more costly) to decrease algal proliferations in eutrophic water bodies than in oligotrophic water bodies. These issues are clearly illustrated by the responses of oligotrophic lakes to changes in P and N loading (Wilcock et al., 2007).

Example 1. Lake Tahoe, USA (Goldman 1988, Jassby et al. 1995).

Lake Tahoe was ultra-oligotrophic until the late 1960s, when catchment development, erosion, sewage discharge, and atmospheric deposition of nutrients from outside the

catchment initiated the process of eutrophication. Before ~ 1990, lake phytoplankton productivity was N-limited. The total N stock in the lake increased by 30% between 1973 and 1987. During the same period, phytoplankton productivity increased by 200%. This pattern corresponded to the rapidly increasing phase of the saturation curve described above. Eventually, the phytoplankton shifted to P limitation, and further increases in N loading had relatively small effects on phytoplankton productivity. This stage corresponded to the asymptotic portion of the saturation curve.

Example 2. Saldenbach Reservoir, Germany (Horn 2003).

The replacement of high P-detergents with low and no P-detergents in Germany, and improvements in catchment management led to a 60-70 % decrease in P-loading to Saldenbach Reservoir, and a 7-fold decrease in DRP concentrations (from 14 to 2.2 µg/L) between the 1980s and the 1990s. However, phytoplankton concentrations doubled during the same period, due in part to a shift in the phytoplankton community that favoured effective competitors for the increasingly scarce P. This example indicates that a relatively large decrease in P loading may not result in a corresponding decrease in algae; an even greater (and probably very costly) decrease in P loading may be required to reduce algal growth.

Q. What mix of land use practices need targeted if we're to manage both P and nitrogen?

A. This varies according to location (and climate) and land use. Furthermore, in addition to considering which BMP would decrease N and P losses most, further consideration should be given to which BMP mitigates N and P most for the least money invested. Such an assessment was undertaken by Monaghan et al. (2008) who listed the effect of BMPs on a price per unit contaminant mitigated. As an example, Table 3 lists BMPs for mitigating N and P losses alone and in combination for a dairy operation.

One conclusion obvious from this analysis is that items covered in the Fonterra Clean Streams Accord are some of the most cost-effective possible. These should be considered before other less effective BMPs are advocated.

Table 3 Best management practices list in order of decreasing effectiveness on an environmental (kg ha⁻¹) or environmental-economic (\$ kg of nutrient conserved⁻¹) basis

for N, P and N and P for pastoral farms in Southland (Monaghan and Houlbrooke, AgResearch-Invermay, *pers. comm.*).

Environmental	Environment and Economic
<i>N</i>	<i>N</i>
Low cost stand-off pad for use during winter and other wet periods	Decreasing volume of effluent at milking shed
Restricted autumn and winter grazing	Deferred effluent application
Nitrification inhibitors	Low cost stand-off pad for use during winter and other wet periods
Low N feed	Stock exclusion from waterways
<i>P</i>	<i>P</i>
Deferred effluent irrigation	Olsen P concentration near the low end of optimal
Low cost stand-off pad for use during winter and other wet periods	Decreasing volume of effluent at milking shed
Stock exclusion from waterways	Stock exclusion from waterways
Timing and placement of P fertiliser application	Deferred effluent application
<i>N and P</i>	<i>N and P</i>
Deferred effluent irrigation	Decreasing volume of effluent at milking shed
Low cost stand-off pad for use during winter and other wet periods	Deferred effluent application
Stock exclusion from waterways	Stock exclusion from waterways
Timing and placement of P fertiliser application	Constructed wetland ¹

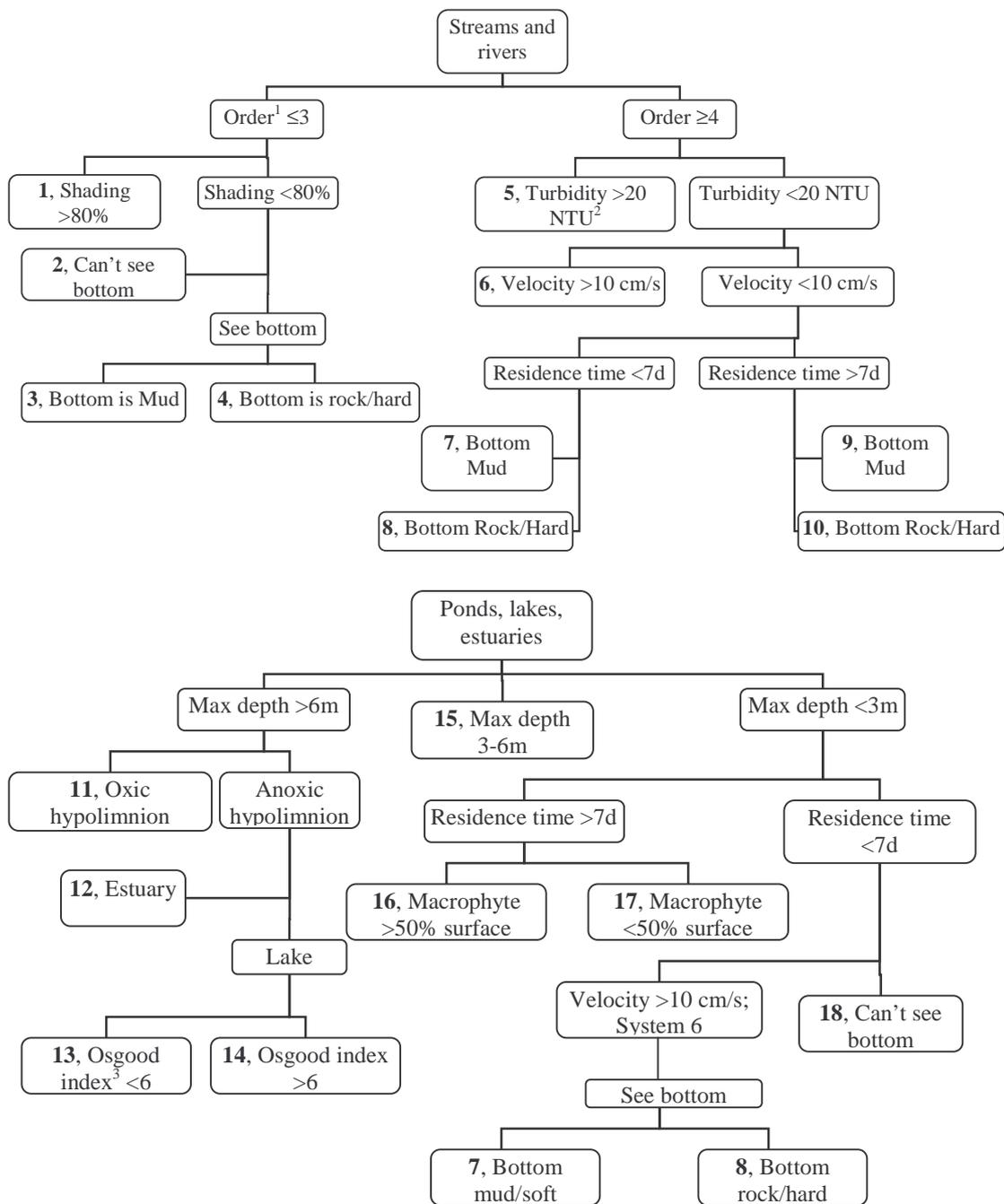
Q. Are best management practices going to have an impact on ecologically sensitive water bodies?

A. This depends on the system in question and the BMPs used. As an indicator, Table 4 and Figure 5 show the relative effect of decreasing P for different water bodies.

Table 4 Water body description, functioning, sensitivity to rising (↑) and falling (↓) P loads and possible management. Adapted from Newton & Jarrell (1999).

Description	Present functioning	↑	↓	Possible management scenarios
1, Heterotrophic small stream	Typically light limited. May be sensitive to increase in SS.	Low	Low	Maintain riparian vegetation, shading and access to flood plain to prevent siltation of riparian areas.
2, Phytoplankton-dominant small stream	Not light limited, but P limited	Medium	Medium	Decrease P inputs and monitor TP concentrations.
3, Macrophyte-dominant small stream	Controlled by vascular plants and macrophytes, which can get sufficient P from sediment.	Low	Low	Decreasing SS load most important.
4, Periphyton-dominant small stream	Responds quickly to changes in P load as little sediment is present.	High	High	As periphyton obtain P from water column, decrease essential. Groundwater P may be important.
5, Heterotrophic large stream	Characterised by high SS, hence, light limited. Little potential for eutrophication.	Low	Low	Possibly decreasing SS load, but trophic state may shift to eutrophic once light limitation is removed.
6, Periphyton-dominant large stream	High flow prevents macrophytes from establishing	Medium	Medium	Decreasing P concentrations in the water column. Groundwater P input may be important.
7, Macrophyte-dominant large stream or lake	Residence time insufficient for phytoplankton growth. Mud acts as good substrate for macrophytes	Low	Low	Efficient P uptake from water or mud means that only decrease of both will prevent growth.
8, Periphyton-dominant large stream or lake	Residence time insufficient for phytoplankton growth but clear enough for periphyton.	Medium	Medium	Susceptible to sustained P inputs. Sediment retention of P is low so should respond quickly to P decrease.
9, Large stream - shifts between phytoplankton and macrophyte dominance	Phytoplankton may shade out macrophytes, but temperature can be a critical factor.	Medium	Low	Macrophyte dominance may benefit fish.

10 , Large stream - shifts between phytoplankton and periphyton dominance	Sufficient residence time for phytoplankton which could shade out periphyton.	High	High	If little soft sediment, then low internal supply likely and decrease in P causes quick response.
11 , Oligotrophic or mesotrophic deep lake	Stratification present, but DO still present in hypolimnion. Oxic conditions bind P to sediment.	Extreme	Med.-High	Increase in P load and algal production, depletes DO and increasing P from sediment causes eutrophication.
12 , Estuary	Sufficient P is supplied from marine water	Low	Low	Not important
13 , Stratified deep lake - hypolimnetic P transfer	P build up in anoxic hypolimnion which is commonly transferred to epilimnion by wind turbulence	Med.-High	Low	Difficult to restore, requires chemical treatment (e.g., Cu addition) or hypolimnetic aeration.
14 , Stratified deep lake	Less hypolimnetic P transfer than above. External loads important. P released to algae at turnover	High	Low	P loads need to decrease to prevent further eutrophication, but dependant upon loading history.
15 , Polymictic lake	Short-term stratification means many mixing events move P to epilimnion, but depends on trophic state	High-Low	High-Low	Sensitive if oligotrophic and insensitive if eutrophic determines effort that should go into P management
16 , Shallow lake - macrophyte dominant	Macrophytes keep SS and P low in water column	High	Low	Macrophyte removal hard without introducing phytoplankton and increasing SS. Leave as is.
17 , Shallow lake - few macrophytes	SS and P higher than above, mean loadings irrelevant.	Low	Low	Ascertain internal – external loads for management.
18 , Shallow lake with short residence time, high turbidity	Residence time insufficient to allow phytoplankton to grow. Light limited.	Low	Low	Sediment and P removal may help but require identification of CSAs.



¹ Stream order is counted from headwaters to ocean and only increase when two stream of the same order meet.

² Nephelometric turbidity units (inorganic turbidity only).

³ Osgood index is defined as $z/A^{0.5}$ where z is the mean depth of the water body and A is surface area (km²).

Fig. 5 The classification of surface water bodies. Bolded numbers refer to Table 4. Adapted from Newton & Jarrell (1999).

Commentary: Table 5 shows the relative environmental and economic effect of decreasing N and P for an average dairy farm in the Bog Burn catchment, Southland, New Zealand. Compared to an average farm in the Bog Burn, practices such as the use of a pad during winter to minimise the return of excreta to the soil decreased N losses by about a third. A similar decrease was noted for P with the use of deferred irrigation, which restricts effluent application to land to times of the year when the soil can soak it up, minimising the potential for drainage and overland flow. However, research by Bewsell and others (e.g., Bewsell et al. 2007; Kaine et al. 2005) have shown that in a non-regulatory environment best management practices are only used if they are cost neutral or beneficial. Hence, each should be assessed with regard to economic impact. Data in Table 5 shows that some of the BMPs are cost beneficial or cost neutral.

Table 5 Economic and environmental assessments of different best management practices to decrease nutrient loss in an average dairy farm in the Bog Burn catchment, Southland, New Zealand (adapted from Monaghan et al. 2007b).

Management	Farm MS production (kg MS ¹ yr ⁻¹)	Farm N or P loss in (kg ha ⁻¹ yr ⁻¹)	Farm EBIT ² (\$) 000's) at \$4.50 kg ⁻¹ MS	EBIT (\$) kg ⁻¹ N or P lost
<i>N management systems</i>				
Current	231,931	30	488	62
Winter pad ³	231,931	20	491	113
Nitrification inhibitors ⁴	250,381	28	538	84
Low input	207,640	23	471	80
<i>P management systems</i>				
Current	231,931	1.31	488	1 422
Optimum Olsen P	231,931	1.19	501	1 607
Deferred irrigation ⁵	231,931	0.89	485	2 080
Low rate effluent application ⁶	231,931	0.92	484	2 008

¹milk solids;

²Earnings before interest and tax;

³current wintering costs of \$162 cow⁻¹ yr⁻¹ were substituted with a covered feed pad system costing \$470 cow⁻¹ to construct, assuming an annual 8% opportunity cost of capital, \$107 cow⁻¹ yr⁻¹ for winter feeding costs, \$22 cow⁻¹ yr⁻¹ for maintenance/cleaning, and a depreciation period of 25 yrs;

⁴calculated at \$63 per application, or \$126 per annum;

⁵assumes 12 wks effluent storage are required, costing \$45 cow⁻¹;

⁶includes 12 wks effluent storage and an upgrade of the current rotating twin-gun travelling effluent irrigator to a low rate applicator, costing approx. \$50 cow⁻¹.

However, the question asks if this will translate into a measurable improvement in water quality. Data for nutrient loads were used to calculate the likely effect on periphyton growth using the equations of Biggs et al. (2000a; see Table 6). If N concentrations are used, periphyton concentrations > 600 mg chlorophyll-a m⁻² are estimated to occur during summer. This is well beyond any guideline for benthic biodiversity or trout habitat and angling (50 and 200 mg chlorophyll-a m⁻², respectively). However, it is important to note that these equations should be used to estimate periphyton growth via the limiting nutrient. In the Bog Burn, N:P ratios in Table 5 vary from 17-31, suggesting that biomass is P-limited. The corresponding periphyton concentrations vary from 501-484 mg chlorophyll-a, much less than concentrations based on N alone, and significantly less than if no BMP had been used. While the BMPs listed in Table 5 can improve water quality in the Bog Burn, the economic vs. environmental value of the catchment should be considered before a blanket approach to regulation is taken (Quinn et al. 2008). In the case of the Bog Burn, Monaghan et al. (2007a) note that farm returns were considered as important as trout spawning and habitat. In addition, values extended to the impact on the receiving Oreti River, which given the much large flow and lower nutrient concentrations, would be minimal.

Table 6 Estimated effect on periphyton growth of farm best management practices (from Table 5) for an average dairy farm in the Bog Burn catchment, Southland, New Zealand.

Best management practice	Median stream N or P concentration ¹ (g m ⁻³)	Periphyton growth ² (chlorophyll-a; mg m ⁻²)
<i>N management systems</i>		
Current	171	741
Winter pad	114	605
Nitrification inhibitors	160	716
Low input	131	649

P management systems

Current	7.5	489
Optimum Olsen P	6.8	467
Deferred irrigation	5.1	404
Low rate effluent application	5.3	411

¹Calculated as the quotient of estimated N or P load and median flow rate within the Bog Burn (175 L s⁻¹);

²Uses N- and P-limited equations from Biggs et al. (2000). For N this is - $\text{Log}_{10}(\text{maximum } chl-a) = 4.285(\text{log}_{10}Da) - 0.929(\text{log}_{10}Da)^2 + (0.50\text{Log}_{10}\text{DIN}) - 2.946$, and for P this is - $\text{Log}_{10}(\text{maximum } chl-a) = 4.716(\text{log}_{10}Da) - 1.076(\text{log}_{10}Da)^2 + (0.494\text{Log}_{10}\text{DRP}) - 2.741$; Da is the days of accrual or the mean time between floods calculated as $(1/\text{FRE3})^{3.65}$ with a flood frequency (FRE3) of 5.85 times a year.

In contrast to this example, McDowell et al. (2008) present modelled estimates of N and P losses for a 100 ha sheep or dairy farm in Southland. Nitrogen to P ratios indicates that in both farm systems P would be limiting algal productivity in water originating from these farms. However, with time the degree of P-limitation has increased due to increased N losses and a decrease in P losses due to improved effluent management from pond, many decades ago, to land application now. The corresponding periphyton response would be halved and would be further decreased if low rate effluent application was used. This indicates that in many situations there is much potential to improve surface water DRP quality with BMPs.

Q. Is a limit on soil Olsen P concentration being considered as one of the suite of soil quality indicators to help Regional Councils to mitigate surface water quality pollution?

A. Some Regional Councils indicated that they are looking at targeting P losses to improve surface water quality. This ranged from a limit on soil Olsen P concentration, to using low water soluble P fertilisers or to better education of farmers. It was mentioned that some areas that currently have rules regarding N inputs or losses don't have a limit for P and that this may be missing the fuller picture of surface water quality and nutrients.

Q. Can a business model (education vs. regulation) be used for environmental good?

A. There is much debate over the use of education or regulation to enact surface water quality aims. In the US, a mix of both methods is used, whereas in Europe there is more regulation. In the short-term, regulation is probably the only method of

enacting surface water quality improvement, whereas education may work, but will take much longer, if at all. Furthermore, it is common with both methods that the majority (c. 90%) of land users will be using good practice and probably losing little nutrients. However, the remaining 10% can negate any potential improvement. Unfortunately, in these cases only regulation will ensure water quality improvement. One way of prioritising policy around management practices was presented by Pannell (2008). Using Pannell's analysis, BMPs can be categorised according to benefit to the farmer and benefit to the community. If a BMP is of benefit to both the farmer and community then it lies in the top right of the diagram where extension is most appropriate. However, while many BMPs have a community-good they represent a cost to the farmer; these would sit in the top left of the diagram where a positive incentive is appropriate. The analysis in Fig 6 includes a diagonal line which represents the point at which benefits to either the farmer or community are negligible compared to the cost to the other party. Best management practices that mitigate P loss tend to sit in the region where positive incentives should be used.

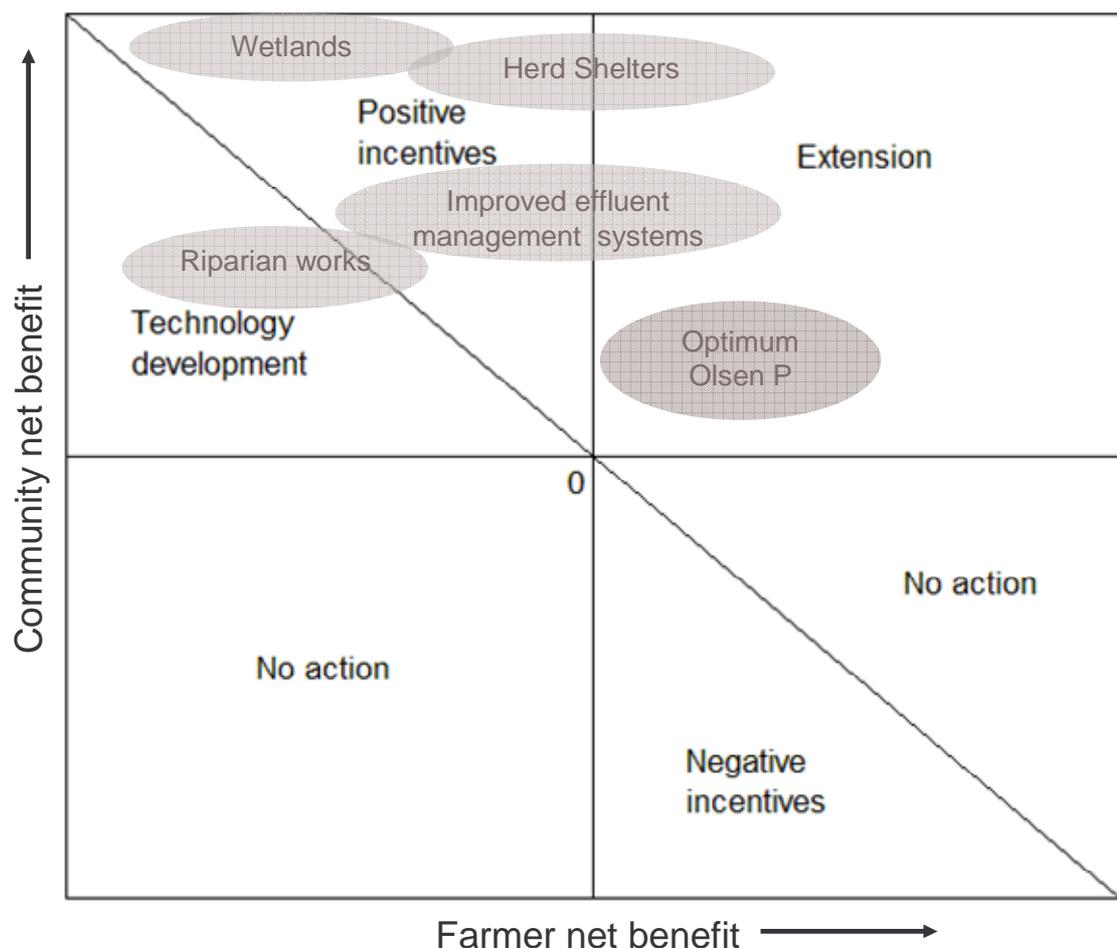


Fig. 6 Classification of policy tools for best management practise uptake to mitigate P loss to water (adapted from Pannell 2008).

Overseas experience has confirmed that the uptake of best management practices is improved via schemes that buffer the farmer from any potential cost. For instance, in the US the Natural Resource Conservation Service provides grants for the establishment of riparian areas. There are also several voluntary environmental compliance programs like the one between Environmental Protection Agency and National Pork Producers Council. Several states also have assistance programmes. Here in New Zealand such pseudo-subsidies are unrealistic due to cost to local or central government, although some schemes for riparian planting do exist. As a consequence we have relied on education with mixed results. Much of this is attributable to the cost of BMPs, which means that in order for any BMP to be utilised they must be proven not only in environmental terms, but also with economics in mind.

It would appear that many of the findings from overseas have been incorporated with free market ideals of protecting supply, market share and branding. As an example, the Dairying and Clean Streams Accord (Fonterra Co-operative-Regional Councils-Ministry for the Environmental-Ministry of Agricultural and Forestry 2003) was precocious in taking the first step to ensure that supply was maintained without large scale prosecution of poor performing suppliers, while also providing a vehicle to showcase environmental stewardship to markets overseas. Such efforts should be applauded, despite some contention over its application, uptake or effectiveness, but must be complemented with additional measures that take environmental stewardship to the next level. Once the easy steps like stream fencing have been done there are still many practices that can be improved to ensure that annual improvements in productivity are achieved without damaging surface water quality.

6.4 Conclusions

Mitigation of P losses from land and the potential impacts once in surface water ways requires a good understanding of the processes involved. While recent work has highlighted the important role of P in New Zealand's surface waterways, research to trace the source of P to its origin is lacking. Compared to overseas, New Zealand is dominated by grazed pastoral agriculture, which means that many of the potential techniques and mitigation strategies are not applicable. The behavioural characteristics and management of grazing animals carry with it unique problems for surface water quality because animals affect both the source and transport pathway of P loss differently during the year. Efforts to mitigate P loss thus far have focused on fencing off streams and improving effluent management systems on dairy farms. While

these represent significant gains in environmental stewardship of the land and surface water, it must be recognised that overall nutrient management (including maintenance of appropriate soil P status, and stock grazing management to minimise P transfer from paddocks to streams) still remains important on dairy farms, and the location of other farming systems within a catchment can also have significant deleterious impacts on surface water quality. This highlights the need to both trace the source of P within a catchment and to determine the suitability of a farming enterprise to a catchment and the likely impact. For instance, the mitigation of P loss from a dairy farm in a eutrophic catchment may not have much of an effect compared to mitigation in an oligotrophic catchment. At the same time this should be put into context with the effect or impact of N losses, the goals of the community, and the value of the waterway for farming and environmental uses. These issues present a complex problem to those that are charged with maintaining New Zealand's surface water quality. While in the medium-term education to mitigate P losses may be effective in improving surface water quality, evidence suggests that significant improvements can only be achieved in the short-term by appropriate regulation. Consequently, one of the major tasks is improving the flow of good-quality science into policy.

7. Acknowledgements

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9. Appendix I

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SURFACE WATER QUALITY AND NUTRIENTS: WHAT SHOULD THE FOCUS BE?

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Abstract

Under the right light and flow conditions, the unwanted growth of weeds and algae is generally controlled by the bioavailability of nitrogen (N) and phosphorus (P). Bioavailability is determined by the concentration and form of N and P, and by their relative abundance (i.e. the N:P ratio). Data from 1100 regional council sites monitored for the last 10 years was used to extract sites with median nitrate/nitrite-N (NNN), dissolved reactive P (DRP), total N (TN) and total P (TP) concentrations in excess of New Zealand guideline concentrations, and the ratio of dissolved inorganic N (ammoniacal-N + NNN):DRP and TN:TP. The number of sites exceeding the guidelines varied from 40-65% depending on the parameter. The Redfield ratio was used group the data into three nutrient limitation classes: ratios <16:1, between 16:1 and 30:1 and >30:1 were predicted to be N-limited, co-limited and P-limited respectively. The ratio for dissolved nutrients is used for lotic (rivers and streams) systems while TN:TP is used for lentic (lakes, reservoirs, estuaries) systems. The DIN:DRP ratio indicated that P-limitation was the most frequent scenario in New Zealand (76% of sites P-limited, 12% N-limited and 12% co-limited). The DIN:DRP ratios were then combined with median concentrations to predict periphyton densities in a year for the relevant scenario (N- or P-limited, or the average of N and P-limitation where the ratio was between 16 and 30). This indicated that 49 sites exceeded the periphyton guideline for protecting benthic biodiversity (50 mg chlorophyll-a m⁻²). The predominance of P-limitation of New Zealand rivers and streams suggests that a cost efficient approach may be to focus on mitigating P losses more than N losses, but this should only be done in the most extremely P-limited cases. Otherwise, water quality may be impacted if there is either a sudden input of the limiting nutrient or if downstream areas respond to the other nutrient. The prudent approach is to mitigate both N and P losses at their sources.

Introduction

The supply of nutrients is a key factor in the proliferation of aquatic weeds and algae. Bioavailability of either N or P often limits growth. Bioavailability can be broken down into three parts: the ratio of N:P for optimal growth, the concentration of N and P and the chemical form of N or P.

In 1963, Redfield published data that indicated a molar ratio of N:P of 16:1 was required for growth: if more than 16 moles of N are present for each mole of P, then growth is likely P-limited, if less than 16 moles of N are present for each mole of P, then growth is likely N-limited (Redfield et al., 1963). On a weight basis, the Redfield ratio is 7:1. The Redfield ratio should always be confirmed by measuring N:P in biomass and against the "gold standard" of bioassays. Such testing may indicate limitation by N or P at ratios different to those indicated by the Redfield ratio due to competition or variable nutrient requirements among periphyton species (Klausmeier et al., 2004). This has been corroborated by bioassays in a few streams

and lakes in New Zealand (e.g., Francoeur et al., 1999; Biggs 2000a). Given this variation, the conservative approach is to use N:P ratios only as an indicator of extreme N- or P-limitation, where limitation by the other nutrient is highly unlikely. Hence, in this paper, N-limitation is considered likely at N:P < 16:1, P-limitation likely at N:P > 30:1.

The ratio of N:P in freshwaters must also be put in context with concentrations. For instance, if elevated concentrations of N and P exist then periphyton blooms may occur even if one nutrient is limiting. Consequently, unless one nutrient is in extreme limitation then it is important to manage both inputs of N and P.

The final aspect of bioavailability refers to chemical form. Both N and P exist in dissolved (or soluble) and particulate form. The distinction in terms of bioavailability to algae is via kinetics. Dissolved inorganic forms of N and P are immediately available for uptake by algae, while particulate nutrients are transported downstream. Hence, the ratio of dissolved reactive P to dissolved inorganic N (DIN: nitrate-N and ammoniacal-N) is used in fast flowing lotic systems (e.g., streams and rivers). In contrast, in slow flowing streams or lentic systems (e.g., lakes, reservoirs, ponds, marshes), particulates may settle out and release dissolved nutrients via biotic (e.g., enzymes) or abiotic (desorption, solubilization) mechanisms. Hence, the total N to total P ratio is more commonly used for lentic systems.

In this paper, we present a preliminary analysis of data from 1100 surface water quality sites sampled over the last 10 years. These represent all sites monitored by regional councils in New Zealand and cover a wide range of landuse, elevation, climate and flow regimes. The vast majority of these are for lotic systems, which will be the focus of this paper. Data is presented for median dissolved and total N and P concentrations. These will give an indication of the importance of either N or P in limiting biomass production. Our analysis assumes that all other factors such as light and temperature are non-limiting. The data raises questions over the effectiveness of BMPs in affecting surface water quality if aimed at the wrong nutrient, and what the issue may look like in the future.

Materials and methods

Data from 1100 sampling sites around New Zealand were imported into a ProQuest database. Data was from the period 1996-2007, except for Environment Southland, which was only able to supply data for 1996-2002. Summary statistics were generated (range, mean, median and normality) for sites with a full set of parameters (nitrate/nitrite-N, ammoniacal-N, dissolved reactive P [DRP] and total P [TP]), geographic coordinates, and with > 6 data points were used in the analysis. Approximately 700 sites met these criteria. For each site and date, DIN:DRP and TN:TP ratios were generated after converting data to molarity from weights. Where the DIN:DRP ratio was < 16 or > 30 the P- or N-limited equations of Biggs (2000a) were used to estimate maximum periphyton growth during a year. This assumed a FRE3 statistic of 6 flood disturbances, or storm events, per annum that remove accumulated periphyton. Data are presented in relation to DIN:DRP and TN:TP ratios and periphyton guidelines (assumed to be 50 mg chlorophyll a m⁻² in periphyton communities dominated by diatoms).

Results and Discussion

The percentage of sites exceeding the ANZECC (2000) guidelines for lowland rivers in slightly disturbed ecosystems varied from 39% for TP in the South Island to 75% for DRP in the North Island (Table 1). Nationally, the number of sites exceeding guidelines varied from about 40-60% among the parameters (Table 1). These guidelines were established relative to

“pristine” riverine systems largely unaffected by agriculture and do not give an indication of the potential water quality impact as sites may be limited by one nutrient or another.

Table 1. Count and percentage of total sites in the South and North Islands and nationally of sites exceeding their respective ANZECC (2000) water quality guideline¹.

Parameter	South Island		North Island		New Zealand	
	Count	Percentage	Count	Percentage	Count	Percentage
DRP	179	46	263	73	442	58
TP	142	39	143	41	285	39
NNN	139	41	237	72	376	55
TN	153	44	144	44	297	43

¹guidelines are: DRP = 0.010 mg P L⁻¹; TP = 0.033 mg P L⁻¹; NNN = 0.444 mg N L⁻¹; TN = 0.614 mg N L⁻¹.

Sites with DIN:DRP ratios >16 or >30 for DIN:DRP indicate, respectively, co-limitation and P-limitation of periphyton growth in lotic systems. This was the case in most sites, occurring at 87, 83 and 85% of sites in the South Island, North Island, and nationally, respectively (Table 2; Figure 1). For TN:TP ratios, the number of sites that were N-limited increased, but was still in the minority occurring at 10, 35, and 22% of sites the South Island, North Island, and nationally (Table 2). Nevertheless, TN:TP ratios are not relevant for most sites since the vast majority were lotic systems.

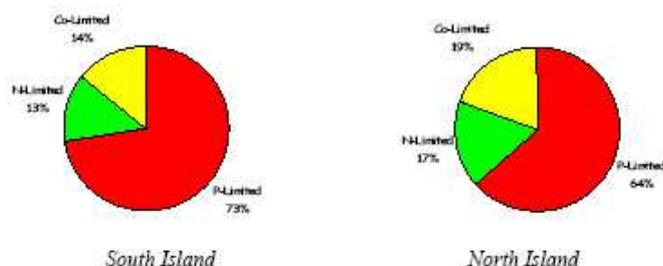


Figure 1. The percentage of sites in the South and North Islands predicted to be N-limited, P-limited, or co-limited.

Table 2. Count of total sites in the South and North Islands and New Zealand predicted to be N-limited, P-limited, or co-limited. (mean N:P ratios > 30, < 16 and between 16 and 30, respectively). Numbers refer to DIN:DRP and TN:TP for riverine and lotic systems, respectively.

Parameter	South Island		North Island		New Zealand	
	DIN:DRP	TN:TP	DIN:DRP	TN:TP	DIN:DRP	TN:TP
P-Limited	265	234	219	105	484	339
N-Limited	49	36	60	114	109	150
Co-Limited	52	71	67	109	119	180
Total	366	341	346	328	712	669

Surface water quality impact is proportional to the concentration of the limiting nutrient. Hence, calculation of the maximum periphyton growth expected in a year was calculated using the limiting nutrient in question for each site and the relevant biomass-concentration equation for DIN or DRP from Biggs (2000a). Where co-limitation occurred, the DIN and DRP equations were averaged. A FRE3 statistic of 6 was used to determine the days of accrual and is indicative of a hill-fed stream (Biggs, 2000b).

The output is given in Figure 2 relative to the periphyton guideline of 50 mg chlorophyll-a m⁻² for the protection of benthic biodiversity values regardless of whether the periphyton community is dominated by filamentous algae or diatom films (Biggs, 2000b). This indicated that the guideline was only breached at 49 sites in New Zealand, 40 in the North Island and 9 in the South Island. Only two of the South Island sites were N-limited, the others were P-limited or co-limited. In the North Island 10 sites were N-limited, and the remainder were P-limited (13), or co-limited (17). These results reflect the predominance of P-limitation in the South Island, and the on-average low nutrient concentration of the data sites.

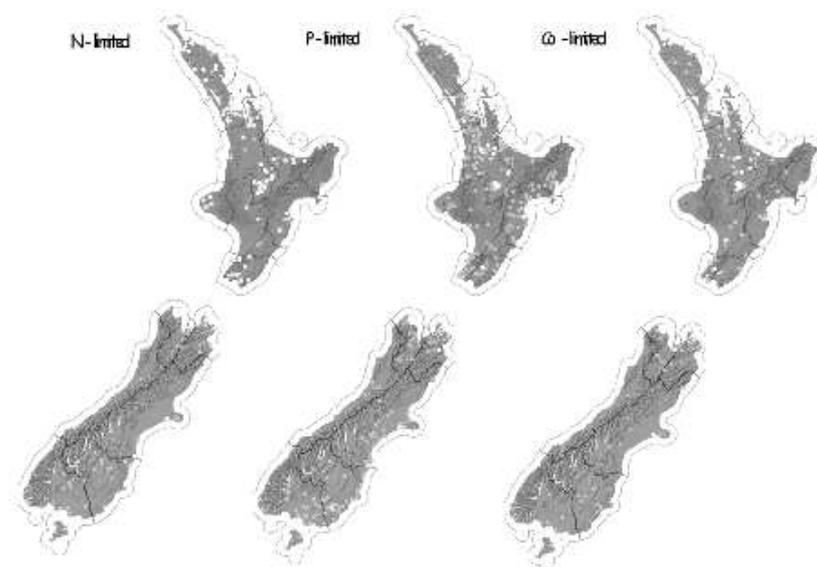


Figure 2. Sites with maximum predicted annual periphyton biomass that exceed (grey circles) or meet (white squares) the 50 mg chlorophyll-a m⁻² guideline. Periphyton densities were estimated using the equations of Biggs (2000a) and the predicted limiting nutrient, or the averaged N and P biomass equations for sites at which the DIN:DRP ratio was between 16:1 and 30:1.

Management Implications

Nitrogen and P losses operate over different spatial and time scales. Nitrogen input to many surface waters is dominated by groundwater and hyporheic flowpaths. It can take a long time (e.g., 40-85 years for Lake Taupo; Morgenstern, 2007) for N to reach surface water, which

means means that current land use practices could take many years to influence stream and lake N concentrations. In contrast, P enters surface waters via rapid transport routes like surface runoff and tile drainage, meaning P losses largely reflect current land use (McDowell et al., 2004).

Different N and P pathways raise a number of issues. The first is that land use has changed much in the last few decades. Those changes have generally been towards intensification (e.g., dairy farming), and intensive systems tend to lose more N and P than extensive or low intensity systems. As a result, while P losses may have stabilized due to quick transmission pathways, the delay in N reaching our surface waters means future N loads are likely to increase. Hence, P-limitation of our surface waters is likely to increase over time. This has significant implications for catchment management as it may be easier and cheaper to focus control or mitigation efforts on one nutrient than on both. However, doing so raises the risk that downstream waters will be negatively affected by the uncontrolled/unmitigated nutrient (Wilcock et al., 2007). This also applies to the form of N or P, whereby particulate-associated N or P may pose a problem at a different scale (i.e. in a lake or estuary). Furthermore, if the concentration of the non-limiting nutrient is high enough, then nuisance or toxic algal blooms may occur regardless of the N:P ratio.

Another issue is that effective management relies on practical solutions. These generally represent practices that cost little or make the farmer money and impose little time or effort (Pannell et al., 2006). Ideally, mitigation is best done at the source, preventing the loss of one or both limiting nutrients by minimising factors such as the direct deposition of fertiliser in waterways, trampling leading to pugging of pastures, and stock access to waterways. One way to minimise nutrient losses may be to manage a catchment management based on a cost effectiveness or \$ ha⁻¹ of nutrient conserved. Such an approach may recognise that some land uses have a potential to lose one nutrient more than another, and may promote broadcast N mitigation strategies while taking a targeted approach to areas of likely surface runoff and P losses (McDowell et al., 2004). Only in the most extreme cases of nutrient limitation is a narrow focus on controlling the sole limiting nutrient warranted.

Conclusions

The eutrophic response of algae in freshwater is controlled by the bioavailability of N and P. Bioavailability is determined by the absolute concentrations and forms of N and P, and the N:P ratio. These factors affect water quality impact by limiting periphyton growth rates. Of the sites used for our analysis, many exceeded guideline concentrations for dissolved and total N and P. However, N:P ratios indicated that P-limitation of periphyton growth was more prevalent than N-limitation. As such, the low concentrations of DRP in many streams indicated that periphyton blooms rarely exceeded the guidelines for the protection of benthic biodiversity. With time and increasing landuse, N concentrations are likely to increase in groundwater and this will in-turn feed surface water. These changes will lead to increases in the severity P-limitation of periphyton growth. Focusing on mitigating a single nutrient (e.g., P), even the limiting nutrient, is perilous should concentrations of the limiting nutrient suddenly increase or downstream is N-limited. Hence, the prudent approach is to mitigate both N and P inputs.

Acknowledgements

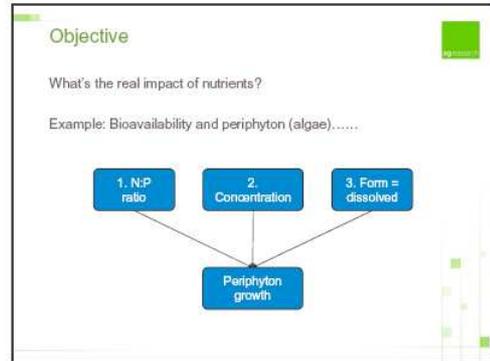
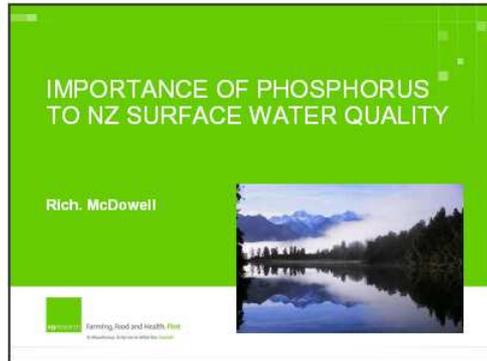
The authors thank data processors and GIS support at both AgResearch (Smale, Srinivasan) and NIWA.

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10. Appendix II

Presentation to New Zealand Phosphorus Workshop entitled: Importance of Phosphorus to NZ Zealand Surface Water Quality by Richard McDowell (AgResearch)



1: N:P ratio & limiting nutrient

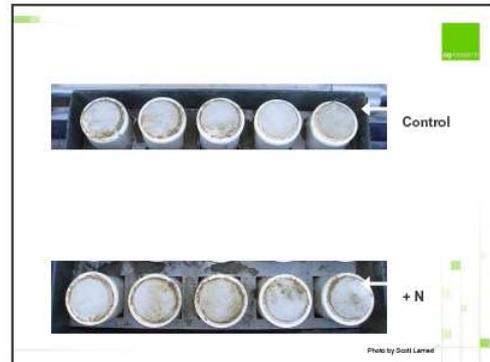
Overall what is the limiting nutrient for NZ freshwaters?

Liebig's (1855) Law of the Minimum: one nutrient can be in shortest supply relative to demand and thus one nutrient can limit biomass production at a given time.

Redfield (1963) ratio

Stoichiometry in biomass

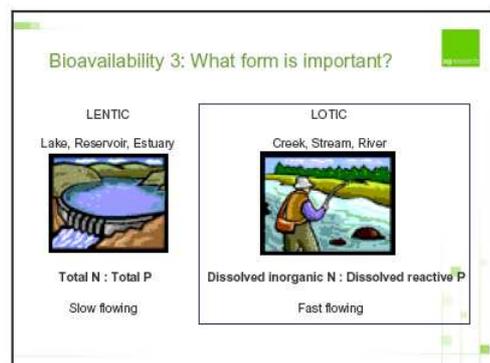
16 : 1
N P



Bioavailability 2: Concentration

N:P ratios need to be put in context of absolute N and P concentrations.

The ratio is irrelevant if both are in-excess, meaning that algal growth is excessive.



Sites

- >1100 across New Zealand
- Land use (exotic and native forest, native bush, pasture, cropping, urban, horticulture)
- Elevation ranges from 1 – 700msl
- Rainfall ranges from 230 – 5000mm
- Mean annual temperature ranges from 6.5 – 15.5°C

Caveats

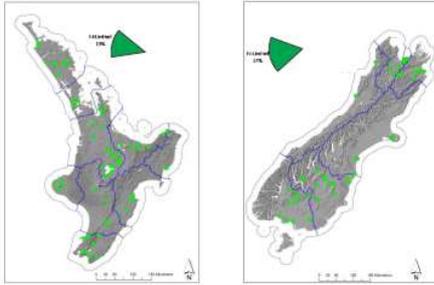
Are data site representative of "true" water quality: Possible bias towards "perceived" sensitive systems or weighted to one landuse (e.g., dairy).

Analysis assumes no influence by:

- light (turbidity, riparian shading)
- temperature
- variable low regimes
- invertebrate grazing
- macrophytes

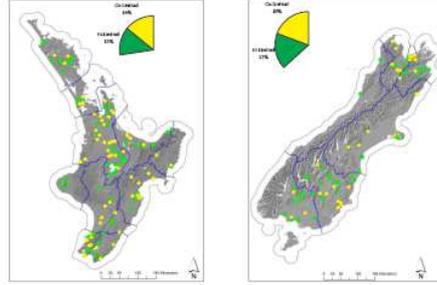
Bioavailability: N-, Co-, or P-limitation

N-limited



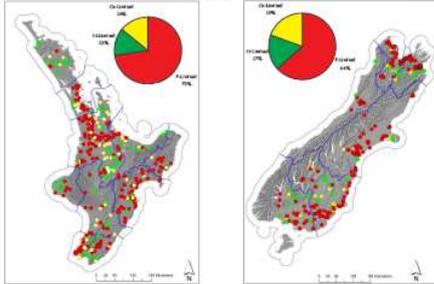
Bioavailability: N-, Co-, or P-limitation

Co-limited



Bioavailability: N-, Co-, or P-limitation

P-limited



Conclusion and questions...

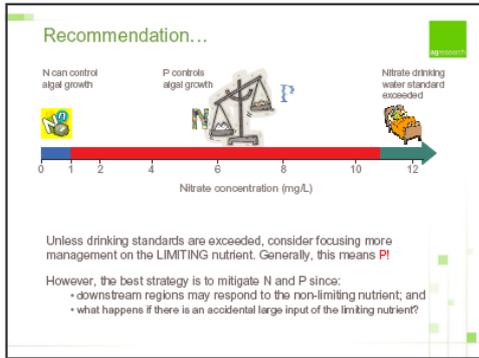
Presently, New Zealand's surface waters are largely **P-limited**.

Is there value in managing the limiting nutrient or just land uses that lose more of the limiting than non-limiting nutrient in a particular area?

- Could be a cheap and quick way of mitigating harmful water quality effects

What is the time frame for mitigation?

- N loads from current agricultural systems can take a long time to reach freshwater systems meaning water quality can take longer to improve if land use/management changes focus on only N.
- Current intensive operations means that future N loads will likely increase. Hence, so will P-limitation



- ### Acknowledgements
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 - Hawke's Bay Regional Council
 - Horizons Regional Council
 - Nelson City Council
 - Tasman Regional Council
 - Marlborough District Council
 - West Coast Regional Council
 - Environment Canterbury
 - Otago Regional Council
 - Environment Southland
 - P. Smale, MS Srinivasan
 - B. McLennan, S. Lamed

Presentation to the New Zealand Phosphorus Workshop entitled Cycling and efficiency of phosphorus transformations in New Zealand soils, by Leo M Condon (Lincoln University).

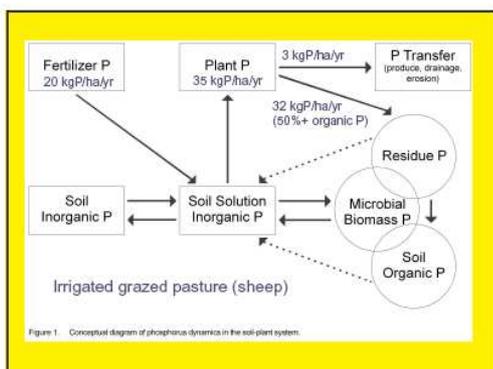
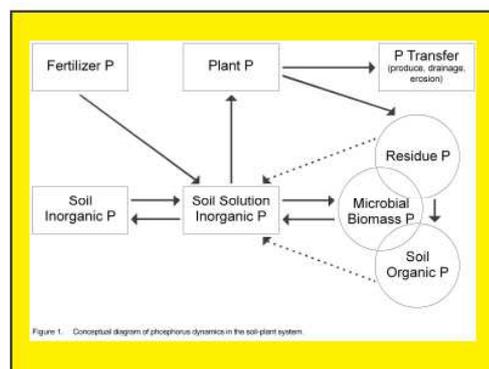
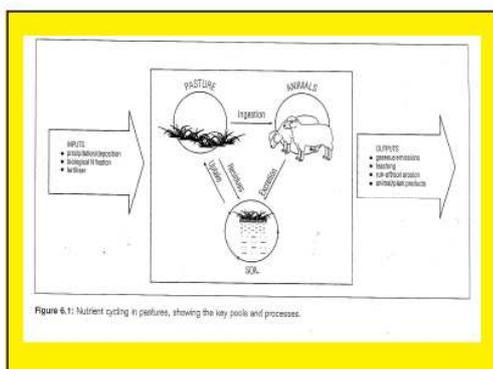
Cycling and Efficiency of Phosphorus Transformations in New Zealand Soils
 Professor Leo Condon – Lincoln University

- Nature and dynamics of phosphorus in soil-plant-animal systems
- Phosphorus requirements in relation to land use and management
- Challenges associated with improving phosphorus efficiency

National Phosphorus Workshop – Palmerston North – 29 July 2008

	pH	Organic C g kg ⁻¹	Total P mg kg ⁻¹	Organic P mg kg ⁻¹ (% total)
Egmont	5.7	53	629	485 (77)
Stratford	5.4	68	2746	1366 (43)
Mangamahu	5.3	39	401	315 (78)
Hawthorn	7.0	28	862	331 (38)
Tapee	5.1	55	1311	479 (37)
Orakei	5.3	99	1127	791 (62)
Pahia	5.2	49	663	524 (81)
Fair	6.0	35	626	242 (32)
Haruru	5.6	79	904	670 (74)
Kohimaram	5.7	36	813	568 (70)
Tanika	6.5	40	1056	509 (48)
Patea	5.7	92	1583	1051 (66)
Tu Kamehaha	5.7	50	938	606 (65)
Mania	5.2	30	136	82 (73)
Oaheka	5.4	130	597	479 (80)
Mean	5.7	57	863	548 (62)

(Chen et al., 2003)

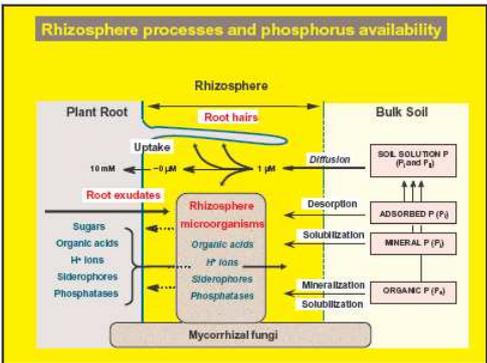
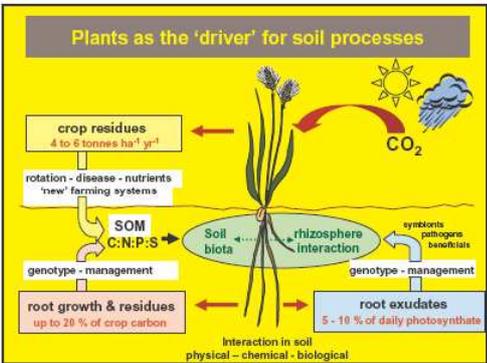
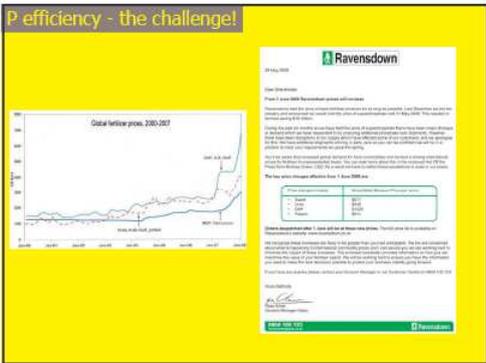




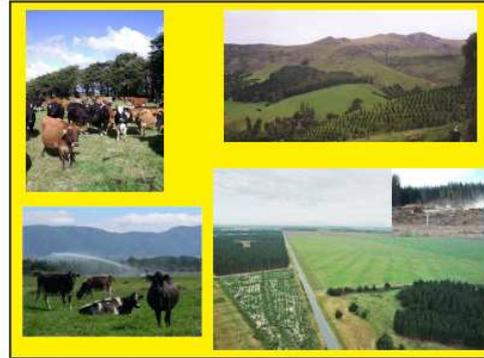
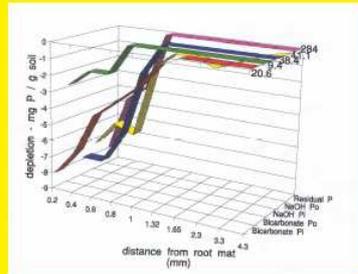
Grazed Pasture (52 years) – 25kgP/ha/yr

	0-7.5cm			7.5-15cm			15-25cm		
	C	Pi	Po	C	Pi	Po	C	Pi	Po
Onland (DLIS)	4.31	727	874	3.31	851	482	2.00	385	323
Irrigated @ 10% (7Years DLIS)	4.28	571	872	3.21	987	568	2.01	388	414
Irrigated @ 20% (7Years DLIS)	3.68	372	706	2.69	543	568	1.74	195	411
Wilderness (Dryland, no P)	4.21	339	872	3.06	288	470	2.48	330	358

Condron et al. (2006)



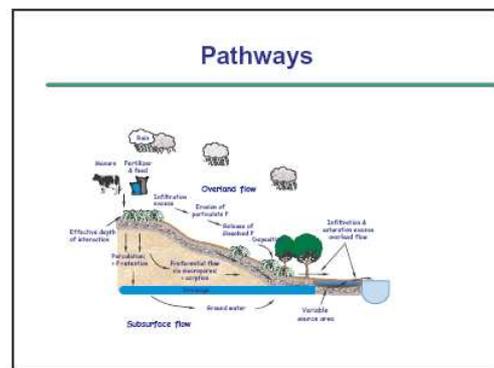
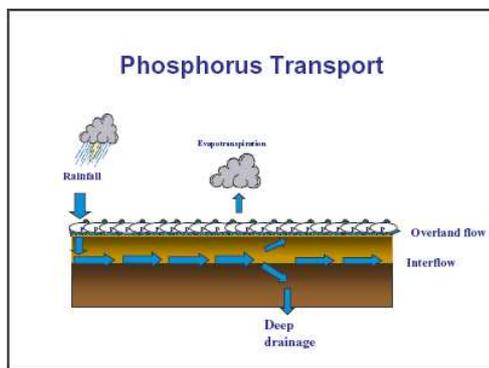
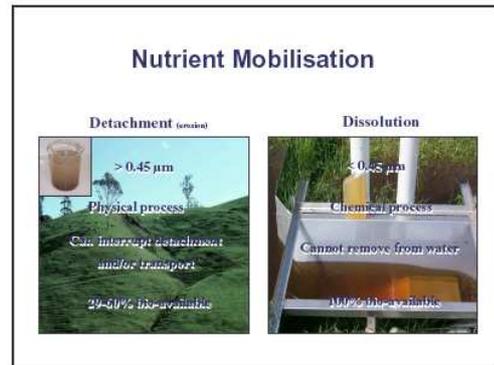
Phosphorus depletion in rape rhizosphere (14 days)
(Gahoonia and Nielsen, 1992)



Selected biological and biochemical properties determined in adjacent soils (0-5 cm) under temperate grassland and 19 year-old pine forest over a one-year period (Chen et al., 2003)

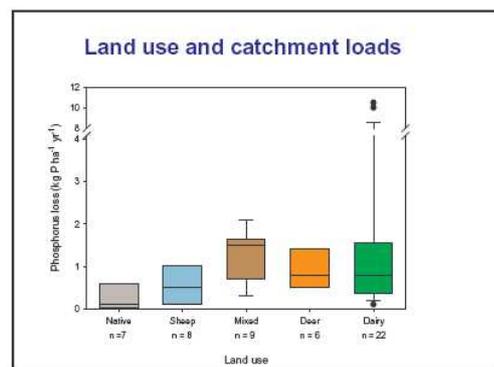
	Grassland mean (range)	Forest mean (range)
Microbial biomass C (mg kg ⁻¹)	1417 (1140 - 1705)	808 (671 - 1027)
Microbial biomass P (mg kg ⁻¹)	42 (22.6 - 51.1)	27.4 (13.6 - 42.9)
CO ₂ respiration (mg CO ₂ -C kg ⁻¹ h ⁻¹)	0.49 (0.11 - 0.84)	0.43 (0.17 - 0.68)
P-Nomicrobials (mg p-NP kg ⁻¹ h ⁻¹)	423 (277 - 596)	206 (178 - 485)
P-Dissolved (mg p-NP kg ⁻¹ h ⁻¹)	13.3 (7.6 - 16.9)	8.5 (5.0 - 12.3)
Annual P Release from Microbial biomass (kg ha ⁻¹)	13.9	16.1

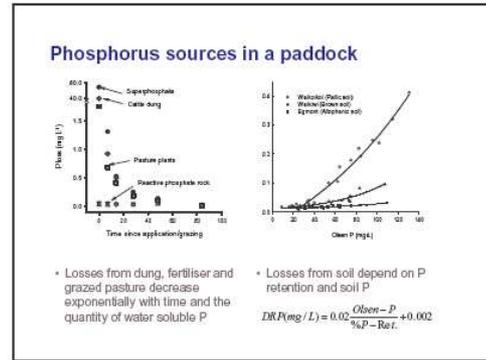
Presentation to the New Zealand Phosphorus Workshop entitled Components of a Water Quality Problem by David Nash (Victorian Department of Primary Industries)



Measured phosphorus losses

Site	Loss (kg P/ha/yr)	Loss pathway
Inchbonnie, West Coast	2.3 - 7.6	Hump and hollow
Tussock Creek, Southland	1.0 - 4.5	Mole-pipe drainage/effluent
Hillend, Otago	2.0	Winter forage crop
Toenepe, Waikato	1.0 - 2.5	Effluent ponds
Lincoln, Canterbury	0.3	Leaching
Rerewhakaia, Bay of Plenty	1.9	Lane runoff
Lumsden, Southland	1.6	Winter forage crop
Telford, Otago	3.9	Walloos
Ballantrae, Manawatu	1.5	Stream access
Winchmore, Canterbury	0.7 - 12.6	Border dyke irrigation





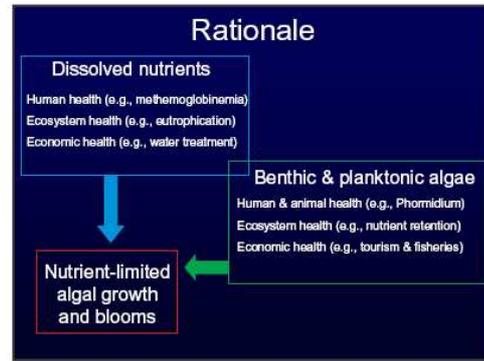
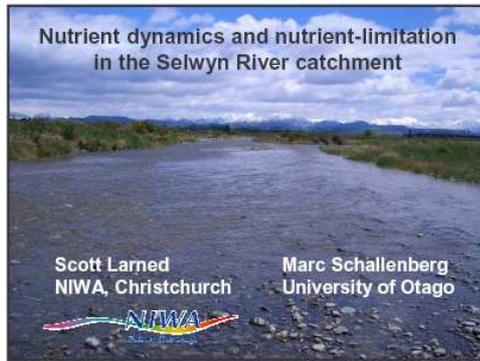
Paddock and farm loss

Source	Typical contribution (%)	Possible contribution (%)
<i>Within paddock</i>		
Soil	35	10-90
Dung	35	10-60
Fertiliser	10	1-80
Plants	20	1-30
<i>Point sources within farm</i>		
Effluent block	35	20-60
Wallows	50	10-95
Lanes/fences/tracks	25	10-50

Giving the right message???

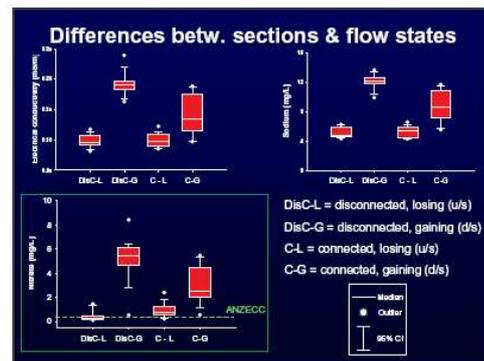
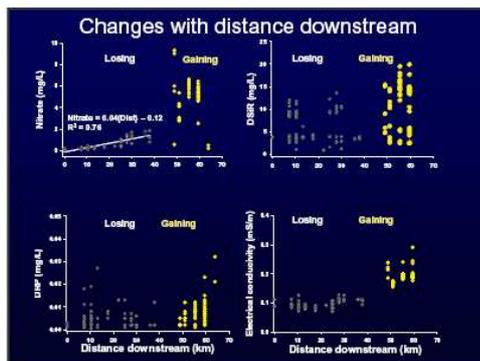
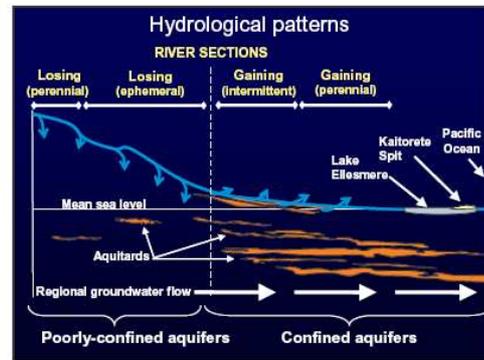
Farm A	
Milk Production	4,500 L/ha
Phosphorus exports (Olsen P = 12 mg/kg)	4 kgP/ha
Farm B	
Milk Production	12,500 L/ha
Phosphorus Exports (Olsen P = 35 mg/kg)	5.2 kgP/ha
To produce 1000 L of milk	
Farm A exports 0.9 kgP/L	Farm B exports 0.41 kgP/L

Presentation to the New Zealand Phosphorus Workshop entitled Nutrient Dynamics and Nutrient-Limitation in the Selwyn River Catchment by Scott Larned (NIWA).

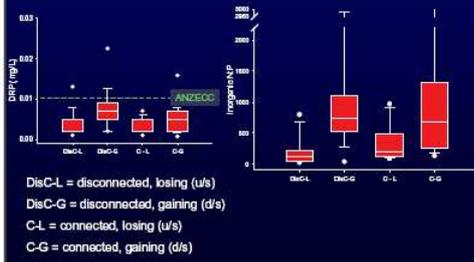


Nutrient dynamics – what controls variability?

Geology	Weathering, major ions
River & GW hydrology	Connectivity, dilution, GW inflow
River & GW biology	Retention, transformation, removal
Land use & land cover	Nutrient input, irrigation, riparian management, soil processes
Lake hydrology	Nutrient loading, residence time, opening regime
Lake biology	Retention, transformation, removal



Differences betw. sections & flow states



What do nutrient concentrations & ratios mean for river algae?



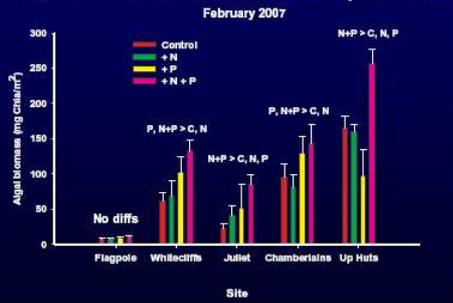
N:P ratios as indicators of nutrient-limited algal growth

- Rules of thumb
 - DIN:DRP < 20:1 N-limited, > 40:1 P-limited
 - Ln(DIN:TP) < 0 N-limited, > 0 P-limited
- Selwyn losing section* DIN:DRP 221:1 ± 292
- Selwyn gaining section* DIN:DRP 1235:1 ± 1307
- Upper Selwyn foothills DIN:DRP 49:1 ± 12
- *disconnected flow periods

Nutrient limitation – NDS experiment



Nutrient limitation – NDS experiment



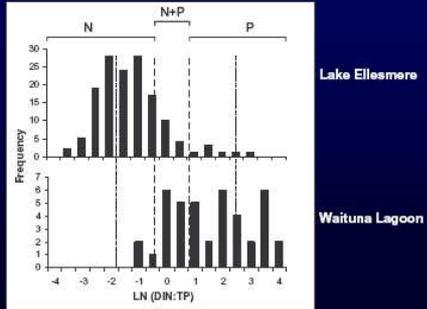
Nutrient pools and (predicted) nutrient limitation in Lake Ellesmere



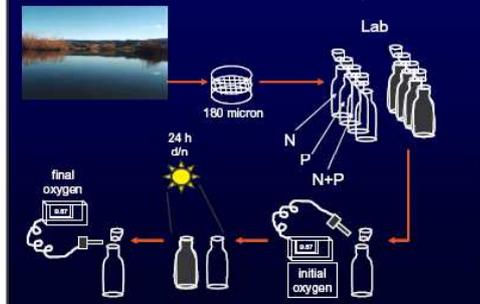
Nutrient pools (mg/L) & ratios in Lake Ellesmere

	DIN	TN	DRP	TP	DIN:DRP	TN:TP
Selwyn R	4.4	4.6	0.02	0.03	219:1	157:1
All tributaries	3.1	3.4	0.03	0.06	107:1	73:1
Lake water column	0.1	2.3	0.007	0.3	62:1	12:1
Rule of thumb					30:1	

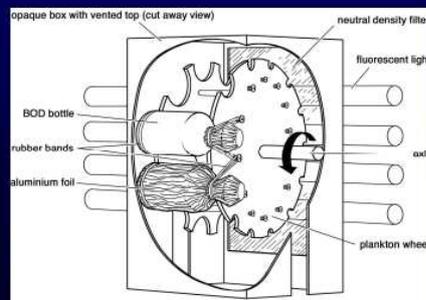
Predicted limiting nutrients



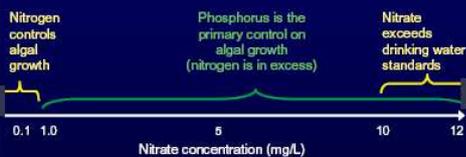
Lake nutrient limitation – bottle experiments



Light vs. nutrient-limitation



Conclusions – nutrient limited algae



DIN concentrations in Selwyn River are generally too high to limit algal growth, & below drinking water standards.

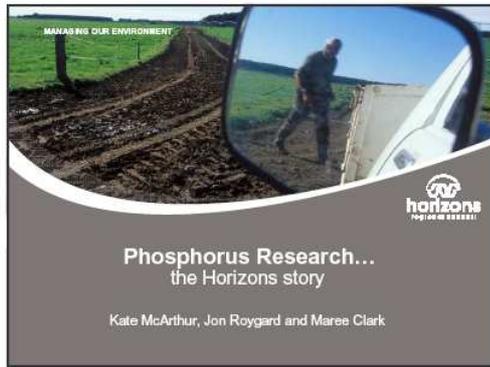
Light is most frequently limiting in Lk Ellesmere, then N or P

Recommendation: Unless drinking standards are exceeded, manage LIMITING nutrients, not just EXCESS nutrients.

Acknowledgements

- **Fieldwork**
 - Glenn Cooper, Matt Dale, Calfy Kilroy, Cary Mohlmann
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 - Sarah Braithwaite, Catherine Chague-Goff, Mike Crump, Chris Cunningham, Margaret McMonagle, Denise Rendle, Karen Robinson, Hill Laboratories Ltd.
- **Funding**
 - FRST Water Allocation Programme (NWA)
 - FRST Sustainable Groundwater Allocation Programme (LVL)

Presentation to the New Zealand Phosphorus Workshop entitled Phosphorus Research: the Horizons Story by Kate McArthur (Horizons Regional Council).

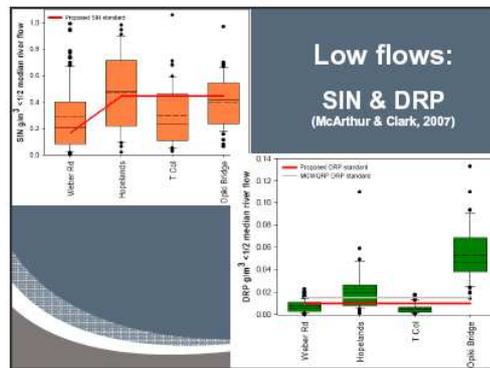


Limiting Nutrients Workshop 2006: key messages (Wilcock et al., 2007)

1. N and P managed in all rivers: limiting nutrient changes between connected catchments and within the same waterway spatially and seasonally (and with flow...)
2. High background concs. of 'non-limiting' nutrient contributes to periphyton if control of 'limiting' nutrient fails
3. Year-round control of N and P: periphyton growth and vigour determined by preceding nutrient conditions and upstream residual algae
4. Not all rivers and streams need nutrient management: e.g. rivers with soft substrates. But... management still needed to reduce nutrient pools in sediments and protect downstream reaches from enrichment

Nutrient standards in the Proposed One Plan...

- Proposed WQ standards defined soluble P and N concentrations for all waterbodies at flows < 3 x Q₅₀ (Ausseil & Clark, 2007)
- DRP conc. Standards Water Management Zone (WMZ) or sub-zone specific
- DRP & SIN standards for each WMZ were influenced by:
 - Values within the WMZ
 - Downstream values
 - Desired periphyton standards
 - River substrate of the WMZ
 - Downstream substrate type
 - Geological influence on background DRP
 - Current state of water quality (N and P)
 - Flow regime (i.e. volcanic plateau rivers)
 - Expert opinion and ANZECC Guidelines



Nutrient status of the Region's Rivers (McArthur & Clark, 2007)

SOE Monitoring Site	SIN		preliminary SIN source	DRP	
	meets SIN standard at 1% MSLF	meets SIN standard at 1% median		meets DRP standard at 1% MSLF	meets DRP standard at 1% median
Matawhiri at Waihou Road	✗	✗	non-point	✗	geology/catchment
Matawhiri at Hapeke	✗	✗	non-point	✗	point source
Mangatawhiri at SH2	✗	✗	non-point	✗	point source
Matawhiri at Teachers College	✗	✗	non-point	✓	✓
Matawhiri at Opeke Bridge	✗	✗	point source	✗	point source
Oroua at Awakuri Bridge	✗	✗	point source	✗	point source
Waipareira at 'The Machine'	✓	✓	-	✗	natural geology
Hautapu at NWA station Tahape	✓	✓	-	✗	geology/catchment
Hautapu upstream Rangitikei	✓	✓	point source	✗	point source
Rangitikei at Mangaroa	✓	✓	-	✗	point source
Rangitikei at Vinegar Hill	✓	✓	-	✓	✓
Whangarei at Cherry Grove	✗	✗	non-point	✓	✓
Whangarei at 'The Island'	✗	✗	non-point	✓	✓
Whangarei at Pipiki	✗	✗	non-point	✗	point source
Whangarei at Kauripara	✗	✗	non-point	✗	point source
Mangakino at DOC Headquarters	✗	✗	non-point	✗	natural geology
Owhango at Branscombe Bridge	✗	✗	non-point	✗	point source

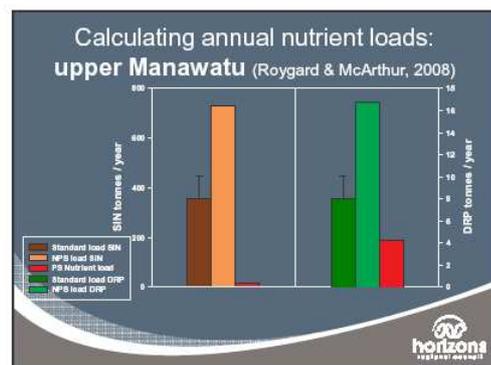
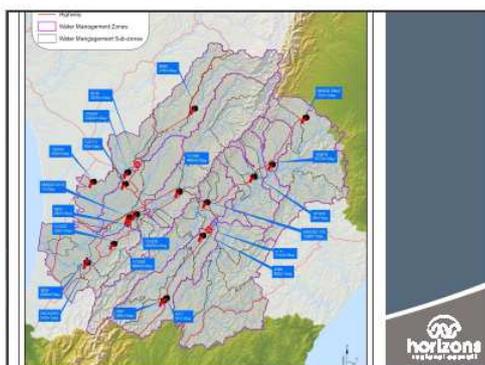
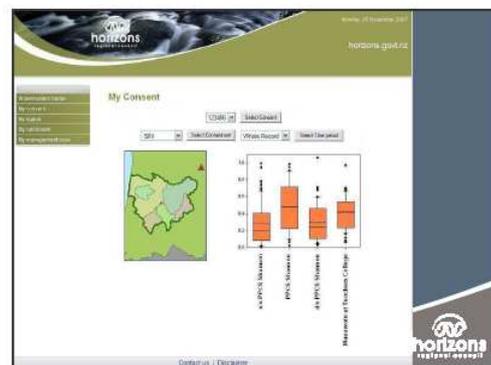
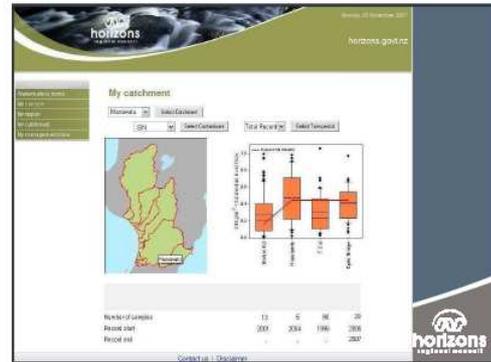
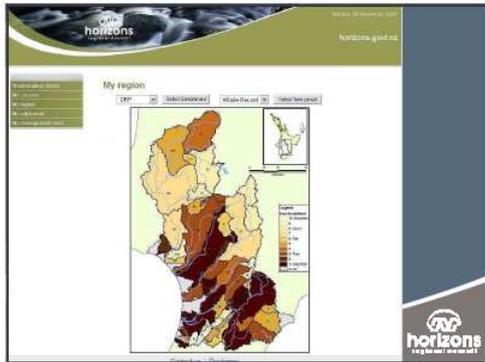
Key: ✗ Does not meet the standard, ✓ Meets the DRP standard, ✓ Meets the SIN standard

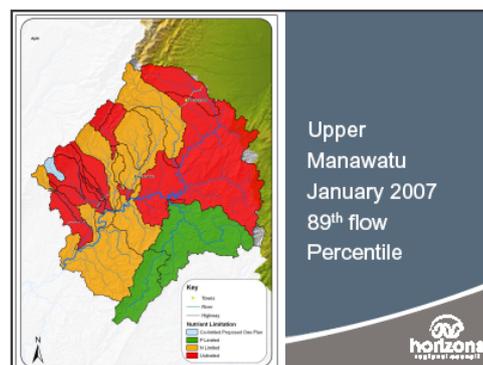
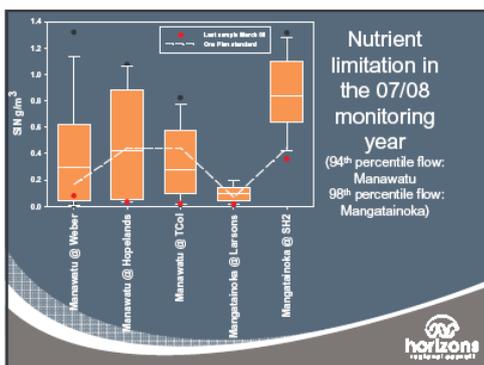
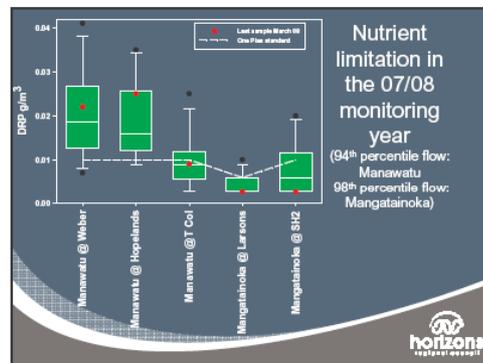
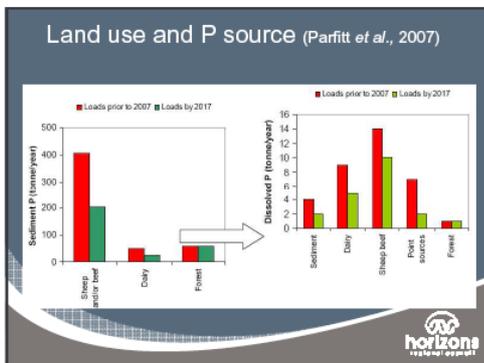
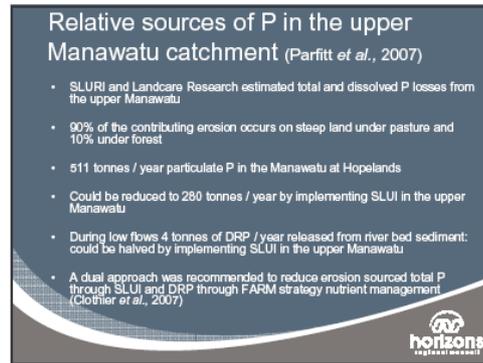
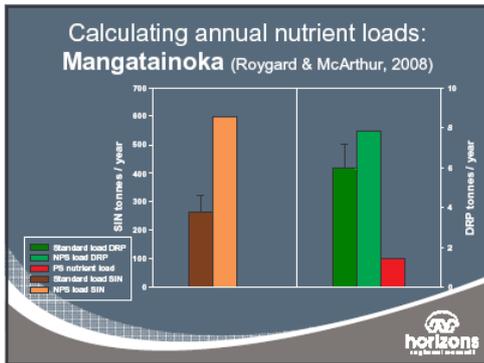
PS vs. NPS: SOE, discharge monitoring and WQ website development

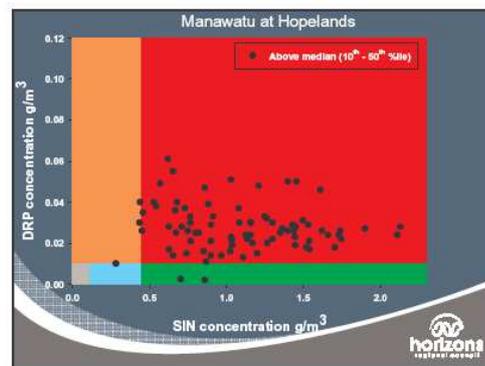
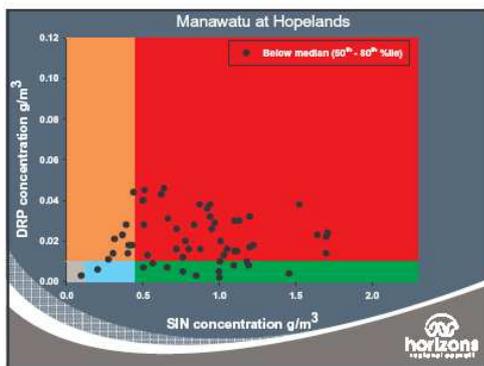
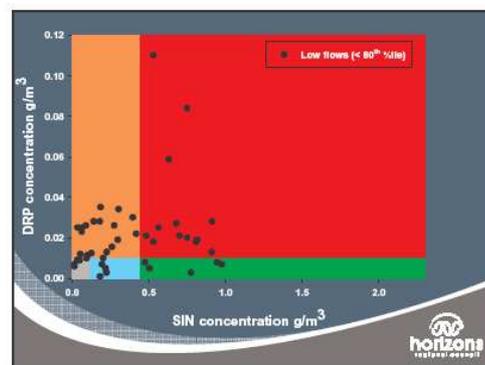
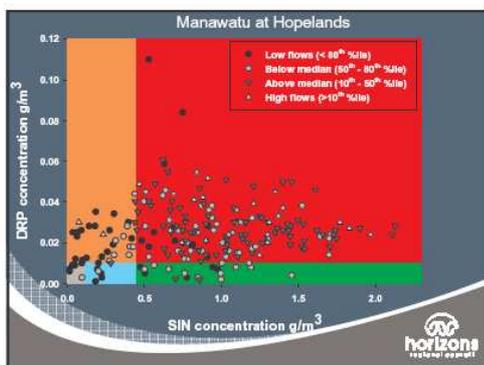
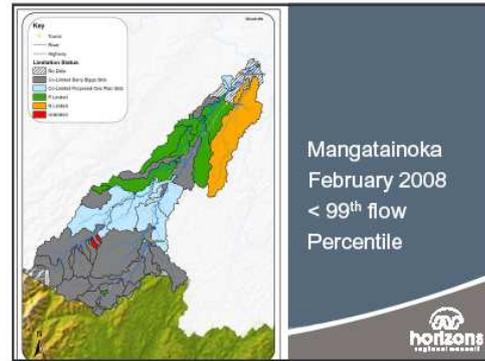
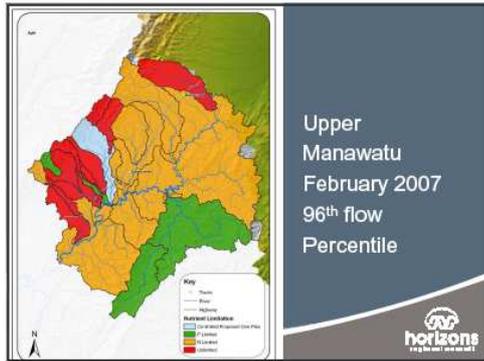
- SOE and PS monitoring on the same day, under the same flow conditions within WMZ
- Flow gauged at the time of sample collection if no flow recorder at site
- Effluent volumes are to be continuously monitored and telemetered to HRC
- Relative loads can then be determined 'on the fly'
- Data from SOE and compliance monitoring should be publicly available...

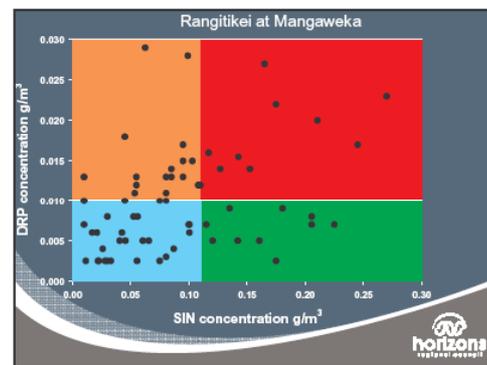
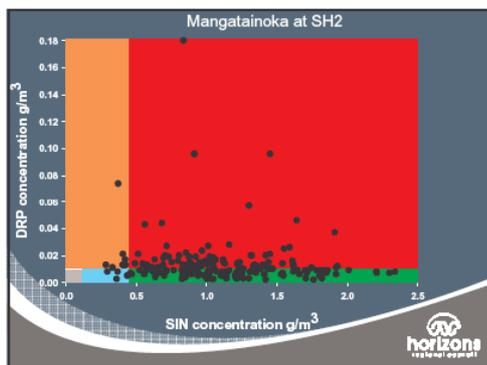
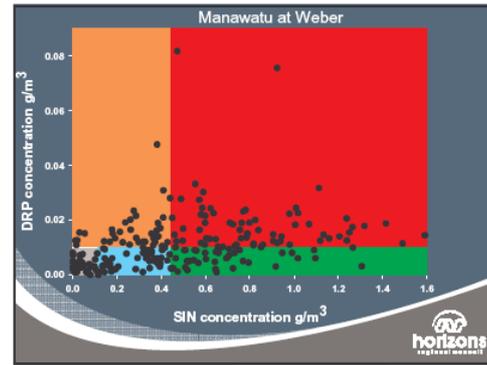
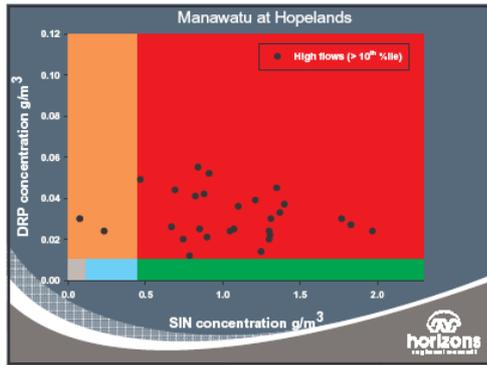
...preferably on the web, in near real time...

Due at the end of August!









Acknowledgements:

Jon Roygard, Maree Clark, Chris Veale, Jeff Cooke, Brent Watson, John Phillips, Olivier Ausseil, Chris Arbuckle, Graham Sevicke-Jones, Ned Norton, Roger Parfitt, John Dymond, Anne-Gaelle Ausseil, Brent Clothier, Markus Deurer, Allan Gillingham, Ross Gray, David Houlbrooke, Alec Mackay, Richard McDowell, Andrew Carran, Ross Gray, Glyn Francis, Maitland Manning, Steve Green, Andrew Manderson, Bob Wilcock, John Quinn, Scott Larned, Chris Hickey, Russell Death, Barry Biggs

EnviroLink Fund

