Tracing critical source areas of phosphorus in grassland catchments

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by
Gina Marie Lucci

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Candidate: Gina Marie Lucci **Date:** 26 April 2011

Supervisor: Prof Leo M Condron **Date:** 26 April 2011

The trouble doll

is not moving mountains

but digging the ground that you're on

Jakob Dylan

Something Good This Way Comes

Abstract

Grassland farming systems are an important contributor to the New Zealand economy, however recent expansions in dairying have lead to increasing concern about its effects on water quality. Phosphorus (P) and sediment losses from intensively farmed dairy pastures can impair surface water quality. Previous research has shown that loading from agricultural catchments is mainly from diffuse sources. However, an alternate hypothesis is that there are small areas within fields that can act as point sources during storm events, due to high concentrations of P in the soil from excreta. It is important to locate and understand source areas of P in agricultural catchments to effectively target mitigation strategies and decrease losses to surface waters. This thesis investigated the relationships between identified sources of P and sediment in grazed catchments, and the transport and transformation processes of P at different spatial scales and during different seasons.

One of the first steps in mitigating the loss of P and sediment was to determine where in a field the potential for P loss was greatest. Once the sources of P were found at the field scale (< 10 ha) the next step was to investigate the importance of the sources at the farm (10-500 ha) and catchment scale (> 500 ha). This study began by comparing P export in overland flow from grazed pasture with areas that receive elevated P inputs and stock traffic (e.g. gateway, water trough, stream crossing and cattle lane). Intact soil blocks were removed in a preliminary investigation of sources areas. Phosphorus loss from the sites was in the order: trough > crossing > gateway > pasture. Total P losses from the trough averaged 4.20 mg P/m^2 while the pasture exported 0.78 mg P/m^2. In addition, runoff from lane soil was measured with total P averaging 5.98 mg P/m^2; however, the method used was different from the other soil blocks. The data suggested that locating and minimizing the size of these source areas in fields has the potential to significantly decrease P loss to surface waters.

To explore the importance of these sources at a greater scale, the soil physical properties and surface runoff from pasture, a laneway and around a watering trough, together with subsurface flows from pasture and catchment discharge, were measured in a small dairy catchment (4.1 ha). Soil measured around the trough and in the laneway was found to be enriched in Olsen P (56 and 201 mg P/kg, respectively) compared with the pasture (24 mg P/kg), as well as having a greater bulk density resulting from more frequent treading by animals. Use of the lane and trough during grazing greatly enhanced dissolved P losses from plots via dung. On a catchment scale, surface transport processes and sources were reflected in stream loads by the dominance of particulate P lost. Sub-surface flow was found to be an important contributor of discharge and likely P losses, which warrants further investigation. The scaling up of runoff plot data suggested that the laneway contributed up to 89% of the dissolved reactive P (DRP) load when surface runoff was likely. This represents a substantial source of P loss on dairy farms. In

addition, the variation of sources and transport processes with season adds another aspect to the critical source area (CSA) concept, and suggests that, given the loss during summer and high algal availability of dissolved P, mitigation strategies should target decreasing dissolved P loss from the laneway.

Spatial and temporal variations in material flow paths, in-stream processes and differing land uses was then measured in a larger catchment using a nested catchment approach. Flow, P and sediment concentrations were measured at 8 sites (5 - 8,000 ha) in the Silver Stream (mixed forest/pastoral) catchment. The hypothesis was that P and sediment concentrations and loads would be a reflection of land use or flow regime, and that this would be a further reflection of the sediment equilibrium P concentration (EPC_0) of suspended or bed sediments. Dissolved reactive P was found to be unrelated to streambed EPC_0 during base flow. The mean DRP concentration under forest (0.006 mg/L) was lower than DRP measured from the grazed catchments (0.09-0.019 mg/L), but similar to the outlet of the catchment, suggesting dilution of P input from the grazed catchment area. During storms, 54-57% of particulate P was exported at the 5-10 ha scale compared with 85-90% at the >300ha scale. The findings of this study suggest that management to decrease P inputs should focus on strategies that target erosion processes and direct stock access to streams, as well as improving grazing management during sensitive times of year.

A Bayesian Network was constructed using results from this study and published data to create a whole farm TP export model. From this network the relative importance of various sources, both point and non-point, could be seen with different management options. Point sources (lanes and stock access to waterways) have the potential to significantly affect TP loads at the farm scale. However, under current practice, loading from diffuse sources is more important than from point sources. Of the various sources, the load from dairy farm effluent was found to be the greatest potential contributor to total diffuse loading.

The key findings of this work were the quantification of the trough and lane as potential source areas, and the effects of these sources with increasing scale. Transport of the particulate fraction of P from diffuse sources was greatest during storm events and found even at the catchment scale. In contrast, the dissolved P fraction from point sources was subject to transformation processes and transport during storm events was only observed at the field or farm scale. One of the remaining questions is what has the greatest influence on P transformations at different scales and with different land uses.

Keywords: phosphorus; sediment; catchment; scale; storm flow; base flow; land use; source areas, dairy, pasture; water quality; overland flow; runoff; season; erosion; equilibrium P concentration.

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Abbreviations

Abbreviations	**Definition**
BN	Bayesian Network
CPI	Conditional probability table
CI	Confidence intervals 95% CI of the mean
CSA	Critical source area
DI	Deionised water
DO	Dissolved oxygen
DRP	Dissolved reactive phosphorus (< 0.45 µm)
DURP	Dissolved unreactive phosphorus (<0.45 µm)
EDI	Effective depth of interaction
EPC_0	Equilibrium phosphorus concentration
H_2O-P	Water extractible phosphorus
K	Hydraulic conductivity
l.s.d.	Least significant difference (5%)
P	Phosphorus
PP	Particulate phosphorus (>0.45 µm)
SEM	Standard error of the mean
SS	Suspended sediment
TDP	Total dissolved phosphorus (< 0.45 µm)
TP	Total phosphorus

Chapter 1 – General introduction

Chapter 1 consists of a New Zealand and overseas literature review of the main sources of P and sediment in grassland catchments. This review also summarizes what is known of the mobilization and transport of P and SS. The second part of the chapter outlines the structure of the thesis, its objectives and related experiments.

1.1 Background

Agriculture is a major contributor to the New Zealand economy. The mild winters and warm summers, coupled with rainfall which is well distributed throughout the year create ideal conditions for pasture growth. The conversion to dairy from sheep farming has been steadily increasing as well as intensification of dairy systems. As an example the average herd size has increased from 139 to 515 in the past 10 years (LIC, 1999; 2009). While this expansion of the dairy industry has brought significant economic benefits, it has also given rise to increasing concerns about the impacts of dairying on the environment and, in particular, on water quality (Hamill & McBride, 2003). It is then important to identify which areas on a farm are the greatest threats to water quality.

In contrast to most dairy farms in the UK, where animals are housed for 180 days and fed silage and concentrates (Haygarth et al., 1998a), animals in New Zealand are grazed outdoors year-round, minimising the need for feed inputs and manure handling. However, this requires special management of animals during the winter. This is the wettest time of the year, when the soil surface is most susceptible to damage by treading, resulting in increased losses of suspended sediment (SS) and phosphorus (P). Average losses from New Zealand pastures grazed by sheep and cattle range from 0.11 – 1.60 kg TP/ha/yr (Gillingham & Thorrold, 2000) with losses from sheep (0.7 kg TP/ha/yr) approximately half that of cattle (1.5 kg TP/ha/yr) or dairy (1.2 kg TP/ha/yr) (Lambert et al., 1985; Wilcock et al., 1999). Moreover sediment losses were more than twice as high for cattle (2743 kg/ha/yr) as for sheep (1107 kg/ha/yr). The associated increase in dissolved reactive P with increasing dairying in Southland (Hamill & McBride 2003) is evidence that dairying is having a greater effect on water quality than traditional sheep and beef farming. It is clear that dairy cattle have a greater potential to generate losses of sediment and P than sheep.

The potential for damage is greatest in areas of heavy livestock traffic. Phosphorus lost from pastures is often the limiting element in freshwater ecosystems and can lead to eutrophication in adjoining waterways (Anderson et al., 2002; McDowell et al., 2009a). Consequently, any additional P input can result in the explosive growth of phytoplankton. Algal blooms can lead to oxygen depletion due to the decomposition of organic material and may also be toxic (Threlfall, 2010). Erosion of soil also increases sediment in streams and waterways which acts as a contaminant by coating the bottom of streams and can hinder the spawning of some fish species. Sediment also creates problems by clogging channels which may lead to flooding.

1.1.1 Forms of phosphorus

Sources of P may be either organic or inorganic. Organic forms of P originate from plant material and microbial biomass while inorganic P is the product of weathering or the mineralization of organic P. Inorganic P is available for plant uptake and is therefore an important fraction of P to evaluate in terms of water quality as it is also available to periphyton (Cotner & Wetzel, 1992). Plants take up P and incorporate it into their biomass until it is returned and recycled through the soil after they die. This organic P then becomes available over time (Turner et al., 2002) depending on hydrolysis and mineralization rates.

As explained by Haygarth and Jarvis (1999), P forms are commonly divided in two ways, either physically or chemically. A filter size of 0.45µm is the limit most often set for particulate (>0.45µm) or dissolved (<0.45µm) P, although P sorbed to colloids < 0.45µm are able to pass through the filter (Haygarth et al., 1997). This physical distinction is important when describing transport processes because particulate P is associated with erosion and dissolved P with the dissolution of P sources into solution. Phosphorus is usually detected colorimetrically following reaction with molydbate, but has limitations such as the hydrolysis of some organic P compounds leading to an overestimation of reactive P. Alternately, in the case of samples with large particulate, the molybdenum blue method may underestimate the total reactive P (Haygarth et al., 1997). In addition dissolved silica (Si) can form complexes with molybdate and interfere with concentration measurements and cause an overestimation of P concentrations by up to 400% (Ciavatta et al., 1990). However, the method is widely used and relatively free of interferences when water samples are handled correctly and have low Si and arsenic concentrations (Shukla et al., 1972). Chemical fractionation of P is important when investigating chemical and physical transformations in transport, but also when evaluating the effects of P loading on surface waters. Reactive P is more readily taken up by algae, resulting in algal blooms and the deterioration of water quality. Unreactive P, sometimes referred to as organic P, may not be readily available as a P source but has the possibility of future availability through mineralization and other in-stream processes (Turner et al., 2002). Although a large proportion of dissolved P (< 0.45µm) is believed to be highly bioavailable, particulate P (Total P – Dissolved P) includes all P fractions >0.45µm, such as P adsorbed to clay, silt and organic colloids, which means that the bioavailability of particulate P is variable and depends on how strongly P is sorbed to particle surfaces.

The form of P also influences how and if P is transported. Organic P, both particulate and dissolved, can make up a large proportion of the total P in the soil solution, soil leachate and in overland flow (Haygarth et al., 1998; Turner & Haygarth, 2000; Toor et al., 2003). Organic P may represent 30 to 65% of the total P in most soils and is primarily present as phosphate esters (e.g. IHP), but also as nucleic acids and phospholipids (Condron & Tiessen, 2005). In the soil, labile organic P rapidly hydrolyzed into inorganic by phosphatase enzymes, while the remaining organic P is only slowly available (Turner, 2005). Soil drying liberates organic P and once the soil is wetted up, dissolved organic P can be found in the first flushes of a storm event (Turner & Haygarth, 2000). It is also during storm events that organic P may be transported from the soil to waterways via rapid transport pathways, such as cracks in the soil and through field drains, before it is hydrolysed. Under these circumstances the organic fraction of P is a direct source of bio-available P. In addition, organic P will undergo continual degradation during transport as was observed by Toor et al. (2003). Some unreactive, or organic forms of P are thought to have greater mobility than reactive forms because they are less strongly sorbed to the soil matrix (Toor et al., 2003). However, Leytem et al. (2002) investigated the sorption of organic P in three different soils and found evidence to the contrary. They found that inositol hexaphosphate (IHP) had greater adsorption than the inorganic P tested (KH_2PO_4) in all three soils, while in two of the soils all forms of organic P tested had greater adsorption than inorganic P.

Inorganic P may be sorbed to soil particles, the most important of which are clays and aluminium and iron oxyhydroxides, or be present as phosphate minerals (Valsami-Jones, 2004). The presence of Fe and Al oxides in soils also affects the mobilization processes of P. For example, Jordan et al. 2005 found that Al-rich soils are less prone to desorb P and concluded that P mobilization was more likely from detachment via erosion than dissolution (Douglas et al., 2007).

1.1.2 Sources of phosphorus and suspended sediment

Sources of P are usually defined in terms of being point, i.e. from a clearly defined point or location, or non-point (diffuse), originating from a variety of sources. The main sources of P found in agricultural catchments are non-point, originating from the soil, from applied fertilisers and from the activities of grazing animals.

Much of the research relating to P in grassland farming systems has been focused on identifying and analysing diffuse sources of P. However, there exist smaller point sources that have received little attention. Some researchers have acknowledged the contribution of certain areas such as lanes, tracks, and water troughs to P transfer (Haygarth & Jarvis, 1999; Bilotta et al., 2007), but few have tried to specifically quantify their importance as P sources. Heathwaite (1997) described them as being major point sources of P export within fields generally assumed to be delivering non-point nutrient export". These point sources are areas of enriched P loading due to alterations in the soil surface caused by the addition of grazing animals. They may also include other impervious areas commonly found in managed grasslands. To date little work has examined their relative influence on P losses at field to catchment scales.

1.1.2.1 Soil phosphorus

The soil can act as a source of P in two ways: 1) P that is present in soils may desorb P to the surrounding soil solution resulting in dissolved P losses and 2) the soil particle itself (with its associated P) may become eroded and carried away, resulting in immediate particulate losses, as well as the possibility of dissolved losses over time.

Olsen P, Mehlich 1 & 3, Bray and Kurtz P-1 are a few of the common methods used to extract plant available P from soils. Soil test P concentrations have been found to play an important role in estimating not only the soils' fertilizer requirements for optimum plant growth, but also the likelihood of P losses from soils (McDowell & Condron, 2004; Gillingham & Gray, 2006). As an example, drainage water was collected via subsurface drains from soils under different fertilizing schemes in the Broadbalk Experiment (Heckrath et al., 1995). The highest concentrations of P in the drainage water came from the plots with the highest Olsen P concentrations. Results also indicated a 'change point' of 60 mg Olsen P/kg below which almost all P was sorbed by the plow layer, and above which DRP concentrations in drainage water increased rapidly. Similar trends were also found by McDowell et al. (2003a) in overland flow from three New Zealand soil types. In these soils the change point of rapidly increasing DRP concentration was between 44 mg and 79 mg Olsen P /kg (Figure 1). The agronomic optimum of Olsen P in these soils for pasture production is close to 25 mg Olsen P/kg, which shows that maximizing soil P efficiency also reduces P losses. Olsen P was also found to be useful in determining both surface and subsurface concentrations of DRP from 44 New Zealand soils (McDowell & Condron, 2004) when combined with the P sorption index (Bache & Williams, 1971), or percent P retention (Saunders, 1964). However the predictive equations found did not include "point" sources like dung patches.

(Figure removed, subject to copyright)

Figure 1 The relationship between dissolved reactive P (DRP) in overland flow and Olsen P for two Pallic soils and the "change point" above which DRP concentrations increase dramatically. Open and closed symbols are soils with superphosphate and manure applied to them, respectively. (Adapted from McDowell et al. 2003a)

1.1.2.2 Fertilizer

Most soils have a great capacity to retain P and common perception is that there is little risk in applying P to soil without loss. However, it has been found that as soils approach P-saturation much of the P applied is lost (Sharpley, 1995). Since much of the New Zealand pasture system is based on clover supplying the bulk of nitrogen required, P inputs have been important to maintain clover populations (Haynes & Williams, 1992). Common fertilizers used for pastures are superphosphate, diammonium phosphate (DAP) and slow release reactive phosphate rock (RPR). Fertilizer inputs can be sorbed to the soil, or lost through overland flow, incidental transfer (i.e. direct loss soon after application) or through the erosion of P sorbed to particles (Haygarth & Jarvis, 1999). In addition, the form of phosphate fertiliser used determines the export of P in overland flow (McDowell et al., 2003b). Nash et al. (2004) found that more dissolved P was lost when DAP was applied to wet soils than when single superphosphate was applied. However, in dry conditions DAP was found to reduce exports compared with single superphosphate. Preedy et al. (2001) measured the incidental transfer resulting from one storm event, of P lost from plots receiving triple superphosphate at a rate of 29 kg P/ha. The load lost was 1.8 kg TP/ha, which is close to the annual load of TP from grazed pastures in that region. In addition, areas that have been repeatedly treaded upon by grazing animals may have little or no vegetation to take up P and poor infiltration capacity. As a result, applied fertiliser may be easily lost through overland flow.

1.1.2.3 Livestock treading

Conventional wisdom is that natural grasslands recycle P efficiently and there is minimal loss from the system (Haynes & Williams, 1992). However, the addition of livestock and fertilizers disrupts the natural P cycle and increases the potential for P loss. Inherent soil properties such as texture, structure and organic matter content also affect the degree of damage caused by treading. Additionally, livestock grazing duration, stocking density and animal type also affect the amount and extent of damage to grassland soils (Nguyen et al., 1998). In one example, land grazed by cattle was found to export almost twice as much P and suspended solids as sheep-grazed land (Lambert et al., 1985). Heathwaite and Johnes (1996) found that heavily grazed plots (>15 stock/ha) exported at least 16 times more TP than ungrazed or lightly grazed grassland plots.

Treading damages both soil and vegetation, but the damage is most severe in areas of heavy traffic such as lanes and tracks, or where animals congregate such as watering troughs and gates. Treading by livestock may lead to increased bulk density, and a decrease in porosity, macroporosity and hydraulic conductivity (Mulholland & Fullen, 1991; Singleton et al., 2000; McDowell et al., 2003c; Bilotta et al., 2007). Areas with treading damage have decreased infiltration capacity, higher sediment loads in overland flow (Nguyen et al., 1998), and ponding of water at the soil surface is more likely to occur (Sheath & Carlson, 1998). However, structural changes in soil properties are usually limited to a depth of 10cm in the soil profile (Mulholland & Fullen, 1991; Singleton et al., 2000).

Areas subject to repeated treading damage often have sparse or absent vegetation due to intense treading or traffic by livestock; for example, deer fence line pacing (McDowell et al., 2004). Without vegetation the soil surface is physically unprotected and lacks below-ground plant material to help keep the soil together and promote infiltration and uptake of P. A North Carolina catchment study (Line et al., 1998) compared the P export from an overgrazed cattle holding area with pasture further up-stream in the catchment. Half of the cattle holding area was bare or impervious, and it contained two unlined ponds for holding runoff and waste water. In addition, cows were allowed unlimited access to the stream resulting in stream bank erosion and direct deposition of faeces in the water. The P export from the cow holding area was 60 times greater than from the pasture during base flow conditions, and 30 times as great during storm events.

1.1.2.4 Livestock excretion

In addition to treading damage, livestock concentrate and redistribute P in excreta, with P loads from cattle faeces ranging between 10-23 g P/cow/day (Betteridge et al., 1986). The amount of P in faeces has been found to be directly related to the P content in pasture (Tate et al., 1991) which increases together with phosphorus fertiliser rates (Rowarth et al., 1988). Therefore, high rates of P fertiliser also indirectly affect P losses from excreta as well. The effects of varying P content in feed have been shown to influence TP in runoff from manure (O'Rourke et al., 2010). In addition, the digestion of pasture by grazing animals can result in a net mineralisation of organic P depending on the digestibility of feed offered (Bromfield & Jones, 1970). The consequences of the variation in pasture digestibility (often high in spring and lower in the summer) (Moller et al., 1996) could then be reflected in the seasonal differences in P leaching from faeces.

(Figure removed, subject to copyright)

Figure 2 The summer distribution of faeces and urine for 36 lactating dairy cows on a 0.74 ha paddock. The concentric circles are spaced at 10 meter intervals (Adapted from White et al., 2001).

The distribution of faeces or dung is also highly variable. Dung deposition on lanes, tracks, water troughs and barnyard areas is high because of frequent animal traffic (Davies-Colley et al., 2004). Animal excreta will also be unequally distributed in the pasture according to areas where cows are more or less likely to rest, e.g. not in steeper areas (Hively et al., 2005). Also, sheep prefer to camp at higher elevations, resulting in increased P deposition on hilltops (Gillingham, 1980). It has also been shown that in warm weather concentrations of faeces and urine are significantly higher within 30 meters of a watering tank (Figure 2) (White et al., 2001) than in the rest of the paddock. The higher concentrations of faeces in specific areas results in greater amounts of P being potentially available for transport. These studies highlight how animal behaviour may significantly contribute to P losses.

Repeated high rates of dung deposition can also change the physical properties of the soil. As previously mentioned, porosity is usually low in areas of high treading intensity. However in one experiment, the greatest porosity (89%) of all sites investigated was an area of heavy usage; the barnyard (Hively et al., 2005). This result was attributed to the high concentration of organic material accumulated on the soil surface from the manure deposited there. The variation in P content and the distribution of faeces on the farm has consequences for the amount, form and location of P lost from grazing systems.

As detailed above, the effects of cattle treading on soil physical properties are well documented, as is the spatial distribution of faeces by grazing stock. However there have been few studies which have combined these two results and identified their potential as source areas of P, and none that have investigated their importance at greater scales.

1.1.3 Mobilization and transport

Phosphorus is mobilized to receiving waters via three main modes of transfer, as outlined by Haygarth and Jarvis (1999): 1) desorption or dissolution of P sorbed to soil 2) erosion of soil particles enriched in P or 3) direct loss from fertilizer or manure if a rainfall event occurs soon after its application or deposition (i.e. incidental). The capacity of soils to retain or fix P is dependent on the number of sites on the soil particle surface which, in turn, is dependent on the clay and organic matter content of the soil. For example the allophonic clays in volcanic Andisols in New Zealand are capable of fixing up to 30 t P/ha (Molloy, 1998).

Sites that are identified as potential P sources will not necessarily contribute to P loading. The amount of P that reaches waterways, and potentially damages water quality, depends in part on hydrology, which is primarily driven by precipitation inputs. Gburek et al. (2000) stated that "Transport factors are what transform potential P sources into actual P losses from a field or watershed".

Phosphorus can be transported to receiving waters either through overland or subsurface flow or a combination of these pathways. Overland flow (or surface runoff) is a thin sheet of water which runs across the soil surface. If vertically infiltrating water reaches an impermeable layer it is converted into lateral subsurface flow. The pathway in which P is transported depends on the soil moisture content, amount of precipitation, infiltration capacity and conductivity of the soil. Impervious surfaces will promote overland flow and the transport of suspended solids and particulate P. On the other hand, dry, compacted soil has a low infiltration capacity, though cracks in the surface may facilitate preferential subsurface flow (Bouma, 1991). Once P is mobilized it follows surface and subsurface hydrological pathways that may result in transformations before it reaches its final destination (Meyer & Likens, 1979).

Transformations of soil P from source to surface waters are regulated in part by the equilibrium between the solid and solution. In this relationship the main factors influencing P release are: the soil P concentration, the rate of release from soil to solution and the time of contact between the soil and solution (Dougherty et al., 2004). When the flow of water through the soil is high (e.g. preferential flow), equilibrium may not be reached. The potential re-adsorption of dissolved P to the soil is affected by the reaction, or residence time, of the solution in the soil pores. However these are not the only contributing factors. In boxes packed with soil, McDowell and Sharpley (2002b) showed that concentrations of P in overland flow from swine manure decreased with increasing flow path length, which was attributed to dilution rather than sorption to soil particles. This highlights the need for further investigation into the exact processes that result in the attenuation of P during transport.

1.1.3.1 Overland flow

Overland flow is a result of either infiltration excess (Hortonian flow) or saturation excess of the soil. Infiltration-excess occurs when the precipitation is greater than the infiltration capabilities of the soil. This type of overland flow is usually localized and is a result of soil surface and shallow soil properties. Saturated overland flow results when the water holding capacity of the soil has reached its limit and can be widespread during the wetter months of the year.

Losses of P in overland flow are related to the concentrations and forms of P in the soil (including any dung or fertilizer derived P) and the volume of overland flow. The depth of the soil surface that contributes to overland flow is 0.1-3 cm (Sharpley, 1985) and is referred to as the effective depth of interaction (EDI). In this thin layer the diffusion of P in the soil water, and dissolution or desorption of P in the solid phase, contribute to P concentrations in overland flow. Light clay soil particles that are enriched in P are preferentially eroded from the EDI at the soil surface. Heathwaite & Dils (2000) measured overland flow on three hillslope positions and found that runoff from the base of the slope had greater concentrations of all P fractions than overland flow measured at the top of the slope. They suggested that this was due to the preferential transport of P attached to eroded soil particles.

The source of P will generally indicate the dominant form of P lost. A study conducted in New York State (Hively et al., 2005) compared the overland flow from 9 different sites on a dairy farm using a rain simulator. Phosphorus concentrations in overland flow from a barnyard were primarily in the form of dissolved P (88% of TP) owing to the large amount of manure deposited there. The cow path in the same study produced much lower levels of dissolved P (< 20% of TP) in overland flow, but instead had higher levels of particulate P (> 80% of TP). This is explained by the site's steep slope, scant ground cover and frequent traffic.

1.1.3.2 Subsurface flow

Subsurface flow has the potential to contribute to the P loading of surface waters, but it is more difficult to quantify and investigate. One subsurface flow pathway, preferential flow, is the gravity driven flow of water through macropores such as earthworm burrows and cracks in the soil etc. An important consequence of preferential flow is that much of the potentially P sorbing soil matrix is avoided. The movement of solution through macropores in unsaturated soil is referred to as bypassing (Bouma, 1991). Dye studies have shown that macropores are the principal transport pathways at depth in the soil column, while at the soil surface the dye evenly infiltrates the soil matrix (Gachter et al., 1998; Jensen et al., 1998; Sinaj et al., 2002). Due to the rapid transport of drainage water via preferential flow pathways, it is possible that the solution is not in equilibrium with the soil matrix, resulting in greater loss of P. A study by Sinaj et al. (2002) examined the concentration of soil P in macropore walls compared with the soil matrix. They found significantly higher concentrations of inorganic P on macropore walls than in the soil matrix, but little difference for organic P. This would imply that inorganic forms of P are more likely to be sorbed to the walls of macropores during preferential flow, while organic forms pass through unaffected. Furthermore, in the same study there was evidence of a complex interaction of old (i.e. pre event) and new (event) water in macropore flow. The concentrations of TP and DRP in preferential flow were low despite the fact that the soil surface was enriched in P. Less than 26% of leachate collected from the lysimeters was new water, and it was reasoned that the rest of the leachate was pre-event water which had had longer contact with the soil matrix, resulting in lower P concentrations.

Other lysimeter studies have found that the proportion of reactive P decreases with depth (Chardon et al., 1997; Haygarth et al., 1998b). This is thought to be evidence that reactive forms of P accumulate in the soil surface, while unreactive P is more mobile and continues down the soil profile. Stratified sampling by Withers et al. (2007) also showed P accumulation in the upper 2.5 cm and proof of vertical P movement down (15 cm) the soil profile. Irrigation may also affect the form of P lost. Toor et al. (2004) found, in a grassland lysimeter study, that particulate forms of P were higher during the irrigated summer months compared with the non irrigated winter. This was attributed to the high irrigation intensity which dislodged soil particles in preferential flow pathways. They also found that unreactive forms of P accounted for 77-91% of TP.

In one of the few studies that looked at the differences in P loads from matrix flow, macropore flow and drain flow (Heathwaite & Dils, 2000) it was found that dissolved P was the main form of P in matrix flow sampled with porous cups. Dissolved P was also the dominant fraction in the drain flow sampled during base flows. However it was found in sampled ground water and macropores flow that the greatest fraction of total P was in the particulate form.

The concentrations of P in subsurface flow are clearly important to the total diffuse loading from grassland environments, but there is little information on how subsurface loads are affected by point sources. Because this is a potential transport pathway for CSAs there should be more work focused here.

1.1.4 In-stream processes

Once P is deposited in a stream, a number of physical, chemical and biological processes influence its form and concentration (Green et al., 1978). Phosphorus is an essential element for growth which is usually in short supply in freshwater ecosystems and therefore quickly used by stream biota. It is possible for P to cycle within a stream between dissolved, adsorbed and particulate forms (Meyer & Likens, 1979), but because transport occurs simultaneously with the cycling, these processes do not act on P in one place. The term "nutrient spiraling" was used to describe the downstream movement of nutrients as they are processed, transformed and recycled (Webster & Patten, 1979).

Sediments and associated P can be removed from stream flow by sedimentation in slow moving areas of water such as pools or dams (Haggard et al., 2001), and then later become re-suspended during periods of high flow. These sediments are derived from local parent material and deposited particulates eroded from surrounding land uses (Kändler & Seidler, 2009). Hence, they reflect the land use and hydrologic processes of the catchment. These sediments have varying abilities to adsorb or desorb P to the surrounding water. The adsorption capacity of bed sediments can be 10 to 20 times greater than the soils from which they were derived (McCallister & Logan, 1978). Meyer (1979) found that silty sediments had a greater capacity to adsorb P than sandy sediments. Additionally, Lottig & Stanley (2007) found that over 50% of the P retention in coarse particles was due to biotic uptake, while physical sorption was solely responsible for P retention in fine particles. This capacity is further influenced by historical P loading. Hill (1982) reported that the highest rates of P removal from Duffin Creek (92%) were downstream of a sewage treatment plant. In contrast, the stretches of stream that had no influence of elevated P inputs from discharge had lower or insignificant removal rates.

However, it is the ability of bed sediment to influence P at base flow that is of most interest because this is when most interaction between sediment and the water column occurs, usually during summer and autumn when, collectively, stable flows and warm conditions promote algal growth. The ability of the sediment to control base flow P concentrations can be estimated by determining the equilibrium phosphorus concentration (EPC_0) and, in turn, whether the sediment acts as a sink or source of P (Klotz, 1988). Some studies have found significant relationships between stream water DRP and the EPC_0 of sediments (Klotz, 1988; Klotz, 1991; McDowell et al., 2003e; Palmer-Felgate et al., 2009). Other studies have found weak (Jarvie et al., 2005; Haggard et al., 2007) or absent (House & Denison, 2002; Smith et al., 2006) relationships between EPC_0 and DRP. Standard analysis of sediments is not used and this may explain the variation in results, however there may be other factors such as flow rate and stream morphology which also determine the importance of sediment sorption and desorption of P.

1.1.5 Temporal variation

1.1.5.1 Flow regime

Different processes dominate during different flow regimes. When there are no point sources present in a catchment, the loss of P and SS during base flow will be minimal. However, if there are continuously discharging point sources such as sewage treatment facilities in the catchment, dissolved P loss may be high even during base flow conditions (Wood et al., 2005; Douglas et al., 2007). Storm events are the focus of many studies because they are known to contribute a large part of the annual P loading to streams. Gburek and Sharpley (1998) found that 70% of the dissolved P load was exported during storm flow and 20% during elevated base flow (i.e. following a storm event). Similarly, in the North Island of New Zealand, mean storm flow concentrations of DRP and TP were 10 times greater than during base flow in a grazed pasture catchment (Cooper & Thomsen, 1988). Kirchner et al. (2000) proposed that peak flows are not simply a result of rainfall or overland flow directly into the stream, but also of a release of pre-storm water from the catchment. The greater export of P and SS during storms can also depend on scale, as found by Haygarth et al. (2005), who showed that at small scales a greater proportion of the annual P load was transported during storm events (Figure 3).

Sampling strategies for determining P and SS loss from catchments must therefore include base flows to detect the presence of any point sources, as well as storm events when the bulk of P and SS is exported.

1.1.5.2 Season

There are also seasonal differences in the transport of P for a variety of reasons, such as the variation in mineralisation rates, the effects of wetting and drying of soils on P sorption and stock management at different times of year. Dils and Heathwaite (1996) found a greater percentage of DRP (87-99% of TP) during storms in early autumn, while later in the year there was more particulate P transported. This pattern was attributed to a flushing effect of stored DRP mineralized over the summer months (Xue et al., 1998). Additionally, in a predominantly dairy catchment, Wilcock et al. (1999) found that the highest DRP concentrations occurred in the spring and autumn due to the use of phosphate fertilizers and the discharge of oxidation ponds to waterways. Furthermore, the majority of measured P losses in overland flow from a cattle-grazed pasture occurred in late winter and spring, especially in the one or two weeks following spring grazing (Smith & Monaghan, 2003).

Hively et al. (2006) observed that 95% of the daily TDP loads originated from impervious areas on a farm during the summer and autumn, compared with a 15% average during the rest of the year. This was thought to be a result of the low rainfall inputs, with overland flow generated mainly from impervious areas and cattle traffic. The TDP loads from impervious areas were smaller during the winter because the cows were kept indoors and precipitation inputs were greater, although it is unclear which factor was more important.

Clearly there is the potential for high concentrations and loads of P from grazed catchments at any time of year depending on the distribution of rainfall and the farming practices employed, but work is required to address which factor dominates, is connected to, or influences others to control P loss.

1.1.6 Connectivity

Investigating the hydrological connectivity between potential source areas and surface waters is not straightforward. A number of approaches have been used but a great deal of uncertainty remains. Most runoff studies are done at the plot scale and it is important to consider if the runoff from these plots is likely to reach waterways at all. One investigation on a hill slope found the development of a local shallow subsurface flow that connected the upland and riparian zones during two to three months in mid winter (Ocampo et al., 2006). This connection depended on the: (i) rainfall regime, (ii) thickness of the unsaturated zone, (iii) soil permeability, and (iv) presence of an impervious layer. A spring located in the slope also proved to be a good indicator of the degree of hydrological connectivity. The greatest discharge from the spring was observed when connectivity was determined to be the strongest.

The relative contribution of overland flow from storm flow vs. base flow also gives an indication of the importance of different sources and pathways. For example, high concentrations of DRP during base flow conditions are likely to be the result of point source discharges (Wood et al., 2005). Transport to streams and surface water depends on the hydrology nearest to the stream. Water table response in a riparian zone up to 40 m from the stream was found to closely mirror runoff patterns (Seibert et al., 2003). However, groundwater measurements made further away from the stream (>40m) showed little correlation with runoff patterns. Similarly, Gburek et al. (2000) found that runoff-producing, hydrologically active areas were within 30m of the channel in a Pennsylvania watershed. There is also evidence that bedrock topography may be more important than the surface topography in controlling the subsurface lateral movement of water down slope (McDonnell, 2003). This may lead to the situation described by Ocampo et al. (2006) where a certain level of precipitation was necessary to activate lateral flow from hillslope to riparian areas.

The implications of these results are that the areas closest to the stream (30-40 m) wet up first and therefore any potential sources of P are likely to contribute to P export. Further away, the connectivity becomes patchy and it is unlikely that sources beyond this range will contribute to stream loads, except in cases of extreme weather. Gburek and Sharpley (1998) hypothesized that the majority of P losses in a catchment originate from areas termed critical source areas (CSA), where sources of P and transport pathways overlap. These areas have both high soil P concentration and large overland flow volumes during storm events. Highly trafficked areas contain greater amounts of available P and are more are susceptible to overland flow and/or leaching of P because of the alteration of vegetation and soil properties due to heavy trafficking by animals and, in some cases, vehicles. When Hively et al. (2006) modelled TDP in an agricultural watershed they found that contributions from impervious areas like barnyards, roadways and cow paths contributed 15% of the total load even though they made up < 2% of the total watershed area. Clearly, it is important to assess these areas further as their relative contribution may not only be disproportionate to their size, but may also change with season and flow regime.

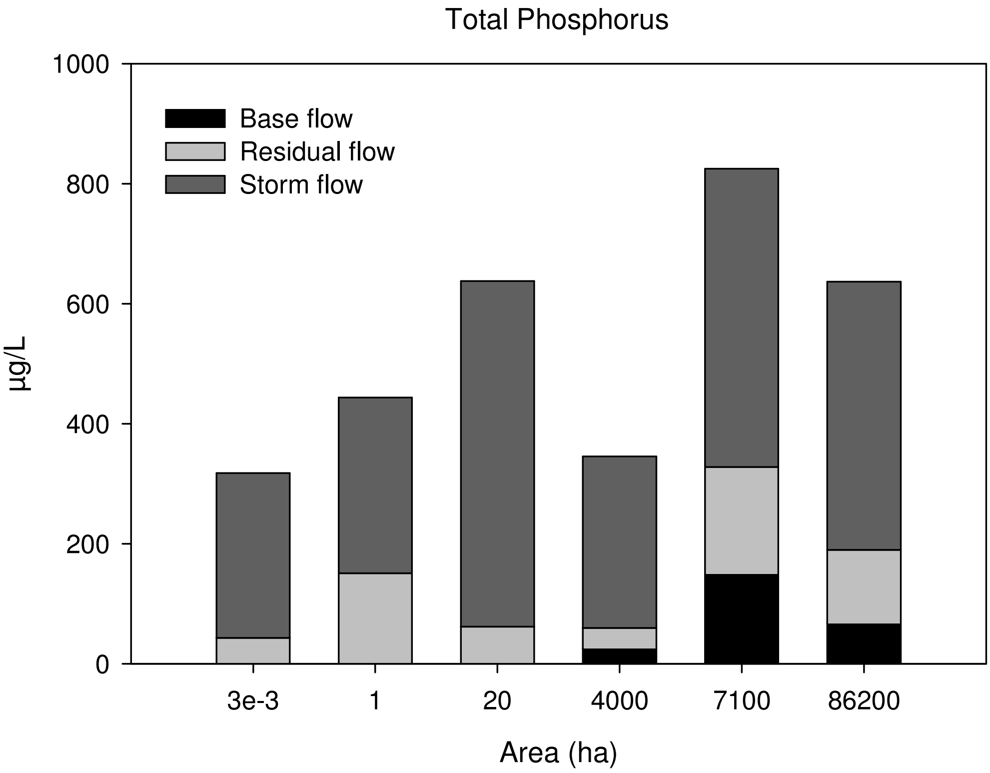

Figure 3 Distribution of total phosphorus concentration from studies at different scales and different flow regimes (Redrawn from Haygarth et al. 2005).

1.1.7 Scale

The transport of P has been researched at many different scales, from plot to catchment. Small scale plot studies are used to examine specific pathways and mechanisms while large catchment studies analyze the effects of flow pathways, land use and best management practices. For landscape and management planning it is important to know how applicable the results from small scales (i.e. plot, field 1 – 2 ha) are at the catchment scale (1 – 2 km^2). There have been a number of nested studies done at different locations attempting to solve this problem and to draw conclusions about nutrient transfer at different scales (Jordan et al., 2005; Wood et al., 2005; Douglas et al., 2007). However, the results vary and a clear picture of how different scales affect the measurement of P transfer is difficult to define. These results highlight how the controls on P transfer are local in nature and vary from one catchment or area to the next.

Catchments are the largest scale studied and may encompass a variety of P sources, land use types, soil types and hydrologically sensitive areas. At this scale, the influence of storm flows and P sources are usually dampened because of increasing inputs and dilution from groundwater (Douglas et al., 2007). As scale increases and the area included in the experiment increases, so do the number of potential sources, hydrological processes and in-stream processes which add to the complexity of the results observed. The catchment is the sum of the processes of the plot, field and slope and can serve as a measurement of connectivity from small to large scales.

Several attempts have been made to compare scales in the same catchment with differing results (Haygarth et al., 2005; Wood et al., 2005; Douglas et al., 2007). Douglas et al. (2007) found no difference in the particulate and dissolved P loading from field (0.15 km^2) to farm scale (0.62 km^2) but did detect a 46% increase in dissolved P loading at the catchment scale. This was unexpected because groundwater inputs should dilute P loads at the landscape scale as per the results from Sharpley and Tunney (2000). Wood et al. (2005) found no difference in the total P export from the plot, lysimeter and hill slope scales, and the DRP was only slightly higher at the landscape scale than smaller scales.

Haygarth et al. (2005) gathered a large amount of data from a variety of scales and flow conditions. At small scales, most P was transported during storm events, while at larger scales base flow gained importance. Forms of P also tended to become dominated by dissolved fractions in base flow and with increasing catchment scale. They concluded that different processes of P transfer operated at different scales.

There are many reasons why measurements at one scale will be different from other scales. As previously discussed, greater scales encompass more land uses and potential sources. The importance of in-stream processing also increases with scale as sediment and associated P becomes alternately entrained and re-suspended and recycled. However, this question of scale is important if we are to take the results of field experiments and relate them to catchment effects.

1.1.8 Methods of investigation

1.1.8.1 Overland flow

Measurements of overland flow are made using several different methods. To investigate overland flow under more controlled conditions, rainfall simulators can be which reduces the variation in rainfall volumes and intensity found under natural conditions. Rainfall simulators may be used over isolated plots (Nguyen et al., 1998; Hively et al., 2005), or intact blocks of soil collected from the field (McDowell et al., 2003d). The problems with soil blocks lie in the cutting and transporting of the soil which may result in cracks or other deformations. The scale of the experiment is also usually small (< 1 m^2). Further, the exposed edges of the soil may contribute additional sediment and P which would lead to an overestimation of loads. This method is adequate for preliminary investigations, but more data is required to ascertain how applicable the results are in the field.

In the field overland flow is generally measured using hydrologically isolated plots (McDowell et al., 2003c; McDowell & Houlbrooke, 2009), or plots with no upper boundaries (Gillingham & Gray, 2006), with a collection gutter collecting the overland flow. Measurement by this method is somewhat limited because it is only a 'snapshot' of the contribution of overland flow in a specific location and does not take into account the full overland flow pathway.

1.1.8.2 Subsurface flow

Subsurface transport can be studied using lysimeters, which collect a sample of soil water. A lysimeter may take the form of an intact soil monolith which is isolated and extracted from the soil, as described by McDowell (2008a), and either reburied on-site or transported to a purpose-built facility in a more convenient location. The inputs (rainfall and/or treatments) are measured and output (leachate) is collected and analyzed. These lysimeters will always be limited by size since, as the size of the intact soil increases, so does the difficulty in excavating and transporting it. Lysimeters of this scale are effective at measuring the attenuation of P with depth (Sinaj et al., 2002) but the transport pathway is mainly vertical, which might not be appropriate in sloping landscapes. Plot lysimeters, as described by Haygarth et al. (1998b) can be much larger and integrate the horizontal and perpendicular movement of water; however they are expensive to install and therefore more appropriate for more long term experiments. Another method to intercept the lateral subsurface flow is to dig a trench perpendicular to the slope as detailed by Wilson et al (1991). As with the plot lysimeter method this requires greater instrumentation to handle the potentially large volumes of subsurface flow it can collect.

Zero-tension samplers, consisting of perforated pipes or wells in bore holes, can also be used to collect a sample of subsurface flow (Simmons & Baker, 1993). On a larger scale, samples are taken from tile drains or seeps/springs to collect information about the load of P from subsurface pathways reaching receiving waters. It is not clear however, to what degree such samples reflect subsurface flow pathways; that is how much of the flow is attributable to matrix flow and how much is macropore flow (Heathwaite & Dils, 2000).

1.1.8.3 Catchment sampling

The accuracy and precision of a catchment monitoring program depends on the sampling regime used. Under ideal circumstances the concentration and the volume of water would be measured continuously so that changes in the loading with time could be clearly seen and investigated. However the expense of such a set up is greater than what the budgets for most experiments can afford. The question then becomes how to achieve the best estimations of load with the fewest number of samples. Johnes (2007) used a large set of daily sampling data to explore differences in sampling frequency and its effect on load estimate uncertainty. They were able to suggest different methods of calculating catchment loads based on the population density, base flow index (proportion of base flow of the total catchment flow) and sampling frequency. In addition Rekolainen et al. (1991) found that load calculation based on sampling concentrated during the high runoff periods had the best precision and accuracy compared with regular sampling intervals.

Rather than sampling only one site in a catchment, nested catchment studies are used to estimate point and non-point sources of P (Grayson et al., 1997; Haygarth et al., 2005). These studies combine the effects of scale, land use, point and non-point sources to understand catchment processes as a whole.

1.2 Objectives and thesis structure

The literature review identified a number of significant gaps in the research surrounding the importance of highly trafficked areas as potential sources of P. Since it is necessary to identify all critical source areas in grazed catchments, investigations were required to confirm or disprove the importance of these potential sources. There was also no clear consensus found on how the scale of measurements affects the relevance of the results. In addition, the connectivity between sources and receiving waters is highly dependent on the hydrology of the site and, although investigated in a number of studies, the implications for transport pathways remain uncertain. Finally, because the temporal variation in loads from sources and in-stream processes is unique to each catchment, this must also be investigated to put the results in context of catchment size and land use.

In light of the above, the goal of this research was to better understand the temporal and spatial losses of P and sediment in dairy and mixed use catchments, by exploring which areas contribute the most to P loading, how this varies with increasing scale, and the nature of connectivity between potential P sources and receiving waters.

The overall hypothesis for this work was that potential source areas in grassland catchments become critical when they are in hydrologically active areas of the catchment and are connected by water flow pathways to receiving waters, but that their importance as critical sources diminishes with increasing scale.

The following specific objectives were formulated in order to reach these goals:

I. Quantify the potential source areas within a dairy farmed catchment (e.g. stock camps, lanes, tracks, water troughs.
II. Identify and quantify the potential pathways of P loss from the source dairy area.
III. Investigate spatial and temporal variations in loads.
IV. Determine connectivity between identified source areas and water flow pathways.
V. Develop rules for assessing connectivity in different landscapes.

To meet these objectives five experiments were conducted (Figure 4):

Potential phosphorus and sediment loads from sources within a dairy farmed catchment. The likelihood for potential source areas to lose P and SS was quantified in a preliminary experiment. The experiment consisted of a rainfall simulation on soils taken from potential source areas like tracks and lanes and compared to pasture. Overland flow was measured from the soils and the soil physical properties were investigated. The results provided information about P concentration in overland flow at the plot scale, and the relative importance of each potential point source area. The results of the overland flow and the soil properties were examined to see which soil factors were statistically correlated with P concentrations in overland flow, and therefore likely to be important at a greater scale. This experiment was one of the methods to meet Objective I.

Phosphorus source areas, pathways and loss from a dairy catchment. Phosphorus and sediment are transported by water and therefore an assessment of the contribution of different flow pathways is important to establishing losses at a greater (catchment) scale. Water flow pathways include overland flow, lateral subsurface flow and percolation to groundwater. Overland flow volume and P and SS concentrations were measured in runoff plots in a dairy catchment in south Otago over a two year period. During this time, subsurface flow was also investigated using zero-tension samplers which collected a sample of the subsurface flow during storm events. The flow rate of the stream leaving the catchment was also monitored and periodic grab samples of water were taken. This experiment was designed to contribute to Objectives I-III.

Effects of scale and in-stream processes on phosphorus and sediment export from the Silver Stream Catchment. Measurements made at one scale do not necessarily represent the sources and transport mechanisms operating at other scales. To investigate the variations in connectivity with scale, the flow rate, P and SS concentrations were sampled longitudinally in a nested catchment study. Samples were taken during different flow regimes and seasons. This experiment was designed to contribute to Objectives III and IV.

Evaluation of base solutions to determine equilibrium phosphorus concentrations (EPC_0) in stream sediments. In-stream processes are important for understanding how the results from P and SS loading from land relate to concentrations and loads measured in streams and rivers. One important measurement of the potential for bed sediments to sorb or desorb P is the equilibrium phosphorus concentration (EPC_0). However, different methods are used in its determination and therefore this experiment looked at what base solution was most relevant to use for streams. The experiment was used to contribute to Objective IV.

Bayesian Network for point and diffuse source phosphorus transfer from dairy pastures. Combining the results from different experiments is problematic because of site and weather variation. Bayesian Networks are able to manage this variation and uncertainty. A network was therefore developed from the literature and from the data gathered in the previous experiments to model the sources and transport processes of P in a dairy catchment typical of the south Otago region. The results from this exercise were used to contribute to Objective V.

The results and conclusions from all of the experiments were used to meet objective V.

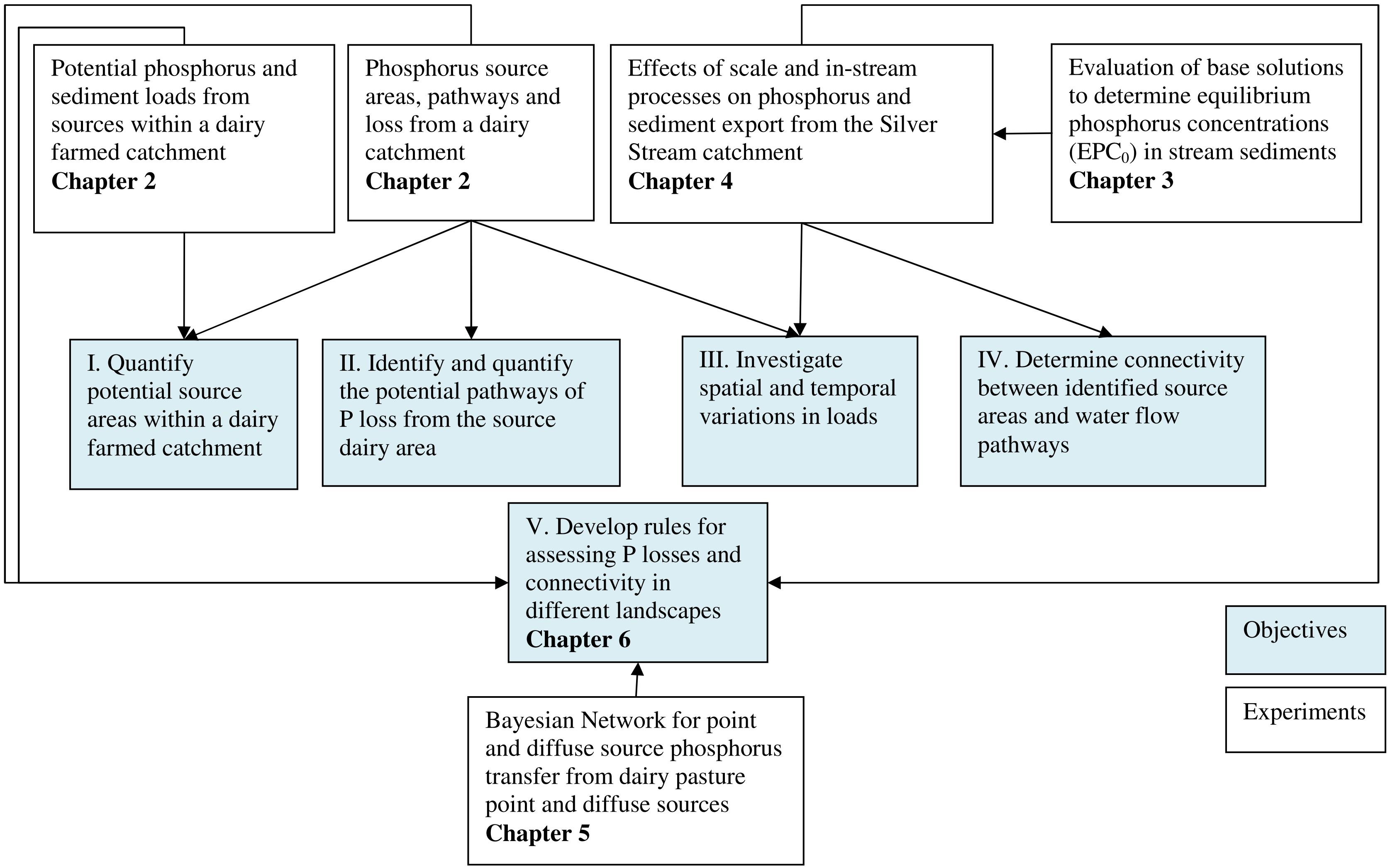

Figure 4 The organisation of the objectives and experiments presented in this thesis.

Chapter 2 - Investigations of critical source areas of phosphorus and sediment in a dairy pasture catchment

This chapter explores the potential source areas and transport pathways of P in dairy catchments at the plot and catchment scale. Few studies have investigated the importance of areas within the paddock scale which are subject to concentrated treading and dung deposition as source areas of P, and none have explored their importance at larger scales. If much of the so called “diffuse” loading from pasture can be localized to discrete sources, then there is the possibility that mitigation can be targeted to these specific areas. This mitigation strategy would be more resource and cost-effective than endeavoring to treat large scale diffuse sources. The experiments described here were designed to meet objectives I, II and III by identifying and quantifying these potential source areas and transport pathways, and investigating the connectivity between sources and receiving waters. This Chapter is divided into four sections: Section 2.1 is an introduction which defines the context of the experiment; Section 2.2 presents the results of a rainfall simulation on soils taken from potential source areas in a dairy farm; Section 2.3 presents the results of two years of monitoring in the same dairy catchment; and conclusions are presented in Section 2.4.

2.2 Introduction

Dairy farming has been found to be one of the principal causes of decreasing water quality in New Zealand’s streams and rivers. In the Otago and Southland regions, the number of dairy cows has increased from 282,900 to over 600,000 during the past 10 years (LIC, 1999; 2009). Increases in P and SS loads are important indicators of declining water quality (McDowell et al., 2009a). In order to effectively gauge the effects of dairying, including future dairy conversions, it is important to accurately identify and describe the sources of potential contaminants from dairy farmed catchments.

Much research has been done to quantify P and SS export from pastoral catchments (Haygarth et al., 1998b; Dils & Heathwaite, 1999; McDowell & Wilcock, 2004). There have also been numerous studies on the effects of treading on pasture soil physical properties (Nguyen et al., 1998; Daniel et al., 2002 ; Tian et al., 2007), but few have directly focused on high use areas within the paddock that may act as sources of P, such as gateways, watering troughs and laneways (Mathews et al., 1994; Dils & Heathwaite, 1996; Hively et al., 2005). Of those studies that have examined high use areas, no connection has been made to water quality. Gburek and Sharpley (1998) hypothesized that the majority of P losses in a catchment originate from areas termed critical source areas (CSA), where the sources of P and transport pathways overlap. Once sources and pathways are identified, mitigation strategies can be more efficiently targeted to CSAs than using costly "blanket" approaches across the whole catchment. The hypothesis is that these high use areas within the paddock may also be CSAs due to the disproportionate time cattle spend there depositing P-rich dung (the source), and intense treading which increases the risk of overland flow (transport pathway).

2.2.1 Experiment aims

Section 2.2 details the initial experiment undertaken to quantify suspected potential source areas. In this study the potential for P and SS losses in overland flow from paddock areas with different intensities of animal traffic, i.e. pasture, a gateway, a water trough, a stream crossing and a cattle lane was examined under wet winter conditions. The aim was to identify potential source areas of P and SS loss from these different areas in a dairy pasture, and to determine a predictive equation to estimate P loss via in this catchment using soil physical and chemical measurements.

Given the sporadic nature of use resulting from rotational grazing by dairy cattle (e.g. the short-term grazing of pastures every 14-50 days) and runoff processes, it was necessary to determine P loads from potential CSAs (e.g. troughs and laneways) over a range of seasons and weather conditions, and relate this to stream P loads. Therefore, the aims of the investigation in Section 2.3 were to: (i) compare the concentrations of P in overland flow collected from a lane and trough with P lost from the pasture and catchment outlet, (ii) investigate the mechanisms involved in runoff generation and the pathways taken, and (iii) gain an understanding of when there is the greatest risk of P loading and why.

2.2.2 Site description

The soils used for the experiment were from a dairy catchment on Pallic soil (New Zealand Soil Classification: Waitahuna silt loam (Hewitt, 1998); USDA Taxonomy: Fragiochrept) at Hillend, Otago (46° 08'S, 169° 45'E). As is common for this soil type, there is a dense fragipan at 20-40 cm depth which limits deep drainage. The catchment is small (4.1 ha) but encompasses an average-sized paddock, representative of that used for grazing in the region. The median annual rainfall (1970-2001) for the area is 1000 mm. Vegetation at this site is a mixed sward of ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*) and the slope ranges from 5-15°. Pastures have received annual maintenance fertilizer inputs of 50 kg superphosphate P/ha and 250 kg lime/ha for the past 15 years, usually in early summer. Nitrogen is added as urea (150 kg/ha) in three applications over the growing season. Stocking density is three cows/ha and paddocks are grazed on a rotational basis every 25-30 days from October to April. There is no grazing in June or July.

The base flow index (proportion of base flow to total flow) for the measurement period was 0.34, which shows that this catchment is very sensitive to storm events and a good candidate for the investigation of CSAs on a small scale.

2.3 Potential phosphorus and sediment loads from sources within a dairy farmed catchment

2.3.1 Materials and methods

Intact soil blocks, 20 cm wide, 80 cm long and 10 cm deep were collected during the summer of 2008 from areas within a pasture that receive increased animal traffic, i.e. water troughs, gateways and stream crossings. Soil was also taken from a cattle lane and packed into boxes. Additional pasture sites were included as a control to represent ambient grazing and treading intensity.

The soil blocks were removed from each site using a cutting blade (except from the lane, see below). The soils were put into wooden boxes of approximately the same dimensions and lined with plastic sheeting. The short end of the box was fitted with a collection gutter inserted just below the soil surface. Any gaps between the soil and box or gutter were filled with petroleum jelly. This was done to ensure that any sediment in overland flow samples was originating from the surface and not washed out from the bottom or sides of the box. Twelve replicates (four replicates at three locations in two adjacent paddocks) were taken from four sites: pasture, crossing, gateway and trough.

Soil from a laneway (lane) was also sampled for this experiment. However, because of the dry and compact nature of the soil it was not possible to remove it with the cutting blade. Instead, the top 1-2 cm of soil were removed and mixed together before being packed into boxes 20 cm wide, 50 cm long and 10 cm deep, with an average bulk density of 1.2 g/cm^3 (estimated as 1.5 g/cm^3 at the field site). Laneways are of anthropogenic origin and are frequently used for moving stock, and therefore the soil is continually being trampled. Thus the use of packed soil boxes from this potential source area is not unreasonable although the method used is different from the intact soil.

2.3.1.1 Rainfall simulation

A rainfall simulation was conducted within the week following excavation of the soil blocks to generate overland flow. The night before the rainfall simulation, the intact soils were saturated and allowed to drain to field capacity. Wetting of the lane soil before simulation caused immediate ponding, therefore the soil was left in its semi-dried state. Artificial rainfall was generated by tap water flowing through a FullJet® (1/4 HH-SS 14WSQ) spray nozzle at a rate of 25-30 mm/h (average recurrence interval (ARI) for Hillend is 15 years; i.e. 15 mm in 1/2 hr). The nozzle was located ca. 2.5 m above the soil surface using the same setup as McDowell and Condron (2004). The boxes were set at 10° (the average slope in the catchment). The time from the start of rainfall until steady overland flow began was recorded and the overland flow was collected every 5 min for a 30 min period for the intact soils. Thereafter the volume of overland flow for each 5 min interval was measured and a subsample taken for P and SS analysis.

The lane soils were subject to the same rainfall intensity, but overland flow was collected every 10 min for a 50 min period. The 10 min interval was used because 5 min was insufficient to collect enough overland flow to analyse. This was because the area of the boxes was smaller (1000 cm^2 vs. 1600 cm^2) and therefore intercepted less rainfall.

2.3.1.2 Overland flow and soil analysis

Due to the high number of samples to be analysed, subsamples of overland flow were frozen before analysis. Unfiltered samples were digested (persulphate) and analysed for total P (TP), while digested and undigested filtered (<0.45 μm) samples represented total dissolved P (TDP) and dissolved reactive P (DRP) respectively. Particulate P (PP) was calculated as the difference between TP and TDP, and dissolved unreactive P (DURP) (or dissolved organic P) as the difference between TDP and DRP. Concentrations were measured colorimetrically using the molybdate-blue reaction (Watanabe & Olsen, 1965) and a spectrophotometer. Suspended sediment concentrations were determined gravimetrically after filtering a known volume of sample through a glass microfibre filter (Whatman GF/F).

After the rainfall simulation, soil in the boxes was analysed for Olsen P and water soluble P (H_2O-P) (Olsen et al., 1954; McDowell & Condron, 2004). Soil was removed from the top 7.5 cm, dried at 40°C, crushed and then sieved (2 mm). Olsen P was determined using a 1:20 soil to solution ratio and a shaking time of 30 min, while H_2O-P used a soil to deionised water ratio of 1:300 and a shaking time of 45 min. The latter test is used as an estimate of overland flow P concentration.

In addition, two undisturbed soil cores were taken from each box to determine hydraulic conductivity (K), and two cores for soil porosity and bulk density. Saturated hydraulic conductivity was measured using a Mariotte vessel to pond water on the soil surface. The soil cores were prepared and analysed according to the method of Drewry and Paton (2000). The cores were sampled from 0-5 cm which has been reported to be more sensitive to grazing effects than depths beneath (Daniel et al., 2002 ; Drewry et al., 2004). Time domain reflectometry (TDR) was used to measure soil moisture before and after the simulation. The saturation percentage was calculated by dividing soil moisture before the simulation by the soil moisture measurement after the simulation when the boxes were completely saturated. The percentage of bare ground not covered by vegetation was visually determined from photographs of the soil boxes taken prior to the rainfall simulation.

After collating the soil physical and overland flow data, regression analysis and analysis of variance calculations were made using GenStat v. 10.2.0.175 (2007) and SigmaPlot v. 10.0 (2006). Where necessary, log or square root transformations were used to account for non-normal distributions and are indicated in the tables.

2.3.2 Results and discussion

2.3.2.1 Soil properties

The soil physical parameters measured were within the same range as on-site measurements made in earlier trials in the same area. In two cattle grazing studies from Hillend the average bulk density measurement for ungrazed pasture was 0.83 g/cm^3, which is lower than the grazed pasture sampled here (McDowell et al., 2003c). Bulk density in the grazed treatments was 1.1 g/cm^3, closer to the average of the crossing and trough soils (McDowell et al., 2005b). Bulk density measured in the boxes was 25% greater in the lane and 10% greater in the crossing area than for the pasture area. However, there was no significant difference in bulk density between the pasture and the gateway which suggests that the sampled gateways were not frequently used. Furthermore, there were 35% more large pores between 30 and 300 μm in the pasture treatment compared with the crossing and trough (Table 1). Greater bulk density and lower hydraulic conductivity result in significant soil structural and hydrological changes in highly trafficked areas compared with the ambient pasture (Willatt & Pullar, 1984; Nguyen et al., 1998). For example, Mulholland and Fullen (1991) found that infiltration rates were 98.5% less in a heavily trampled and poached area compared with the rest of a field.

Saturated hydraulic conductivity (K_{sat}) measurements on pasture soils made after the rainfall simulation were similar to previously published results on ungrazed pastures (McDowell et al., 2003c). Although K_{sat} measurements were variable, the pasture site was significantly greater than the crossing and gateway. Saturated hydraulic conductivity was at least 54% less in the trafficked areas of the pasture (e.g. trough, crossing, gateway) compared with undamaged pasture areas. This was similar to grazed dairy soils in the Waikato (North island, New Zealand) where K_{sat} was 56% less in soils from recently grazed compared to ungrazed pastures (Singleton et al., 2000). Results from grazed pasture trials at Hillend and on similar soil types indicate much lower K_{sat} values (0.6 - 4 cm/hr) than those measured in this experiment (Monaghan et al., 2002; McDowell et al., 2005b) and this is likely to be due to seasonal variation. In the studies by Monaghan et al. (2002) and McDowell et al. (2005b), K_{sat} was sampled in the spring after grazing, whereas soils for this experiment were sampled in the summer. Recovery of soil physical properties (on a similar soil type) has been documented previously by Drewry et al. (2004) over summer and autumn. Saturated hydraulic conductivity in the lane was very low (< 1 cm/hr) and would suggest that most precipitation would be immediately converted to overland flow under saturated conditions. The unsaturated hydraulic conductivity (K_{unsat}) measurements for all soils were much less than K_{sat} (Table 1). For example, the trough soils were able to conduct over 80 times more water in a saturated state than in an unsaturated state (-1 kPa). Therefore, in all sites macropores would have been the main pathways for transport into the subsoil.

The greatest extractable soil P concentrations were found in the lane (99 mg Olsen P/kg) and were significantly greater than the pasture, gateway and crossing (33, 56 and 67 mg Olsen P/kg respectively, Table 1). These concentrations reflect differences in grazing intensity and dung deposition for the different sites (Williams & Haynes, 1990; Mathews et al., 1994). The lane is used daily while the pasture and gateway are used for a couple of days every 30-60 days after every rotation.

Table 1 Mean values for soil physical and chemical properties and their range (in parenthesis) collected from pasture, gate, crossing, trough and lane. Values for K_{sat}, K_{unsat}, Olsen P and H_2O-P were transformed prior to ANOVA. Geometric means are used for K_{sat} values.

Site	Porosity	Bulk density	Pores 30-300µm	Pores <30µm	K_{sat} [1]	K_{unsat} -0.1 kPa [2]	K_{unsat} -1 kPa [2]	Olsen P [2]	H2O-P [1]
	(%)	(g/cm^3)	(%)			(cm/hr)		(mg P/kg soil)	(mg P/L)
Pasture	62.1^{ab}	0.99^{ab}	9.4^{a}	45.6^{ab}	83^{a}	11.4^{a}	1.4^{a}	33^{a}	0.058^{a}
	(56-68)	(0.82-1.15)	(7.5-12.6)	(42-50)	(34-426)	(3.0-21.8)	(2.6-27.8)	(17-50)	(0.02-0.1)
Gate	63.4^{a}	0.95^{a}	8.2^{a}	46.8^{a}	26^{b}	7.1^{b}	0.6^{b}	56^{ab}	0.103^{b}
	(59-72)	(0.72-1.05)	(5.2-10.7)	(42-51)	(1-183)	(0.2-22.0)	(0-22.9)	(28-113)	(0.04-0.2)
Crossing	57.3^{c}	1.11^{c}	6.1^{b}	43.3^{b}	27^{b}	6.7^{b}	0.8^{b}	67^{b}	0.149^{bc}
	(48-69)	(0.80-1.36)	(4.1-9.0)	(38-50)	(4-554)	(0.1-41.0)	(0-33)	(40-111)	(0.08-0.3)
Trough	58.7^{bc}	1.07^{bc}	5.9^{b}	45.3^{ab}	38^{ab}	5.5^{b}	0.5^{b}	76^{bc}	0.157^{bc}
	(50-67)	(0.87-1.28)	(3.9-8.0)	(41-50)	(5-330)	(0.7-12.7)	(0-18.6)	(46-104)	(0.07-0.22)
Lane	52.3^{d}	1.24^{d}	-	-	< 1	-	-	99^{c}	0.201^{c}
	(51-53)	(1.21-1.27)						(55-136)	(0.12-0.29)
	P<0.001	P<0.001	P<0.001	P<0.001	P<0.01	P<0.01	P<0.01	P<0.001	P<0.001

Data in each column with the same letter (a, b c or d) are not significantly different from one another at the given significance level based on a two-tailed t-test

[1]Natural log transformation

[2]Square root transformation.

2.3.2.2 Loss of P and SS in overland flow

The average overland flow volumes are given in Figures 5 and 6. Most of the intact soils reached an almost constant overland flow (equilibrium flow rate) 15 min after overland flow initiation except for the crossing (Figure 5). Initial overland flow can be attributed to infiltration-excess because precipitation > K_{unsat}. However, saturation of the soils was soon reached due to the plastic-lined boxes, resulting in saturation-excess overland flow. The runoff mechanisms on the site at Hillend would function in the same manner because of a restricting fragipan at ca. 20-40 cm depth. This layer restricts the vertical flow of water and would cause saturation from the bottom up.

From the start of overland flow to 30 min later, the 5 min loads (i.e. concentration × volume) of TP and DRP lost from the pasture and gateway sites showed no significant trend with time (Figure 5). The load of TP and DRP lost from the crossing, and to a lesser extent the trough, increased with time, although the variation was high. The presence or absence in trends may be attributed to differences in the sources of P contributing to overland flow. Hively et al. (2005) found that TDP concentrations from sites that had not received fresh P inputs from manure neither increased nor decreased with time after overland flow initiation. Meanwhile, the increase in DRP loads from the trough and crossing may have been a result of the history of much greater dung P application. Many studies have shown that increased manure addition to a soil results in the accumulation of inorganic forms that may translate into increased DRP loss (e.g. Smith et al., 2001b; McDowell et al., 2007).

The lane soil was not wetted before the rainfall simulation and did not duplicate the runoff response of the intact soils used. Since K_{sat} of the lane soil was close to zero, it can be assumed that if the soil was wetted before the simulation almost all applied rainfall would have been converted to overland flow. It has been shown that dry soils are prone to export more sediment than wetted soils (Le Bissonnais & Singer, 1992; McDowell & Sharpley, 2002a). This may result in an overestimation of sediment loss from the lane soil. In this experiment the lane soils exported over four times as much sediment as the trough soil (Table 2).

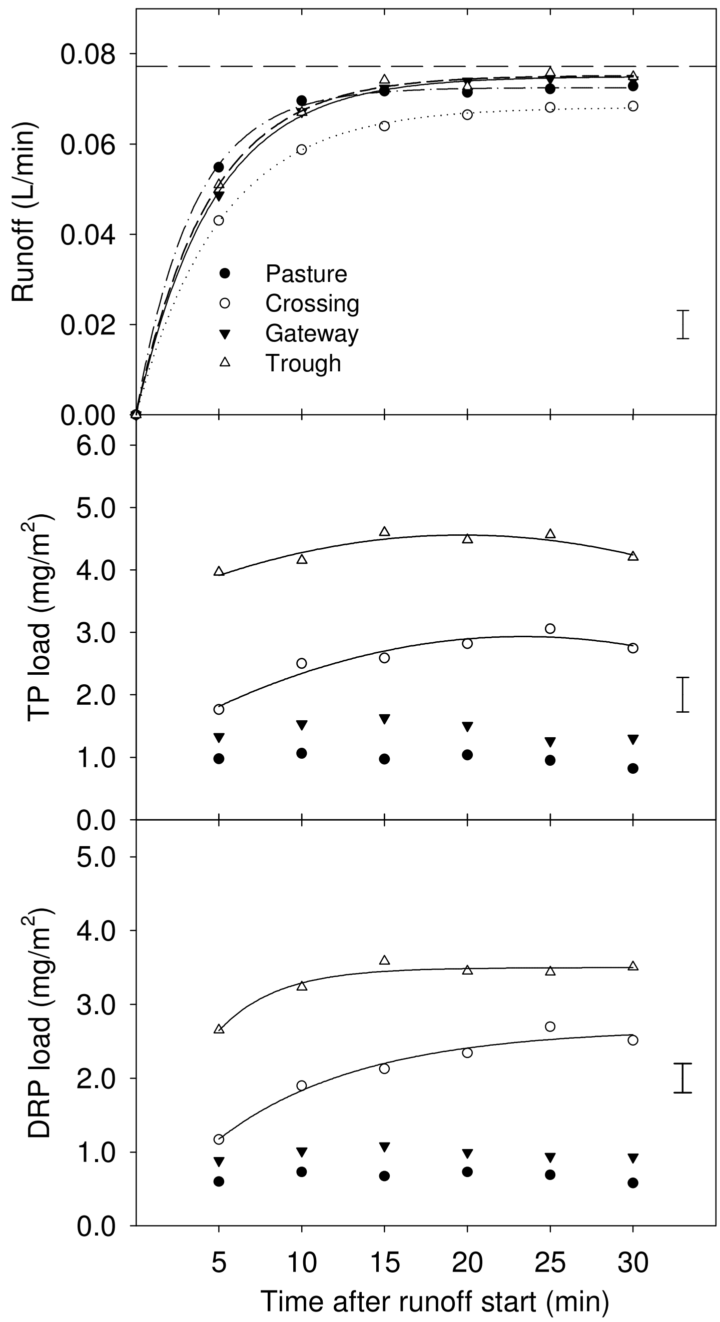

Figure 5 (A) Overland flow measured with time in each treatment (pasture, crossing, gateway and trough), rainfall intensity is indicated by the long dashed line. The mean total phosphorus (B) and dissolved reactive phosphorus (C) loads with time from the start of overland flow. Error bars indicate LSD (5%) between treatments.

Loads of TP and DRP increased with time. However, DRP comprised less of the TP with time and instead PP made an increasingly greater contribution. The packed lane soil had not reached a constant overland flow rate by the end of the simulation (Figure 6) and hence these data may underestimate the load lost if these soils show the same trend as the trough and crossing soils, i.e., that loads increase with increasing flow rate until equilibrium is reached (Figure 5). It could also be reasoned that if the soils were first wetted there would be little infiltration of rain into the soil and only the P present on the soil surface would be available for desorption into solution. This could mean that the load measured here is too high although the enriched Olsen-P and H_2O-P concentrations of the lane soil would imply elevated P export (Pote et al., 1996; McDowell & Condron, 2004).

Table 2 Mean loads of phosphorus fractions and sediment in overland flow collected during the final five min interval (30 min after runoff start). All values were transformed prior to ANOVA.

Site	DRP [2]	PP [1]	TP [1]	SS [1]
		(mg/m^2)		
Pasture	0.48^{a}	0.235^{a}	0.78^{a}	73.14^{a}
	(0.04-1.27)	(0-0.56)	(0.34-1.4)	(10-225)
Gate	0.83^{a}	0.375^{a}	1.31^{ab}	147.9^{b}
	(0.21-2.34)	(0.07-1.16)	(0.59-2.51)	(49-786)
Crossing	2.25^{b}	0.239^{a}	2.60^{bc}	103.9^{b}
	(0.3-5.3)	(0-1.2)	(0.7-6.1)	(41-159)
Trough	2.78^{b}	0.694^{a}	4.20^{cd}	190.2^{b}
	(1.06-4.50)	(0.03-2.5)	(1.5-9.5)	(39-707)
Lane	3.20^{b}	2.029^{b}	5.98^{d}	875.6^{c}
	(1.2-6)	(0.56-3.83)	(3.4-11.1)	(425-1386)
	P<0.001	P<0.001	P<0.001	P<0.01

Data in each column with the same letter (a, b, c or d) are not significantly different from one another at the given significance level based on two-tailed t-test

[1]Natural log transformation

[2]Square root transformation.

Nevertheless, using the described methods the total P load was greatest from the lane (6 mg/m^2) followed by trough (4 mg/m^2), crossing and gateway, while the pasture site had the lowest P loading (< 1 mg/m^2) of all the sites (Table 2). The DRP load was similar for the crossing, trough and lane although there was a great difference in TP (Figure 7). The trough and lane had greater PP and DURP in overland flow. Increased particulate P may be attributed to a high percentage of bare ground (lane 100%, trough 82%) due to frequent animal traffic and treading damage. In a similar experiment with simulated rainfall and intact soil blocks, PP made up 32% of TP in the treading treatment compared with 24% without treading (McDowell et al., 2007). In the same study McDowell et al. (2007) found that 10% of P export originated from fertilizer, 30% from dung, 20% from pasture plants and 40% from soil. In areas such as the trough and lane the plant input would be absent, but the soil and dung components would make up a much larger part. In all of the sites, DRP was the dominant P fraction which is consistent with the findings of Haygarth et al. (1998b) and McDowell et al. (2007).

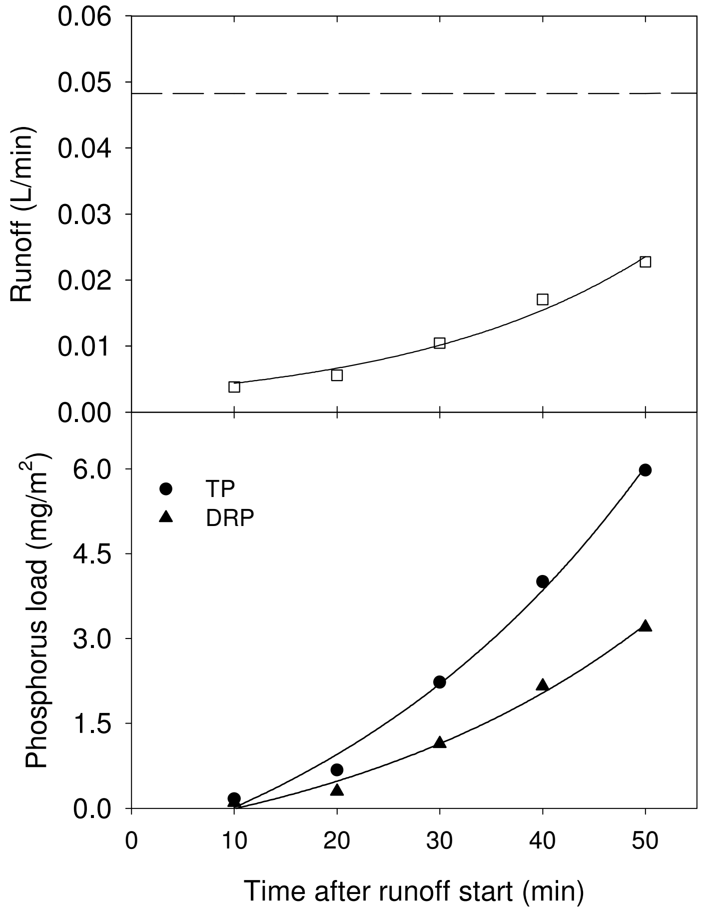

Figure 6 Overland flow measured with time in the lane (A) and mean total phosphorus and dissolved reactive phosphorus loads with time (B). Rainfall intensity is indicated by the dashed line.

2.3.2.3 Predicting P loss for sites

To further investigate the relationship between P loads in overland flow and soil variables, stepwise linear regressions were performed to discover the factors accounting for most of the variation in P loss. The following parameters were used for the regression: Olsen P, H_2O-P, days after treading, % bare ground, % saturation, the time to runoff, bulk density, % pores 30-300µm, % pores < 30µm (micropores), K_{sat}, and K_{unsat} at 0.1 and 1 kPa. Regressions were performed for TP and DRP individually, as the dominant fractions from each site, and with combined data from all sites (Table 3). The trough and the gateway regressions, accounted for over 90% of the variance in both TP and DRP loads, while TP regressions from the crossing and lane were not significant. The relevance of the parameters is explained below together with a discussion of some of the relationships that were not as expected.

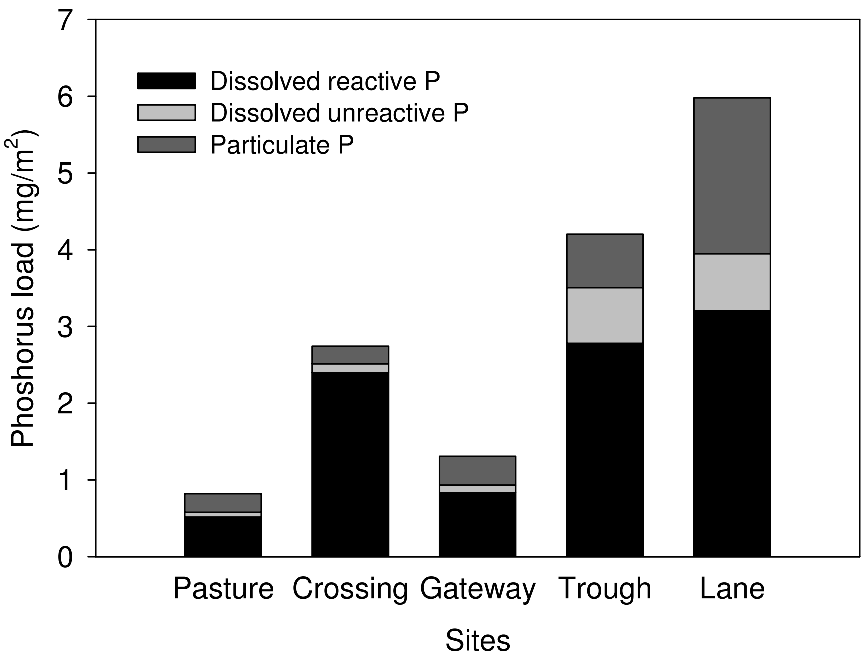

Figure 7 P load fractions from pasture, crossing, gateway, trough and lane at the final sampling of overland flow.

Soil test P concentrations have been found to play an important role in estimating soil fertilizer requirements for optimum plant growth, but also the likelihood of P losses from soils. Two methods of extraction were used here: Olsen P because of its standard use in many parts of the world (Tunney et al., 1997), and H_2O-P which has shown a positive relationship with DRP in overland flow (McDowell & Condron, 2004). H_2O-P was a significant (P < 0.001) variable in both TP and DRP loading from the combined data from all sites (Table 3). It was also a significant variable for estimating DRP load from the trough, however the coefficient was negative. This negative relationship would imply lower P loading from soils with greater pools of available P which is counterintuitive. This inverse relationship can be explained by examining Figure 8 which shows DRP plotted against H_2O-P for all sites. When all sites are included there is an increasing trend, but for individual sites it appears that the range of H_2O-values is not wide enough to outweigh sample variation. The diagram for Olsen P and TP loads is similar to Figure 8 and helps to explain the negative relationship between Olsen P and TP (Table 3). Olsen P was a significant (P < 0.05) variable in the TP and DRP load regression from the pasture. The gateway was the only site where neither P soil test method was significant for P loading, yet the regression was able to account for 94% and 93% of the variance in TP and DRP respectively.

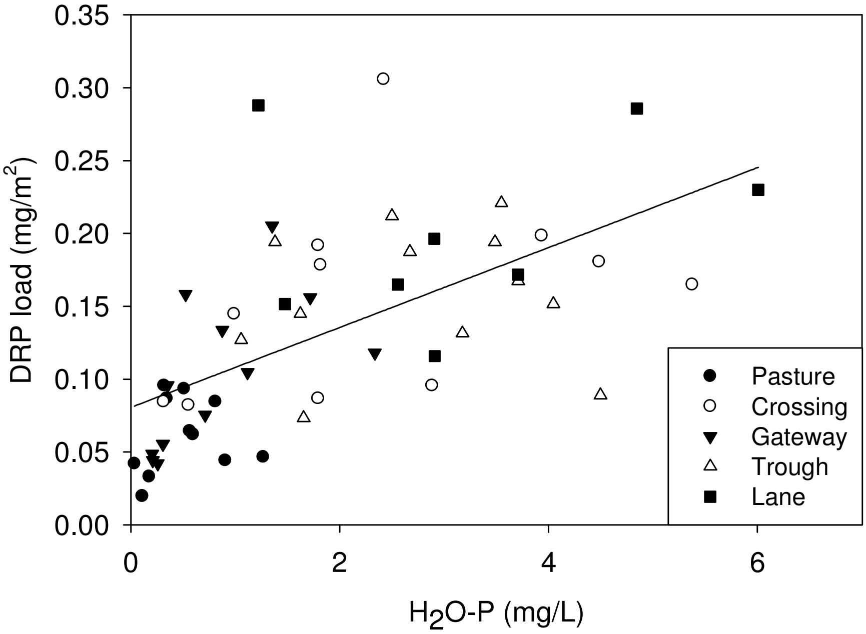

Figure 8 Plot of dissolved reactive P load versus H_2O-P concentration for all sites investigated. Line represents linear regression for all sites y = 0.0274x + 0.0806 (R^2 = 0.38; P < 0.001)

The number of days since treading is an indirect measure of soil recovery. With time since grazing damage, infiltration rates and macroporosity increase, although different studies have reported that this recovery process can take from months to years (Nguyen et al., 1998; Drewry et al., 2004; McDowell et al., 2007). Additionally, it has been shown that the greatest potential for P loss is during or soon after grazing (McDowell et al., 2006). Three different lengths of time since grazing were used in this experiment and this variable was only significant in the gateway for DRP loading.

The percentage of bare ground was significantly related to TP and DRP loss in the gateway, trough and all sites combined. When vegetation is present the soil surface above ground is physically protected from the impact of rain, while below ground roots and organic material facilitate soil aggregation, promote infiltration and the uptake of nutrients such as P (Brady & Weil, 2002). The coefficient was positive for the gateway and all sites, but negative for the trough area (Table 3). Sharpley (1985) found that the effective depth of interaction (EDI) between surface soil and overland flow decreased by 60-80% when a 0.5 mm^2 mesh was introduced to simulate vegetative cover. This would suggest a positive relationship with increasing bare ground and P loss, especially where the bare ground is enriched in P. In addition, Seeger (2007) investigated overland flow from a variety of sites in Spain and found there was a negative correlation between vegetation cover and both overland flow volume and sediment loss. However, there was evidence of a vegetation threshold above which there was a dramatic decrease in overland flow volume and sediment loss. This threshold seemed to vary according to plant type and composition which suggests a more complex relationship between vegetation cover and infiltration. It is not clear why trough soils, with less vegetation, would export less P. It is possible that the soils are more compact (i.e. greater bulk density) or subject to slaking and subsequent surface sealing. This might effectively decrease the interaction of the soil with overland flow, resulting in less TDP exported.

Time to overland flow was found to be an important parameter in other rainfall simulations (McDowell et al., 2003d; Hively et al., 2005; McDowell et al., 2007), but here it was only significant in the gateway and the lane. All soils (except the lane) were at or near field capacity before the simulation and, in most cases, the time to overland flow was < 5 minutes. The gateway soils were quick to produce overland flow and exported more TP than other sites. However, the opposite was true in the lane. This is probably because the lane soils were not wetted before the simulation, and the longer time to runoff may represent a more thorough wetting of the soil and greater desorption of DRP.

Total P export from the pasture, trough and all sites combined was negatively related to bulk density (and therefore positively to total porosity). Therefore, soil with greater total pore space facilitated greater contact with water and desorption of P. However, bulk density was positively related to DRP loads in the lane, probably because the nature of the lane soil was different from the other soils used. Higher bulk density here meant infiltration-excess conditions, a shorter time to runoff and thus the possibility for greater DRP export. The percentage saturation of the soil before the rainfall simulation was positively related to TP and DRP in the pasture and trough regressions, but not in all sites. Greater saturation before the simulation means more "old" water (i.e. longer residence time) in contact with the soil, resulting in an enriched P concentration (provided inflowing water contained little P) (Kirchner, 2003).

The loads measured in this experiment were meant to resemble winter or spring conditions where the soil is near field capacity and receives a moderately-sized rainfall event. In small events the contribution from overland flow would be minimal, with a greater contribution from subsurface flow. The real risk lies in larger events where saturated areas expand and could connect source areas with nearby streams. However, source areas like laneways, which have little infiltration and could produce overland flow even in small events, are commonly bounded by drainage ditches which serve as a conduit for P-rich overland flow to receiving waters. Additional research is needed on site to assess P export in different sized events from different source areas, while monitoring is also needed to determine the proportion of P from these areas at the catchment scale.

Table 3 Results from stepwise linear regression analysis for total P (top) and dissolved reactive P (bottom). Coef. is the coefficient for the regression variable and F pr. is to the probability value assuming an F distribution. [a]Statistical significance is denoted by * = P<0.05; ** = P<0.01; *** = P<0.001; n.s. = P > 0.05.

TP (mg/m^2)	Pasture		Gateway		Crossing		Trough		Lane		*All sites*	
	Coef.	F pr.	Coef.	F pr.	Coef.	F pr.	Coef.	F pr.	Coef.	F pr.	Coef.	F pr.
Olsen P (mg P/kg)	0.102	0.004	–	–	–	–	-0.0423	0.002	0.304	0.018	–	–
H_2O-P (mg P/L)	–	–	–	–	-20.1	0.115	–	–	-125.6	0.021	13.78	<.001
Days after treading	–	–	–	–	-0.817	0.045	–	–	–	–	–	–
Bare ground (%)	–	–	1.495	<.001	-10.83	0.060	-12.04	<.001	–	–	2.592	<.001
Saturation (%)	15.06	0.005	–	–	-23.46	0.037	17.95	<.001	–	–	–	–
Time to runoff (min)	–	–	-0.149	<.001	–	–	–	–	–	–	–	–
Bulk density (g/cm^3)	-8.85	0.026	–	–	–	–	-20.16	<.001	60.6	0.176	-5.08	0.049
θ_v -1 kPa/pores 30-300 (%)	-30.04	0.031	-16.89	0.006	67.0	0.132	–	–	–		–	–
θ_v -10 kPa/pores < 30 (%)	-24.21	0.031	–	–	–	–	–	–	–	–	–	–
K_{sat} (mm/hr)	–	–	–	–	–	–	–	–	–	–	–	–
K_{unsat} 0.1 kPa (mm/hr)	0.009	0.012	–	–	–	–	–	–	–	–	–	–
K_{unsat} 1 kPa (mm/hr)	–	–	–	–	-0.345	0.037	–	–	–	–	–	–
Constant	4.31	0.312	9.56	0.003	28.9	0.103	23.81	<.001	-90.3	0.126	4.95	0.049
r^2	.88*[a]		.94***		.68 n.s.		.96***		.77 n.s.		*.45****	

DRP (mg/m^2)	Pasture		Gateway		Crossing		Trough		Lane		*All sites*	
	Coef.	F pr.	Coef.	F pr.	Coef	F pr.	Coef.	F pr.	Coef.	F pr.	Coef.	F pr.
Olsen P (mg P/kg)	0.0461	0.013	–	–	0.066	0.047	–	–	0.174	0.004	–	–
H_2O-P (mg P/L)	–	–	–	–	-33.23	0.024	-29.31	<.001	-67.40	0.005	10.47	<.001
Days after treading	–	–	-0.182	0.002	-0.624	0.050	–	–	–	–	–	–
Bare ground (%)	–	–	2.311	<.001	-13.46	0.013	-11.53	<.001	–	–	1.261	0.013
Saturation (%)	7.14	0.004	–		-16.44	0.063	9.63	0.001	–	–	–	–
Time to runoff (min)	–	–	–	–	–	–	–	–	0.581	0.010	–	–
Bulk density (g/cm^3)	–	–	–	–	–	–	–	–	69.4	0.008	–	–
θ_v -1 kPa/pores 30-300 (%)	–	–	–		–	–	60.0	0.007	–	–	–	–
θ_v -10 kPa/pores < 30 (%)	–	–	20.93	0.013	–	–	-54.14	<.001	–	–	–	–
K_{sat} (mm/hr)	-0.0004	0.158	–	–	–	–	–	–	–	–	–	–
K_{unsat} 0.1 kPa (mm/hr)	0.005	0.099	–	–	-0.010	0.117	–	–	–	–	–	–
K_{unsat} 1 kPa (mm/hr)	–	–	–	–	-0.412	0.010	–	–	–	–	–	–
Constant	-7.73	0.004	-7.22	25.50	32.77	0.028	29.72	<.001	-95.4	0.006	-0.053	0.878
r^2	.74*		.93***		.85*		.93***		.93*		*.42****	

2.4 Phosphorus source areas, pathways and loss from a dairy catchment

2.4.1 Introduction

The results of the previous experiment proved that loads are potentially higher from the trough and lane, than the pasture. The limited scale and controlled conditions of this experiment meant that further investigation was needed to understand more specifically under what conditions these potential sources contribute to P and SS loads, and the pathways that are used to connect the sources with receiving waters. To do this, the overland flow from the pasture, trough and lane were measured for two years in the same location where the soil blocks were taken in the first experiment. In addition wells were installed to collect a sample of the subsurface flow. The stream draining the catchment was also monitored for two years, both for discharge and P and SS concentrations during base and storm flows.

2.4.2 Materials and methods

2.4.2.1 Overland flow plots

Overland flow was measured using plots isolated on all sides with either wooden boards or metal frames. Metal frames were used on the laneway and near the trough because they needed to withstand repeated treading. The plots measured 50 cm wide and 100 cm long and were arranged in groups of three; one group on the lane, one group just down slope of a watering trough (3 m from trough) and three groups on surrounding pasture sites at different hillslope positions (Figure 9). The size of the plots was restricted due to their placement near the watering trough and on the laneway. The pasture plots were made to the same specifications in order to remove the potential effects of different flow path lengths on overland flow concentration (Le Bissonnais et al., 1998; Sharpley & Kleinman, 2003).

The overland flow from each plot was collected, via a hose connected to the frame, into a 20 litre bucket. There were 17 events where at least one of the three collection buckets was filled to capacity and on occasion overland flow greater than 20 L was lost. Since it would be impossible to estimate the excess overland flow reliably, it was decided to use 20 L as the overland flow volume for the event. A manual rain gauge was used at the site to measure rainfall between site visits. There were seven occasions when the manual rain gauge failed and, in its place, NIWA’s Virtual Climate Station (VCS) data (interpolated weather data) was used (Tait et al., 2006).

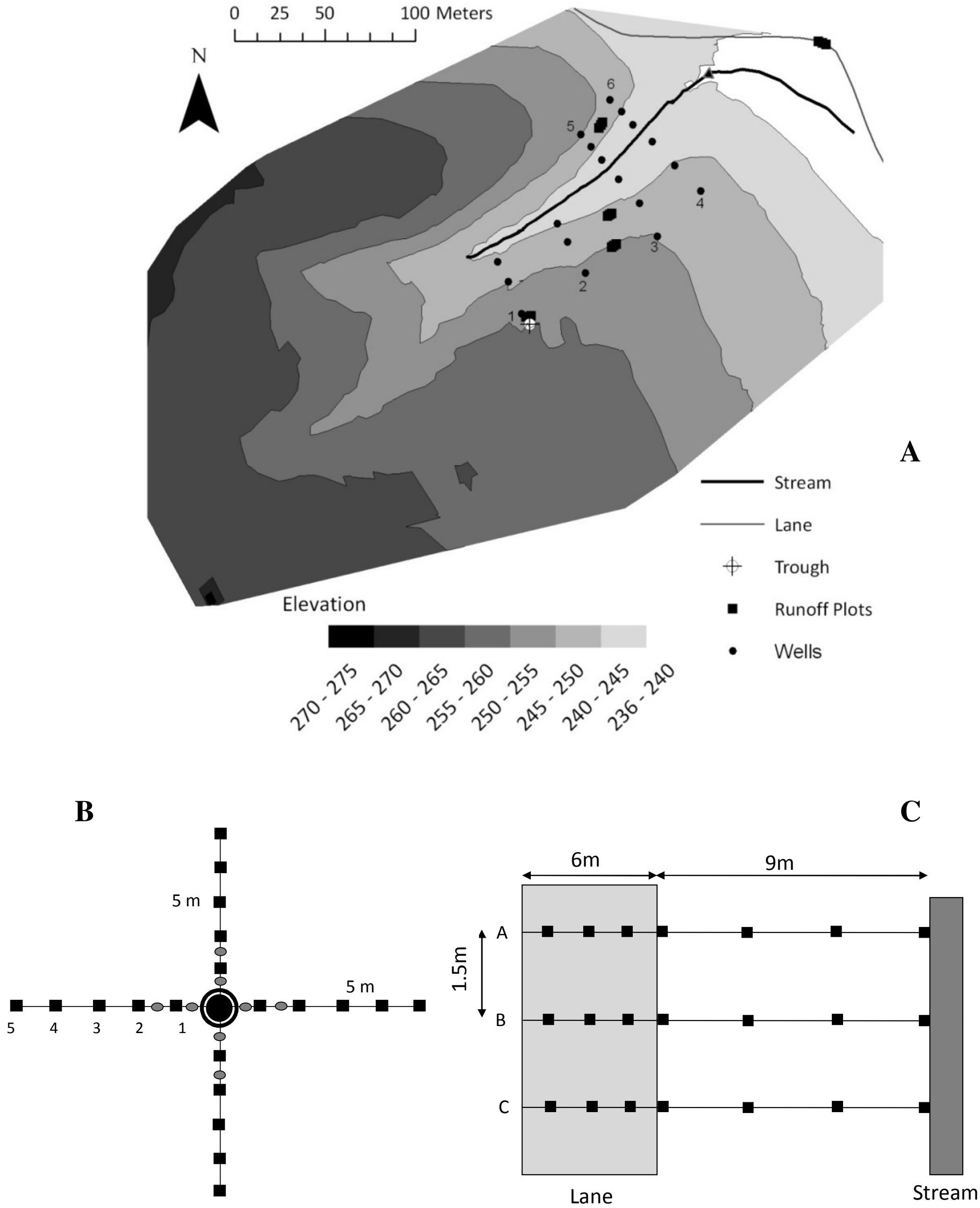

Figure 9 A: Map of well transects (1-6) and location of runoff plots in the Hillend catchment. B: the soil sampling strategy for the trough with black squares and grey circles representing the location the soil samples taken in relation to the trough in the centre. C: Three transects were measured across the lane and to the edge of the stream with black squares representing sampling points.

Runoff events can happen at any time during the year. Therefore, in addition to rainfall, soil moisture was also used to explain some of the variation in runoff, P and sediment loads. Soil moisture measurements from a nearby meteorological station (17 km away) were used as an estimate of soil moisture in the catchment. These soil moisture levels served as a general indication of wet periods vs. dry and transition periods when the soil was either wetting up or drying out. The year was also divided into four seasons to see if changes in temperature, pasture growth and grazing operations had any influence on P and SS concentrations or loads.

In order to measure the contribution of subsurface flow to P loss, 18 wells were installed at 6 transects to intercept subsurface flow, and placed at three different hillslope positions; top, mid-slope and toe. The top of the slope was approximately 30 m from the stream, the mid-slope was 20 m and the toe was 5 m. Transects on the opposing slope (5 & 6) were spaced closer together because the slope was steeper, but were at approximately the same elevation as the opposing slope (Figure 9). The wells were similar to the zero-tension samplers described by Simmons and Baker (1993) and were made of PVC pipe approximately 5 cm in diameter. Ten centimeters of the length of the well was perforated and then covered in a fine mesh to keep fine soil particles from contaminating the sample. This perforated inlet was positioned just above the fragipan at 20-40 cm.

2.4.2.2 Soil sampling strategy

In the summer of 2009, soil samples were taken to characterize soil physical (bulk density and saturated hydraulic conductivity [K_{sat}] using the methods of Drewry and Paton (2000)) and chemical properties at a number of sites: pasture used for dairy grazing, a watering trough in the pasture, and a laneway adjacent to the pasture.

For the pasture, bulk density measurements were collected from four transects (well transects #3-6) running perpendicular to the contour; two transects each on opposing slopes in the catchment (Figure 9), and at three depths: 0-5 cm, 10-15 cm and 20-25 cm. Saturated hydraulic conductivity (K_{sat}) was measured at the same positions, but only for the surface layer (0-5 cm). Bulk density and K_{sat} measurements were also made from four transects radiating from the trough (Figure 9). Duplicate samples were taken every meter from the trough edge up to and including 5 m away.

A different sampling method was used for the lane due to the difficulty of removing the compact surface material in summer, and of getting an intact sample in winter when the surface is covered with patches of soft muck. To measure bulk density, an area of the lane was dug out to the depth of 2-3 cm and the soil reserved for drying and weighing. The space was filled with clover seed, which is small (oblong, 2-3 mm) and pours easily, and the volume of seed used was measured. The mass of the dry soil from the hole was divided by the volume of the seed used to fill the hole and an estimate of the lane bulk density calculated (1.5 g/cm^3). To determine K_{sat}, soil was taken from the surface of the lane and packed into stainless steel rings. Saturated hydraulic conductivity measured this way was 0.01 (±0.05) cm/h. However, the bulk density of the soil packed into the rings was only 0.9 g/cm^3 and therefore the actual K_{sat} of the lane is likely to be less than that measured.

Olsen P (Olsen et al., 1954) and water-extractable P (H_2O-P) (McDowell & Condron, 2004) were determined as measures of plant available P and as an estimator of P in overland flow, respectively. Samples were taken on two occasions. The first was in the winter of 2009 when samples of topsoil (0-5 cm) were collected from all of the runoff plots. The other occasion was in summer, when pasture soil was sampled at three depths (0-10 cm, 10-20 cm and 20-30 cm) at all 6 transects and slope positions (Figure 9). Intensive sampling was undertaken within or near to the trough and lane, and at distances along transects away from the trough and down slope of the lane towards a receiving stream (Figure 9). The intensive sampling around the lane and trough was from the 0-5 cm depth. However, as mentioned previously, it was difficult to sample to that depth on the lane. The lane material was sampled down to 2-3 cm which, because of stratification, would lead to an overstated soil test P concentration. However, because of the low permeability of the lane, the top 2-3 cm would be representative of the depth of interaction with overland flow.

2.4.2.3 Chemical analysis

Phosphorus in overland flow samples was measured colorimetrically using the method of Watanabe and Olsen (1965). Total P (TP) in samples was determined by persulfate digestion. Total dissolved P (TDP) was determined by first filtering the solution (< 0.45 μm) before the filtrate was persulfate digested. Dissolved reactive P (DRP) was simply filtered (< 0.45 μm). Dissolved unreactive P (DURP), sometimes referred to as dissolved organic P, was calculated as the difference between TDP and DRP, while particulate P (PP) was calculated as the difference between the dissolved fraction (TDP) and TP. Suspended sediment was determined by filtering a known volume of stream or overland flow sample through a 0.7 μm glass fibre filter paper, drying the sediment on the paper and weighing the residue.

2.4.2.4 Stream sampling and load estimation

At the catchment outlet, a 29 cm high H-flume was installed capable of measuring flows up to 55 L/s. This was linked via a stilling well to a datalogger that recorded flow (L/s) every 20 minutes which was averaged for each day (Q_{di}). Flow-stratified grab sampling of the stream occurred in drier months (Oct-April) every three to four weeks, and for the remainder of the year every one to two weeks. A total of 68 samples were collected from 1/5/2008 – 30/4/2010. This was augmented by an auto-sampler which took a composite sample of 10 out of the 15 storm events.

To estimate load, daily concentrations of P fractions and SS were derived by linear interpolation between sampling points (Rekolainen *et al.* 1991 Kronvang & Bruhn, 1996). The load was calculated as per Method 1 used by Rekolainen et al. (1991) or Method 2 used by Johnes (2007):

$$\text{Load} = K \sum_{i=1}^{n} (C_{di} Q_{di})$$

where; C_{di} is the concentration of sample (mg/L), Q_{di} is daily mean flow (L/s); K is a conversion factor to convert mg/s to kg/d (0.0864); and n is the number of days in the relevant measurement period. Flow was also separated into base flow and storm flow using BLFOW (also used in the Soil and Water Assessment Tool; SWAT (Arnold & Allen, 1999)) from which the base flow index was calculated. This automated method was used because of its reproducibility and because it removes the subjective nature of manual separation.

Where appropriate, analysis of variance and regression analysis were assessed using GenStat (11^{th} Edition, 2008). Standard error of the mean (SEM) and least significant differences (l.s.d.) were also calculated using the same program. If the differences in treatment means are greater than the given l.s.d. the differences are significant at the 5% level, unless otherwise stated. Significant relationships described in the text are at a $P < 0.05$ level unless otherwise stated. Where confidence intervals (CI) are given (mean ± CI), they refer to the 95% CI of the mean.

2.4.3 Results and discussion

2.4.3.1 Soil sampling

The overall means of soil physical measurements for pasture sites are presented in Table 4. There were no significant differences between transects for K_{sat}, bulk density or soil test P. This indicated that differences in transect or slope position did not significantly affect these soil properties. Olsen P decreased with depth (Figure 10), similar to the findings of Haygarth et al. (1998b) who measured Olsen P in pastures down to 80 cm but found the most significant changes in the top 30 cm of the profile. In contrast, there was no significant difference between H_2O-P at the 10-20 cm and 20-30 cm depths (Figure 10) which is the approximate depth of the fragipan in the catchment. Dils and Heathwaite (1996) measured H_2O-P in a drained catchment in the UK and found greater H_2O-P at the same depth as a drain than in the C-horizon, which they attributed to preferential flow. While this could be the case in our study, it may also be indicative of enhanced P-release due to anoxia (Patrick & Khalid, 1974).

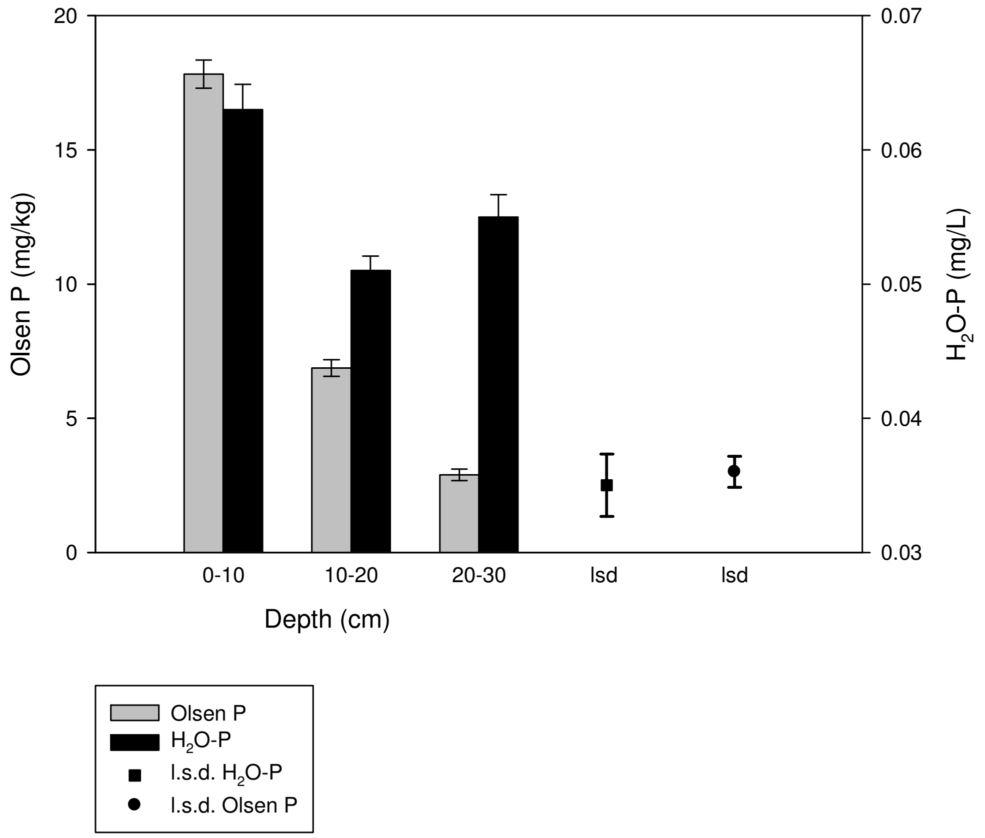

Figure 10 Soil test phosphorus concentrations measured in the 0-10, 10-20 and 20-30 cm depths at the pasture site. Different letters above bars, within groups (lower or uppercase), denote significant differences at the P < 0.05 level. Error bars represent ± 1 standard error of the mean (SEM).

The results of soil test P sampled from the pasture, trough and lane runoff plots are presented in Table 5 and show significant differences between the sites. The Olsen P concentrations were least in the pasture (24 ± 3 mg P/kg), higher around the trough (56 ± 21 mg P/kg) and greatest on the lane (201 mg P/kg). However, in winter, Olsen P concentrations at the lane plots (200 ± 2 mg P/kg) were less than those measured in summer (300 ± 15 mg P/kg). This is probably because the lane gets more use during the summer and more dung accumulates on the lane than during winter. The mean Olsen P concentration in the pasture plots (24 mg P/kg) did not differ greatly during the year, and is used here as a comparison with the intensive sampling around the trough and lane.

Table 4 Soil physical properties measured in the pasture with depth. The number of observations is in parentheses at the least significant difference at *P*<0.05 given for comparison between depths.

Depth (cm)	Bulk density (g/cm^3)	Pores < 30 μm	Pores >300 μm (%)	K_{sat} (mm/h)
0-5	1.06	41.8	8.6	209.8
	(23)	(24)	(22)	(22)
10-15	1.21	39.4	5.5	
	(24)	(24)	(24)	
20-25	1.32	37.1	4.5	
	(21)	(21)	(21)	
l.s.d.	0.05	2.1	1.6	

Around the trough, bulk density was greatest near the trough (1.36 ± 0.06 g/cm^3) and least 5 m from the trough (1.05 ± 0.05 g/cm^3), and at that distance, was not different to the pasture samples (1.06 ± 0.04 g/cm^3) (Table 4; Figure 11). Fine pores (< 30 µm) were the least affected by trampling around the trough and were similar to those measured in surrounding pasture (42%). However, the percentage of macropores (>300 µm), which has been shown to be inversely related to the potential for saturation-excess runoff (McDowell et al., 2003d), was 3.6 (± 2.3) % closest to the trough, and increased until 5 m away from the trough where porosities were similar to that measured in pasture (Figure 11). In contrast, Olsen P and H_2O-P concentrations were most enriched 3 m from the trough. This relates to approximately one cow length from the trough, creating a ring of dung deposition and enriched soil P.

Table 5 Mean soil test P concentrations in runoff plots. The least significant difference (l.s.d.) is given at *P*<0.01 for comparison between sites along with the number of samples (n).

Site	Olsen P (mg P/kg)	H_2O-P (mg/L)	n
Pasture	24	0.04	21
Trough	56	0.12	3
Lane	201	0.64	3
l.s.d.	13	0.04	

Both Olsen P and H_2O-P measured on the lane were significantly greater (P < 0.001) than in the section downslope of the lane and in the pasture plots. There were no significant differences measured between the three lane transects (Figure 9). The Olsen P concentrations measured on the lane were very high (300 ± 15 mg/kg) but, as mentioned earlier, can fluctuate during the year. Additionally, the Olsen P content in soil directly adjacent to the lane (82 ± 11 mg/kg) had a significantly greater Olsen P concentration than the pasture transects (24 ± 3 mg/kg), indicative of P transfer (probably via overland flow) from the lane to surrounding soil. The enriched P concentration in the lane, very low infiltration, high bulk density and proximity to the stream suggest the lane is a CSA.

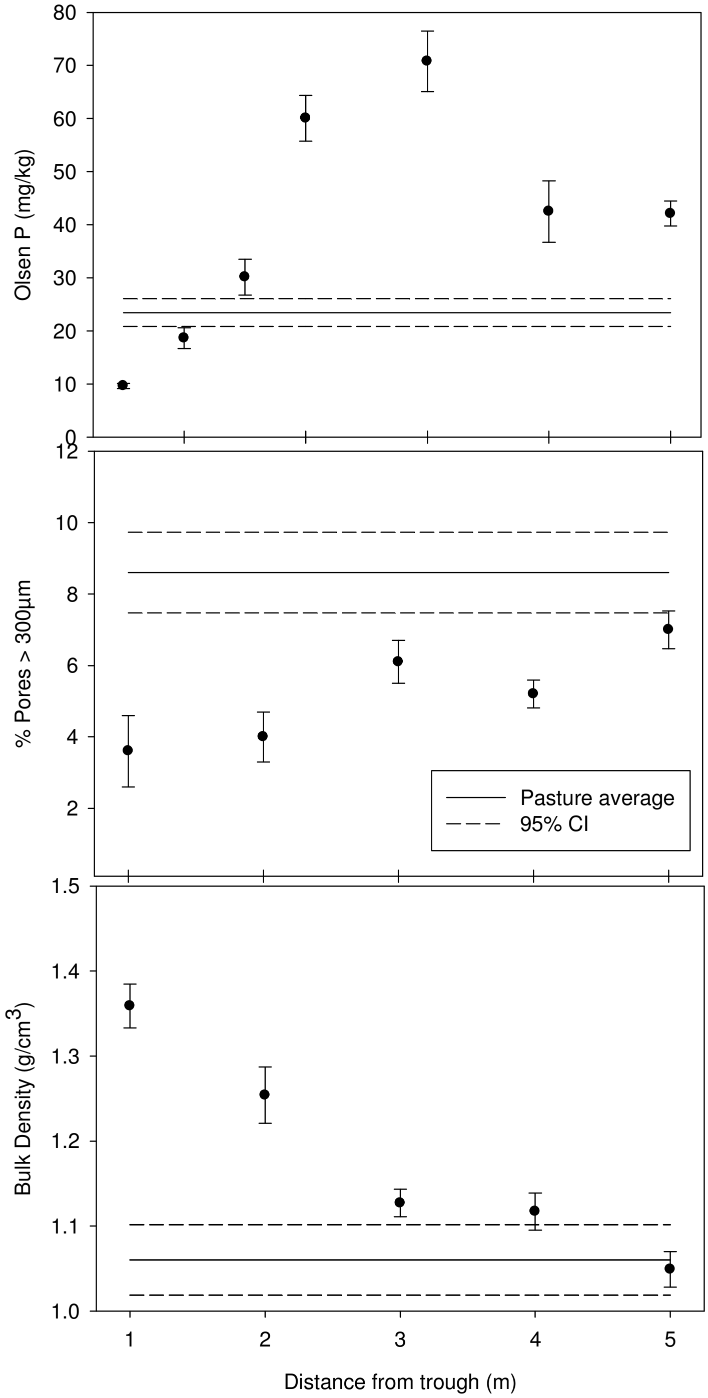

Figure 11 Variation in Olsen P concentrations, bulk density, percentage of macropores and saturated hydraulic conductivity (K_{sat}) with distance from the trough. The horizontal black line represents the mean of the measured property measured in the adjacent pasture and the dashed lines represent the 95% confidence interval of the mean. Bars represent ±1 standard error of the mean (SEM).

2.4.3.2 Overland flow plots

The average rainfall event size, together with the average depth of overland flow measured from each site for each season, is presented in Table 6. The total sum of overland flow from the pasture for the measurement period was 53 (± 35) mm, while the sum of storm flow for the same period was 617 mm. Therefore, approximately 3 – 14% of the storm flow was overland flow while the rest originated from subsurface flow. Overland flow volumes from the pasture showed no significant differences between seasons or soil moisture levels (Table 6) which may indicate that overland flow was generated by both infiltration-excess and saturation-excess mechanisms. Values for K_{sat} measured at the surface of the pasture were over 200 mm/h (Table 4) and therefore overland flow during wet winter conditions was probably via saturation-excess, since the average event size was < 30 mm (Table 6). In the summer when the soil is dry it can become hydrophobic which makes infiltration-excess runoff possible. Soils may switch from being hydrophilic to hydrophobic once soil moisture falls below a certain threshold (Doerr et al., 2000).

The mean concentrations of P fractions and SS in overland flow from the pasture, trough and lane were significantly different and decreased in the order of lane > trough > pasture. At the pasture sites, concentrations of SS were greater in winter than the rest of the year (0.38 and 0.10 g/L, respectively). Particulate P and SS concentrations in overland flow from the pasture were also greatest when soil moisture was > 60% (v/v), which suggests there was a greater detachment of soil particles, or dispersion of soil, under saturated conditions (Figure 12).

Table 6 Mean event size, overland flow volumes and the mean soil moisture for the season. The least significant difference (l.s.d.) at *P*<0.05 is given for comparison between seasons along with the number of samples (n).

	Rainfall	Pasture	Trough	Lane	Soil Moisture	n
			Overland flow			
	----------------------------	(mm)	----------------------------		(%v/v)	
Dec-Feb	23.4	0.4	8.5	8.9	33	7
Mar-May	25.1	2.6	6.9	16.9	46	9
Jun-Aug	16.5	1.5	22.0	11.6	63	15
Sep-Nov	19.8	0.7	11.5	1.7	51	6
l.s.d.	n.s.	n.s.	11.2	10.7	11	

Dissolved reactive P and TDP concentrations in overland flow from pasture appeared to vary little between seasons, in contrast to particulate P losses which increased during winter (Table 7). This was probably associated with the greater SS concentrations that were also observed in winter. Smith and Monaghan (2003) measured P losses in overland flow from a cattle-grazed pasture and found that most losses occur in late winter and spring. In contrast, concentrations of DRP and

TDP in overland flow from the trough sites were significantly greater in summer (1.1 and 1.6 mg/L) than in the rest of the year (0.4 and 0.5 mg/L), which coincides with the period of most animal use and hence dung deposition. Trough DRP and TP concentrations were similar to those measured by Dils and Heathwaite (1996) who sampled ponded water next to a drinking trough near Leicester, UK (0.58 and 2.40 mg/L, respectively). In contrast, the greatest DRP and TP concentrations from the lane were measured in autumn (6.1 and 11.3 mg/L). During the summer, dung may accumulate on the lane because it is too dry for decomposition. In the autumn, when breakdown is rapid, dissolved P from this temporary storage is released, in addition to the recently deposited dung (Rowarth et al., 1985). Mean particulate P concentration from the lane was greater in winter (2.1 mg/L) and autumn (4.0 mg/L) than in spring (1.0 mg/L) and summer (0.3 mg/L; Table 7). The concentrations of TP lost from the lane were in the same range as measured by Edwards and Withers (2008) from farm track runoff in the UK (0.24 – 7.30 mg/L). Linear regression revealed that DRP concentrations measured in lane runoff were related to total rainfall for the event (DRP [mg/L] = Total rainfall [mm] × 0.1155 + 0.939; R^2 = 21.2%; P = 0.004) across all seasons. This relationship implies that more rain will produce overland flow with greater concentrations of DRP and the source of P is not diluted by rainfall. Withers et al. (2009) also measured increases in P concentrations with increased flow rate in storm runoff from a farmyard. The maximum concentraton of DRP measured in that study (5.68 mg/L) was similar to the mean concentration from the lane in the autumn in our study (Table 7).

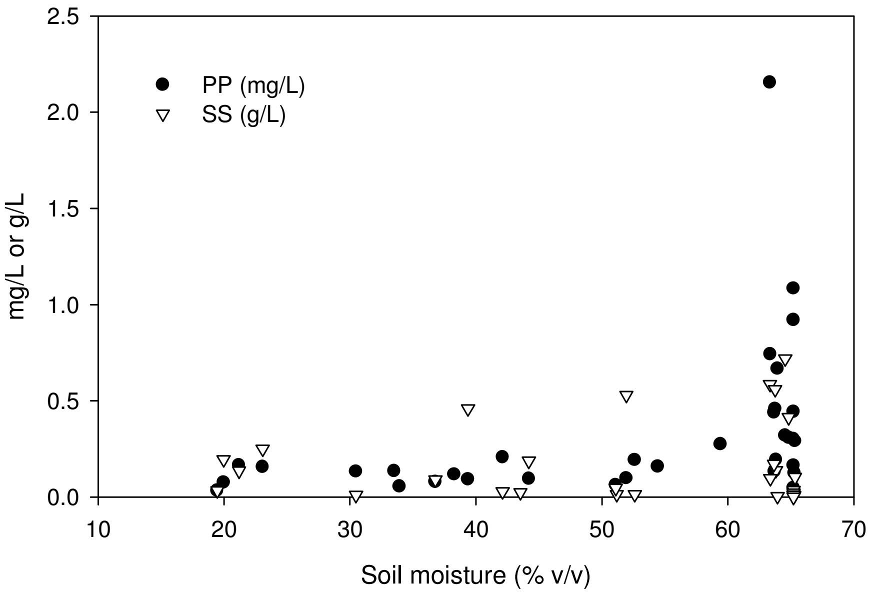

Figure 12 Suspended sediment (SS) and particulate phosphorus (PP) concentrations measured in overland flow from the pasture sites at different soil moisture levels.

Total P loads from the trough were greatest during summer and winter months, while most of the export from the lane occurred in autumn (Table 8). The pasture exported a relatively large percentage of its total load during autumn and winter. On average, the overland flow from the lane was most enriched in DRP (Table 8), while overland flow from the pasture and the trough was dominated by particulate P. However, TP losses fluctuated during the year. One example was from the lane where DRP comprised almost 80% of TP in the summer, but gradually declined through the year to a minimum of 33% of TP in spring (Table 8). As mentioned previously, warm, dry weather in the summer limits the decomposition of dung, resulting in an accumulation of dung on the lane which is released during summer rainfall events. This may explain the higher proportion of DRP observed in summer runoff. The coincidence of enriched DRP and warm conditions may stimulate algal growth in streams where P is limiting (Anderson et al., 2002). For the trough, DRP accounted for 42% of summer loads, while DRP from the pasture was around 18% of TP for most of the year, except in the winter when it decreased to 7% of TP. Particulate P was more responsive to soil moisture than other fractions (Figure 12). Dissolved fractions were not related to soil moisture, but instead season and possibly management (e.g. stock movement) effects.

Table 7 Mean concentrations of P fractions and suspended sediment in overland flow from pasture, trough area and lane plots. The least significant difference (l.s.d.) at P<0.05 is given for comparison between seasons along with the number of samples (n).

Season	DRP	DUP	TDP	PP	TP	SS	n
	-------------- (mg/L) --------------					(g/L)	
			PASTURE				
Dec-Feb	0.04	0.06	0.11	0.15	0.25	0.04	7
Mar-May	0.04	0.03	0.08	0.27	0.35	0.11	8
Jun-Jul	0.03	0.04	0.06	0.52	0.59	0.42	13
Aug-Sep	0.05	0.06	0.11	0.16	0.27	0.16	6
l.s.d.	n.s.	0.02	0.05	n.s.	n.s.	n.s.	
			TROUGH				
Dec-Feb	1.13	0.46	1.58	2.63	4.22	0.65	5
Mar-May	0.41	0.13	0.53	0.36	0.89	0.12	6
Jun-Jul	0.28	0.11	0.39	1.38	1.77	0.46	13
Aug-Sep	0.54	0.20	0.74	1.33	2.07	0.17	6
l.s.d.	0.38	0.21	0.50	1.92	2.28	0.43	
			LANE				
Dec-Feb	3.63	0.45	4.08	0.29	4.37	0.19	7
Mar-May	6.08	1.24	7.33	3.97	11.29	0.94	9
Jun-Jul	2.56	0.29	2.85	2.07	4.92	0.50	13
Aug-Sep	0.41	0.12	0.52	0.97	1.50	0.31	5
l.s.d.	2.19	0.56	2.22	2.95	4.32	0.67	

It has been previously reported that P loads from pastures are dominated by dissolved P because erosion should be limited (Haygarth et al., 1998b). However, as with more recent studies, (Ballantine et al., 2009; Bilotta et al., 2010) this data suggests a significant role for particulate P loss. Cornish et al. (2002) reported mean particulate P concentration in pasture runoff from a dairy paddock in Australia was 0.52 mg/L, and comparable with that found here (0.40 mg/L), while the measured TDP concentration in this study was much lower (0.08 mg/L; Table 6) than measured by them (0.66 mg/L). Cornish et al. (2002) also postulated that much of the particulate P lost was not from soil, but from organic P particles > 0.45µm. However, in this study, there was a significant correlation between the SS and PP concentrations (PP = 1.08 (SS) + 0.08; $R^2 = 67\%$; $P < 0.001$), which would suggest that the majority of particulate P was in fact soil-related, since most particulates in dung tend to be colloidal. Furthermore, dissolved unreactive P has been found to be a major fraction in runoff where sources of P are mainly faecal (Dils & Heathwaite, 1996). This was contrary to what was measured in the lane and trough, which was dominated by DRP, but concurs with the small proportion of DURP, and dominance of DRP, in overland flow from dung measured by McDowell (2006a).

2.3.3.3 Subsurface flow

Most subsurface flow samples were collected in winter, but some of the wells also intercepted subsurface flow during the summer. On average, the concentration of TP (0.48 ±0.04 mg/L) in subsurface flow was not significantly different to the mean TP concentration in overland flow (0.40 ±0.14 mg/L) from the pasture. This is in line with overland and subsurface flow measurements made by Uusitalo et al. (2001) who found no significant differences in TP concentrations between overland or subsurface flow. The overland flow TP concentrations were similar to those measured at Hillend (0.46 – 0.80 mg TP/L). However, the concentration of TDP was significantly greater in subsurface flow (0.15 ±0.02 mg/L) than in overland flow (0.08 ±0.02 mg/L; $P < 0.001$) at Hillend. Unlike concentrations in overland flow from the pasture, the concentrations of TDP and TP in subsurface flow were greatest during summer and least during winter. This would support the idea that P concentrations in the soil are built up during warmer months, due to mineralization of P, and depleted via leaching in the winter time (Dils & Heathwaite, 1996; Xue et al., 1998).

Table 8 The sum of the load of total P and suspended sediment for each season from the pasture, trough and lane plots and the mean percentage load of TP occupied by each P fraction. The least significant difference (l.s.d.) at *P*<0.05 is given for comparison between seasons along with the number of samples (n).

Season	TP	SS	DRP	DUP	TDP	PP	n
	(kg/ha/season)		------------------- (%)	------------------			
			PASTURE				
Dec-Feb	<0.1	1	18.2	26.7	44.9	55.1	7
Mar-May	0.1	39	16.7	15.0	31.7	68.3	8
Jun-Jul	0.2	92	6.6	10.4	17.0	83.0	13
Aug-Sep	<0.1	5	20.2	20.5	40.7	59.3	6
l.s.d.	-	-	9.7	8.4	14.4	14.4	
			TROUGH				
Dec-Feb	5.5	907	42.0	11.4	53.3	46.7	5
Mar-May	0.8	105	36.6	14.3	50.8	49.2	6
Jun-Jul	6.3	1651	20.9	8.6	29.5	70.5	13
Aug-Sep	2.2	138	34.9	14.9	49.9	50.1	6
l.s.d.	-	-	19.5	6.8	21.4	21.4	
			LANE				
Dec-Feb	3.6	116	78.8	11.0	89.8	10.2	7
Mar-May	19.0	1814	59.6	14.0	73.6	26.4	9
Jun-Jul	7.7	871	54.7	6.4	61.1	38.9	13
Aug-Sep	0.5	79	33.2	18.8	52.0	48.0	5
l.s.d.	-	-	17.6	8.3	17.4	17.4	

It has been proposed that the risk of contamination from CSAs occurs only when CSAs and streams are connected by overland flow (Gburek and Sharpley, 1998). However, the volume of overland flow measured in the pasture was only a small fraction of the runoff generated by the whole catchment. Shallow subsurface flow made up a greater proportion of total runoff, and because of the sizeable concentrations measured in the subsurface flow flushes, it would be sensible to explore the subsurface pathways further. Dils and Heathwaite (1996) also found that subsurface pathways were responsible for the transport of soluble P, but focused on measurements from tile drains. Srinivasan and McDowell (2009) also investigated subsurface flows in a similar pasture catchment and confirmed the need to investigate this pathway further.

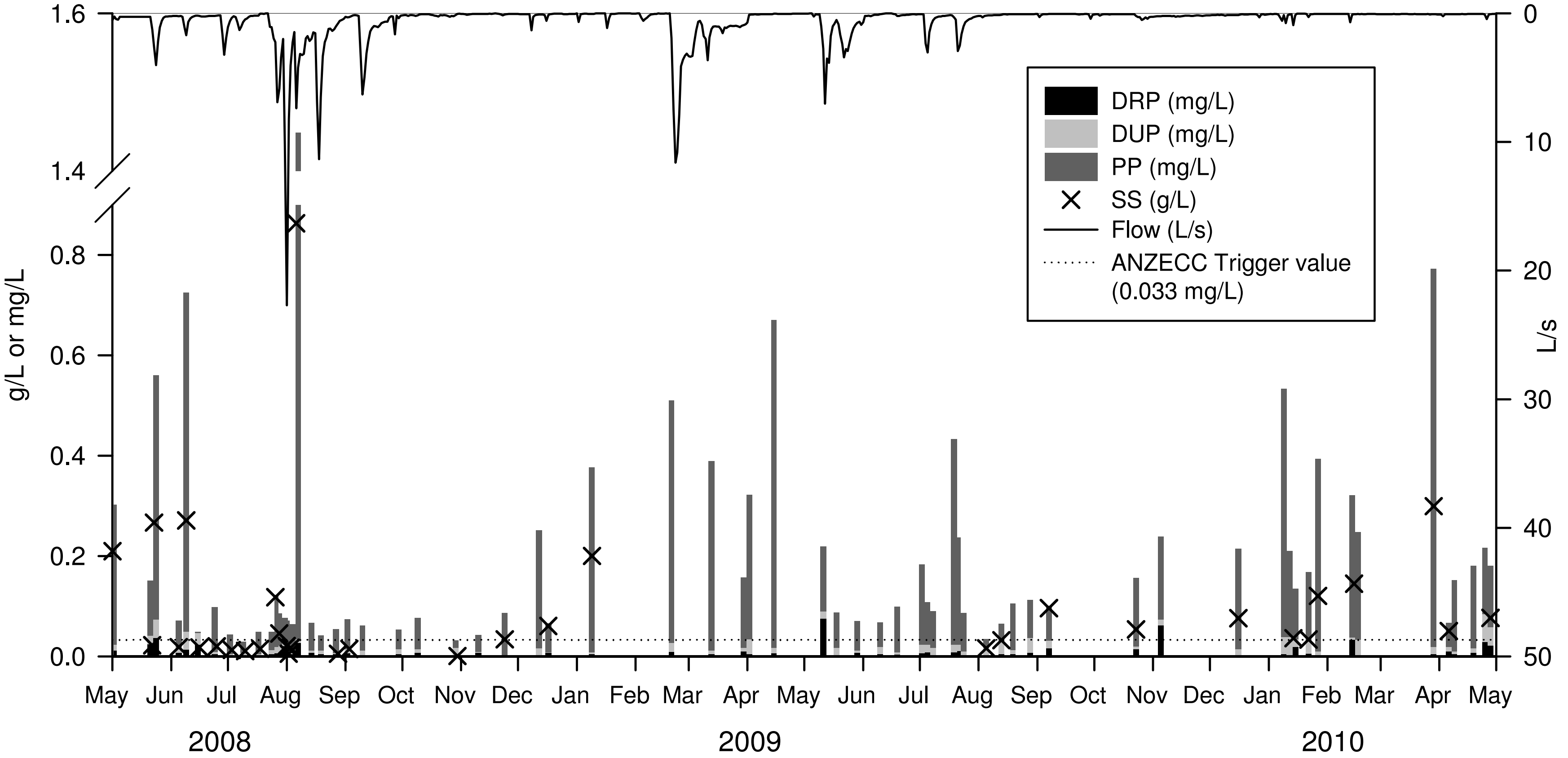

Figure 13 Phosphorus and SS concentrations, and flow measurements of stream water from Hillend catchment. Crosses represent suspended sediment concentrations. The dotted line marks the ANZECC trigger values TP concentration for an adverse effect on slightly disturbed lowland streams.

2.3.3.4 Catchment concentrations and loads

As can be seen in Figure 13, the greatest measured concentrations of TP did not always coincide with storm flow peaks, and some of the greatest concentrations of TP occurred during periods of low flow. Almost all of the samples taken exceeded the ANZECC trigger values (ANZECC-ARMCANZ, 2000) for TP in slightly disturbed New Zealand lowland streams (0.033 mg TP/L). When samples were divided into those taken at base flow and those taken during storm flow, there were no significant differences between mean DUP, PP, TP or SS concentrations. This was not consistent with results from other catchment studies (Cooper & Thomsen, 1988). However, the concentrations of DRP and TDP were significantly greater during storm flow (0.017 mg DRP/L and 0.031 mg TDP/L) than during base flow (0.007 mg DRP/L and 0.018 mg TDP/L). The greater concentrations of dissolved P during storm flow in the Hillend catchment could, in part, be the result of flushes of overland flow which, can be enriched with dissolved P from the source areas. As a comparison, the mean storm flow concentration of DRP and TP in a grazed pasture catchment (11 ha) in the North Island of New Zealand, was 10 times greater than during base flow (Cooper & Thomsen, 1988).

Despite enriched concentrations of dissolved P in storm flow, overall, the load of P from the Hillend stream was primarily as particulate P, in line with the high proportion of particulate P in overland flow from pasture. Enriched concentrations of PP and SS during base flows could be the result of cows grazing close to the stream. The soil type (Pallic soil) and large, steep, south-facing slope, is also prone to treading damage that would act as a source of sediment and particulate P, even during small rainfall events (Russell et al., 1998). However, the greatest load of SS and P from the catchment occurred during a series of large events in August, 2008 (Figure 13). Other major events were in February, March and May, 2009, which also shows that the greatest loads from this dairy pasture were not always in winter, but whenever a large storm event occurred. Substantial loads of P are a potential threat to surface water quality in the summer months when the warm conditions, together with excess nutrients, create a high risk of algal blooms (Anderson et al., 2002).

2.3.3.5 Potential contribution of CSAs and management

To assess the relative importance of lanes and troughs, loads were calculated from the runoff plots assuming 100% connectivity with the stream. Eight events were chosen to estimate the contribution of the pasture, trough and lane to total catchment load via overland flow. The criteria for selection were that overland flow was recorded at all sites and that the average overland flow was more than 0.5 mm. This selection resulted in only winter events when the soil was at or near saturation, and therefore selects for cases where it was likely that CSAs were connected to the stream.

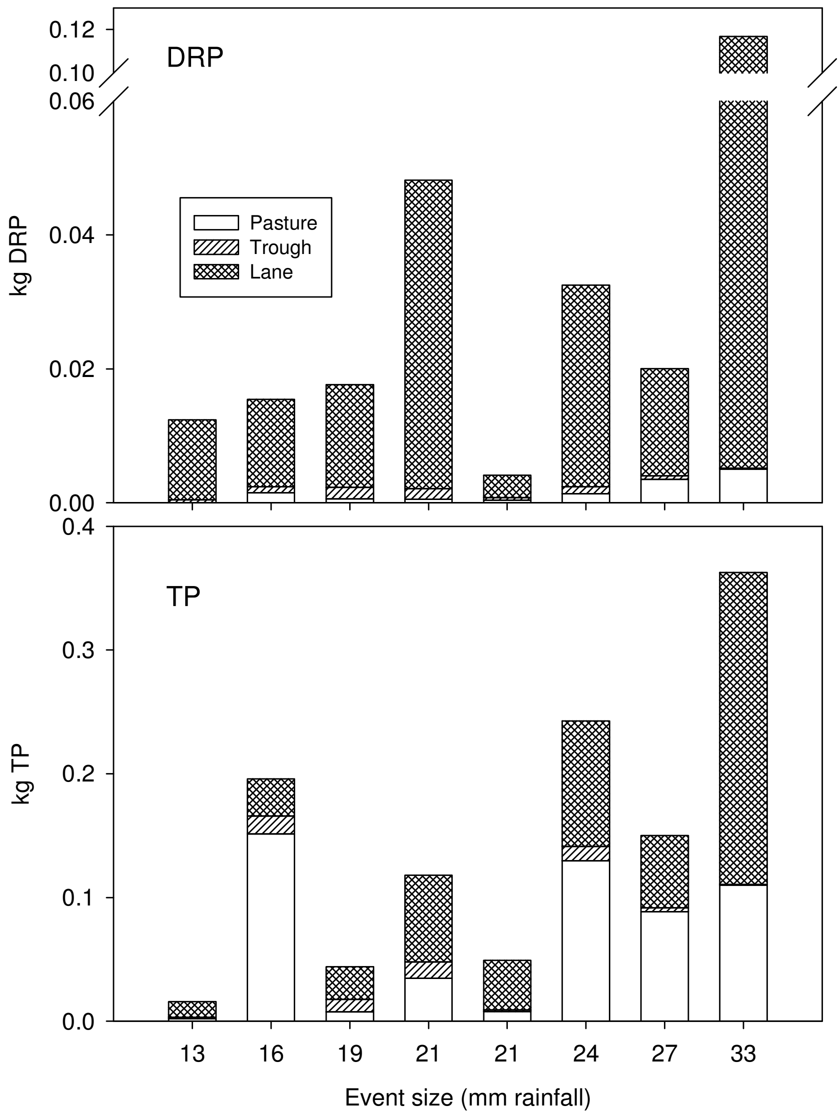

Figure 14 Modeled loads of DRP and TP in overland flow from the pasture, trough and lane sites for each of eight storm events under saturated conditions, ranked by storm event size.

The total catchment area is 4.1 ha. From the soil physical measurements, treading effects were restricted to within 5-6 m of the edge of the trough. Accounting for the diameter of the trough, the affected area was 143 m^2 or 0.4% of the catchment area. About 100 m of the lane was included in the catchment, representing 0.06 ha or 1.4% of the catchment. The load of each P fraction and SS was calculated by multiplying the measured load by the area.

On average, pasture was estimated to contribute most particulate P loss during the 8 events (Figure 14). Although the lane amounted to 1.4% of the total catchment area, it was estimated to contribute 80 to 96% of the DRP load. The lane was also a major contributor of TDP (84%), and in turn > 50% of the TP load from the catchment. The load from the trough was typically < 10% of the total load for all P fractions and SS lost from the catchment implying that, even under saturated conditions, the contribution from the trough areas would be of minor importance. Figure 14 shows that rainfall event size was not a reliable predictor of event loading from any of the sites, although the general trend was for a greater contribution from the pasture with increasing event size.

This exercise demonstrates the potential importance of areas such as lanes to act as CSAs of P loss to a stream. Minimising either the source or transport factors will decrease the risk of P loading. Regular treading means that it would be difficult to try to improve the infiltration capacity of the lane. This, coupled with regular dung deposition, means that even if successful, the lane would continue to contribute significantly to stream P loading. A potential solution for the lane would be to target the P source with alum (aluminium sulphate) which can bind up P by forming a stable complex with aluminium before it is lost to overland flow (Smith et al., 2001a). Vegetated buffer strips would be another strategy that could intercept nutrients in overland flow before they reach the stream (Aye et al., 2006). Alternatively, the land could be engineered or redirected so that overland flow does not reach the stream.

2.4 Conclusions

Section 2.2 in this chapter identified variables to predict P loss among the sites: soil test P (Olsen P or H_2O-P), % bare ground and % saturation. All of these factors are easily measurable or visible to the farmer or land manager. In terms of simplicity, and being able to make good decisions to decrease P losses on a day to day basis, these data are encouraging news. In general, soil test P is more insensitive to change than the percentage of bare ground or the percentage saturation. Hence, the data would suggest focusing management more on decreasing the percentage of bare ground or grazing when the soil is drier. The latter could be achieved with the use of a stand-off area that cows utilise until pastures dry and are in better condition to graze.

This initial experiment in Section 2.2 has shown that highly trafficked areas in dairy pastures are significant source areas of P loading to waterways. The greatest potential threat found here was from lanes and around troughs. Crossings and gateways also had the potential to export significantly greater loads of P than the surrounding pasture, but how much depended on usage. This was a first step in defining the relative importance of specific sources in catchment loss of P. However, of equal importance are the soil conditions that induce stream connectivity and when these conditions occur. It is obvious that judicious decisions need to be made over the positioning of areas like lanes and gateways that have potential to contribute large amounts of P loss to a stream. Accordingly, these areas should be positioned well away from a stream or routes that could connect them to the stream.

Sampling of watering trough and lane areas in Section 2.3 revealed some significant differences in soil physical and chemical properties to those of the surrounding pasture. The treading and deposition of dung by dairy cows compacted soil, decreased infiltration and enriched soil Olsen P concentrations. This lead to greater specific yields of P compared to the pasture. Saturated conditions (soil moisture ≥ 60% v/v) increased the loads of particulate P and SS lost from the pasture compared to the lane or trough. However, lower infiltration and greater deposition of dung on the lane and trough compared to the pasture, meant that loads enriched with dissolved P from the lane and trough occurred year round, independent of soil moisture. The variation in sources and their potential transport was also evident on a catchment scale. Loads of P within the stream were dominated by particulate P which was consistent with the high proportion of particulate P in overland flow from pasture. The contribution to P loss by sub-surface flow was important given that overland flow comprised only 14% of discharge and contained a mean P concentration similar to overland flow from the pasture. Scaling up runoff plot data for 8 storm events where overland flow was likely, and the contribution of sub-surface flow less, suggested that the lane, which made up < 2% of the catchment, contributed up to 89% the DRP load. The variation of sources and transport processes with season, coupled with this data, enables the targeting of CSAs to decrease any deleterious effect by P loss on surface water quality. Given the loss during summer and high algal availability of dissolved P, mitigation strategies should target decreasing dissolved P loss from critical source areas such as the lane areas studied here.

Chapter 3 - Methods selection: evaluation of base solutions to determine equilibrium phosphorus concentrations (EPC_0) in stream sediments

Once sediment is transported from land to water it continues to interact with the surrounding water. Generally, interactions between sediment and the water column occur during the stable base flows of summer and autumn. These flows coincide with warm conditions that promote algal growth and are of great interest. One measure for determining to what extent the sediment acts as a source or sink of P is the EPC_0. This chapter outlines an analysis of methods used for determining the EPC_0 of sediments. A review of the literature was performed to determine an appropriate method to be used in this study for measuring the effects of sediment sorption and desorption of P. However, there was no satisfactory method found and therefore it was decided to do a comparison of the different base solutions most used in the literature before deciding which method to use.

3.1 Introduction

The EPC_0 is calculated by incubating a known mass of sediment with base solutions (or background matrix) spiked with P, but the exact method used varies. Many authors have shown that using dried or fresh sediments, different sediment size fractions, and different sampling depths all influence determination of the EPC_0 (Klotz, 1988; Baldwin et al., 2000). However, most agree that fresh sediment that has been sieved to < 2 mm represents the most reactive fraction of sediment under natural conditions. However, many different base solutions have been used containing salts such as $CaCl_2$ or KCl in varying concentrations. Klotz (1988) demonstrated a wide range in EPC_0 values measured for the same sediment using base solutions with different Ca concentrations (0-400 mg Ca/L). Calcium chloride concentrations used in the literature range from 0.0005*M* to 0.01*M* (Table 9), which are believed to reflect the water's natural chemistry, although few authors supply measurements of the sample water's chemistry. These differences make comparing studies difficult, and also have the potential to influence conclusions about sorption or desorption of P from sediments.

Table 9 Base solutions (or background matrices) used for equilibrium phosphorus concentration (EPC_0) determination by different authors.

Base solution	Reference
0.01*M* $CaCl_2$	(White & Beckett, 1964; Kunishi et al., 1972; McCallister & Logan, 1978; Nair et al., 1984; Stutter & Lumsdon, 2008)
0.0005*M* $CaCl_2$	(Klotz, 1988)
0.01*M* KCl	(Reddy et al., 1998; Gimsing & Borggaard, 2001; Lair et al., 2009)
Tap water	(James & Larson, 2008)
Deionised water	(Ryden et al., 1972; Sharpley et al., 1981; Koski-Vahala & Hartikainen, 2001; McDaniel et al., 2009)
Stream water	(Popova et al., 2006; Smith et al., 2006; McDaniel et al., 2009)

3.2 Materials and methods

3.2.1 Sampling strategy

The sediments used for evaluating EPC_0 methods were sampled from sites located in the South Island of New Zealand, primarily around the city of Dunedin (45° 52'S, 170° 30'E), with one additional site 100 km north (N. Otago) and one site 250 km to the west (Hauroko). The sites sampled represented a variety of stream orders, catchment sizes, land uses and geology that are common in this region (Table 10). The predominant land use in the area is pasture, grazed by sheep and beef cattle, with forestry and some urban areas also represented. Two of the catchments are covered in native bush vegetation (Native and Hauroko) and one was subject to historic wastewater treatment discharge (Riccarton). The underlying geology is generally made up of schist and volcanic basalt on the hills, and sedimentary rocks (including sandstone and limestone) in the plains and valleys. Records indicate that average annual rainfalls at the Dunedin, N. Otago and Hauroko sites were 850, 625 and 1200 mm/yr respectively.

Sediment samples were taken during base flow conditions in the summer when the sediments would be most likely to be in equilibrium with the overlying water. Sediment samples from the 0-4 cm depth were collected with a plastic scoop. Approximately 10 L of stream water was also collected at each site for use as the base solution and for later analysis (see below).

Once back in the lab, sediments were wet-sieved through a 2 mm sieve, centrifuged for 15 minutes at 1100 × *g*, and the supernatant discarded. The residual slurry was kept refrigerated until analysis (within 6 days). A subsample of sediment slurry was oven dried (105°C) for 24h to determine its moisture content and to calculate dry weight equivalents. Sediment retained on a 2 mm sieve was dried and expressed as a percentage of the total sediment sampled.

Table 10 Characteristics of the sampled stream sites.

Site	Land use	Stream order (Hortonian)	Geology	Soil (US soil taxonomy)
Native	Native bush	1	Schist	Dystrochrept
INV	Sheep, deer and beef	1	Volcanic	Fragichrept
Riccarton	Forestry, urban, sheep and beef	4	Sedimentary	Fluvent
Rail R	Forestry and pasture	3	Schist, sedimentary in valley	Ustochrept
Owhiro	urban and pasture	3	Schist and sedimentairy	Ustochrept
Brockville	Sheep, bush and some urban,	1	Volcanic	Dystrochrept
N.Otago	Sheep	4	Sedimentary including limestone and sandstone	Rendolls
Abbotts	Urban and pasture	3	Schist and sedimentary	Dystrochrept
Hauroko	Native bush	2	Granite	Orthod

3.2.2 Sediment and stream water analysis

Sediment pH was determined on fresh sediments using a 1:10 sediment-water suspension. Particle size fractionation was performed on dried sediments using the dry sieving method of Sheldrick & Wang (1993) for sediments > 63 µm, and wet sieving for sediments < 63 µm. Total P concentrations in sediment samples (TP_{sed}) were measured by *aqua regia* digestion according to the method of Crosland et al (1995). Analysis of total C and N was performed using an elemental analyser (Elementar Vario-Max CN). Concentrations of total P, C and N were measured on dried (oven, 105°C, ≥ 24 h) sediments.

Measurements of pH were made on site with a portable HI 9812 pH/EC/TDS meter (Hanna Instruments) which was calibrated before each use using two buffer solutions. In addition, the dissolved oxygen (DO) concentration of streambed sediments was measured *in situ* at 6 cm depth with a DO6 dissolved oxygen palm-top meter (Eutech/Oakton Instruments). Suspended sediment was determined after passing a known volume of liquid sample through a glass-fibre filter paper and weighing the oven-dried residue. Dissolved reactive P (DRP) was determined on filtered (<0.45 µm) water samples via the colorimetric method of Watanabe and Olsen (1965). The remaining nutrient concentrations were measured using ICP-OES.

3.2.3 EPC determination

One gram (dry weight equivalent) of slurry was measured into a 30 mL centrifuge tube and 20 mL of base solution added. Fresh sediments were used because drying of sediment has been shown to increase adsorption capacity (Klotz, 1988). The addition of chloroform, toluene or other reagents to minimise the effect of biota was avoided to ensure that the concentration of P measured in the sediment-slurry was not artificially increased (Meyer, 1979). Six base solutions (stream water, 0.01*M* KCl, 0.01*M* $CaCl_2$, 0.0005*M* $CaCl_2$, deionised water (D.I.) and tap water) were used which contained initial P concentrations of 0, 0.5, 1, 2, 5, 10, 20, and 50 mg P/L (as KH_2PO_4). These base solutions (or background matrices) have been commonly used in the measurement of EPC_0 in other studies (Table 9). The samples were mixed for 20 hours using an end-over-end shaker and then centrifuged at 3600 × *g* for 10 minutes. An aliquot of the supernatant was used for determining P concentrations. Centrifugation did not always remove all of the fine material from a few of the samples. In these instances a replicate of the same sample and volume was measured in the spectrophotometer, but without molybdate-blue added, to serve as a blank.

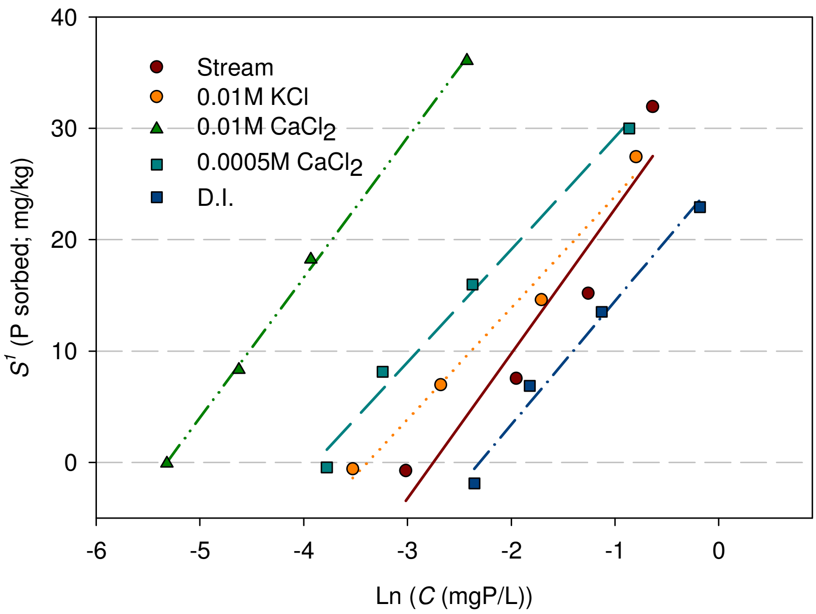

Figure 15 An example of EPC_0 determination of sediments taken from the Rail site with the different base solutions used. A linear regression was used to calculate the EPC_0 which is where there is no adsorption or desorption of P i.e. where $S^1 = 0$. All the calculated values are found in table 11.

At low P concentrations, there is typically a linear relationship between P sorbed or desorbed, (S^1, mg/kg) and the concentration of P in solution after equilibrium (*C*, mg/L) (House & Denison, 2000). This linear relationship was found only at one site (INV) and a linear isotherm was used to calculate EPC_0 for that sediment. However, this was not the case for most of the measurements.

Instead, the first four points of the sorption isotherm (i.e. solutions < 2 mg P/L) were used and the concentration of P (C) was log transformed to obtain a linear relationship that fitted the data better than the commonly used Freundlich equation (Figure 15). The linear equation used was as follows:

$$S^1 = m * \ln C + b$$

EPC_0 was calculated as the concentration where there was no adsorption or desorption of P, or where $S^1 = 0$. The terms m (slope) and b (y axis intercept) were obtained from the regression.

Statistical analyses, including correlation and regression analyses, were carried out using GenStat (v. 11.1). Correlation probabilities were evaluated using a two-sided test different from zero. In all other analyses the standard error of the mean, and/or correlation coefficients (r) and probabilities, are as specified in the text.

3.3 Results and discussion

3.3.1 EPC_0 comparison among base solutions

Equilibrium P concentrations measured using stream water (EPC_0^S) ranged from 0.004 to 2.64 mg P/L, illustrating the diverse nature of sites sampled (Table 11). A correlation analysis between EPC_0^S and stream water constituents showed EPC_0^S values were significantly related to the concentration of Ca in base solutions(log transformed; r = 0.74, $P < 0.05$), but not with any other constituent measured. Further analysis indicated that those solutions that best predicted stream water EPC_0^S values were solutions with Ca concentrations close to those of stream waters. This is demonstrated in Figure 16 as the relative difference between EPC_0 measured with different base solutions and EPC_0^S, and the difference between base solution Ca concentration and stream Ca concentrations. The points closest to $y = 0$ are the base solutions with EPC_0 values closest to the stream waters, and the points closest to $x = 0$ are the base solutions with Ca concentrations closest to the stream waters at each site. This figure shows that EPC_0 determined with either 0.0005M $CaCl_2$ or tap water gave values similar to EPC_0^S. For EPC_0 values < 0.5 mg P L^{-1}, a solution of 0.0005M $CaCl_2$ gave the best prediction of stream EPC_0 values (y = 1.01x+0.01; R^2 = 0.55, P<0.05) followed by tap water (y = 1.23x+0.02; R^2 = 0.55, P<0.05). The tap water and dilute $CaCl_2$ solutions had similar Ca concentrations, 18 and 20 mg Ca/L, respectively, and were close to the average stream water Ca concentrations of 21 mg Ca/L (Table 12) and the average baseflow Ca concentrations measured across New Zealand by Close & Davies-Colley (1990).

Table 11 Equilibrium phosphorus concentrations (EPC_0) measured using different base solutions. Figures in parentheses are the standard errors of the linear regressions used to derive EPC_0 concentrations.

	Stream	KCl	Tap	0.01*M* $CaCl_2$	0.0005*M* $CaCl_2$	Deionised water
Abbotts	0.018	0.004	0.026	0.009	0.007	0.032
	(0.002)	(0.002)	(0.001)	(0.001)	(0.000)	(0.003)
Brock	0.532	0.013	0.155	0.003	0.103	0.527
	(0.015)	(0.005)	(0.017)	(0.002)	(0.010)	(0.025)
Hauroko	0.096	0.116	0.127	0.025	0.101	0.152
	(0.008)	(0.011)	(0.013)	(0.003)	(0.027)	(0.017)
INV	2.639	0.247	2.007	0.020	1.157	3.328
	(0.011)	(0.009)	(0.033)	(0.002)	(0.010)	(0.047)
Native	0.110	0.158	0.131	0.017	0.113	0.320
	(0.024)	(0.010)	(0.013)	(0.006)	(0.009)	(0.013)
N Otago	0.044	0.286	0.165	0.020	0.130	0.366
	(0.005)	(0.012)	(0.007)	(0.008)	(0.021)	(0.015)
Owhiro	0.162	0.209	0.205	0.019	0.186	0.376
	(0.019)	(0.017)	(0.006)	(0.003)	(0.017)	(0.036)
Rail	0.064	0.034	0.033	0.005	0.020	0.100
	(0.020)	(0.005)	(0.014)	(0.000)	(0.003)	(0.010)
Riccarton	0.004	0.016	0.006	0.003	0.005	0.017
	(0.002)	(0.004)	(0.004)	(0.001)	(0.003)	(0.006)

Of the other solutions evaluated, EPC_0 values determined using 0.01*M* $CaCl_2$ were all less than EPC_0^S values, and EPC_0 values determined with D.I. water were all much greater. This is due to the different ionic strengths of the solutions, i.e.: those with greater ionic strength tend to suppress desorption (Barlow et al., 2004). The KCl base solution had the greatest spread of the solutions used and KCl is sometimes preferred in alkaline soils to avoid the precipitation of Ca-P (Gimsing & Borggaard, 2001). However, unlike Ca, K is a monovalent cation and is more easily displaced than Ca (except where preferential adsorption to some vermiculites occurs) (Page et al., 1967) thus affecting P sorption.

There were two sites where measured EPC_0 values were at least three times greater than found for other sediments: INV and Brockville (Table 11). At these two sites, EPC_0^S concentrations were high (> 0.5 mg P/L), but coincided with the lowest Ca concentrations in stream water (3-4 mg Ca/L) and greatest sediment C concentration (17 and 87 g/kg) (Table 12).

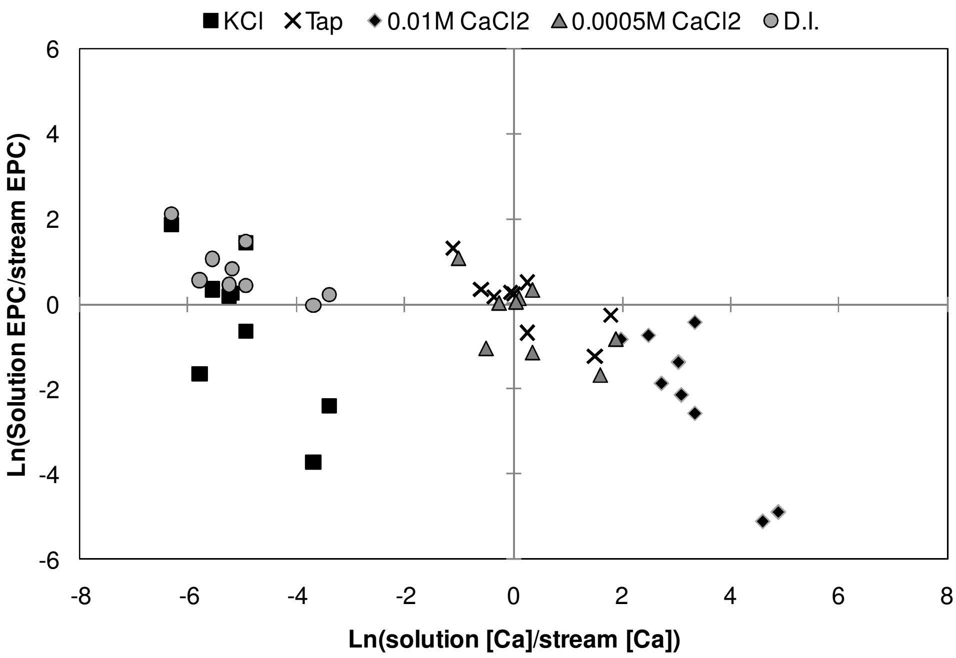

Figure 16 Comparison of EPC_0 determined using with different solutions with the EPC_0 measured using stream water, and of the calcium (Ca) concentrations in each solution compared with stream water Ca concentration. Those closer to the origin (0) are most closely associated with one another (i.e. The points closest to y = 0 are the base solutions with EPC_0 values closest to the stream waters, and the points closest to x = 0 are the base solutions with Ca concentrations closest to the stream waters).

Low Ca concentrations decrease the likelihood of insoluble Ca-P forming (House and Denison, 2002), while organic C has been found to be poorly P sorptive and to bind P in labile humic-P complexes (McDowell & Sharpley, 2001). Both mechanisms function to enrich P in solution. Although no reason was given, Klotz (1988) also found a negative correlation between organic matter concentration and EPC_0. Contrary to other findings, no significant correlation was found for EPC_0^S and any of the particle sizes measured (McDowell et al., 2003e; Palmer-Felgate et al., 2009).

The data from this study supports the premise that solutions using Ca are suitable for use in measuring EPC_0 values in stream waters. Popova et al. (2006) used both $CaCl_2$ solution and stream water as a background matrix, and concluded that filtered stream water may be appropriate to use because using $CaCl_2$ produced EPC_0 values three times greater than when filtered stream water was used. However, they did not specify the concentration of $CaCl_2$ used, but only that it had an "electrical conductivity similar to that of ambient stream water". This emphasizes the need to determine the Ca concentration of stream water before choosing a suitable Ca-based base solution.

Table 12 Some chemical and physical characteristics of stream water and sediment sampled. Parameters presented are: dissolved reactive phosphorus (DRP), suspended sediment (SS), calcium (Ca), sodium (Na), dissolved oxygen (DO, measured at 6 cm depth), carbon (C), carbon to nitrogen ratio (C:N), total sediment less than 2 mm (< 2mm) and total phosphorus (TP_{sed}). Standard error of the mean (SEM) is given in parenthesis after TP_{sed} (n = 3).

Site	------------------ Stream water -------------------						-------------------- Sediment ---------------------				
	pH (H_2O)	DRP (mg/L)	SS (g/L)	Ca (mg/L)	Na (mg/L)	DO (%)	C (g/kg)	C:N	pH (H_2O)	< 2mm (%)	TP_{sed} (mg/kg)
Native	8.0	0.054	1.39	26	14	68	5.4	53.5	7.4	62	259 (7)
Riccarton	9.0	0.003	2.69	14	13	-	3.4	11.1	7.7	4	243 (5)
Rail R	7.8	0.011	1.12	14	14	68	2.3	44.1	7.4	20	237 (24)
Brockville	8.0	0.007	2.01	4	6	80	17.7	47.9	7.2	15	875 (74)
Owhiro	7.7	0.076	5.10	18	19	16	1.6	65.9	7.3	32	238 (24)
INV	7.1	0.022	5.42	3	13	24	87.1	13.3	-	100	1311 (18)
N.Otago	7.7	0.010	1.92	55	26	87	1.9	18.8	7.8	15	392 (16)
Abbotts	5.2	0.003	2.51	33	28	23	6.8	20.1	5.7	9	1426 (209)
Hauroko	8.0	0.015	1.38	19	5	64	0.6	5.6	7.6	53	271 (6)

3.3.2 EPC_0 and DRP

All sites were sampled at base flow conditions on the assumption that sediments would be at or near equilibrium with dissolved P concentrations in the stream. However, at all of the sampled sites EPC_0 values were greater than DRP concentrations, implying that the sediments were acting as a P source. Many authors have used 0.01*M* $CaCl_2$ as a base solution and found the opposite, i.e. that EPC_0 < DRP (e.g. Klotz, 1988). However, this study has shown that using 0.01*M* $CaCl_2$ as a base solution decreases EPC_0 values, perhaps to levels lower than would be expected if stream water was used. In contrast, Haggard et al. (2007) used filtered stream water as a base solution and found EPC_0 > DRP. In most of the sediments studied by Jarvie et al. (2005) and Palmer-Felgate et al. (2009), DRP was also greater than EPC_0.

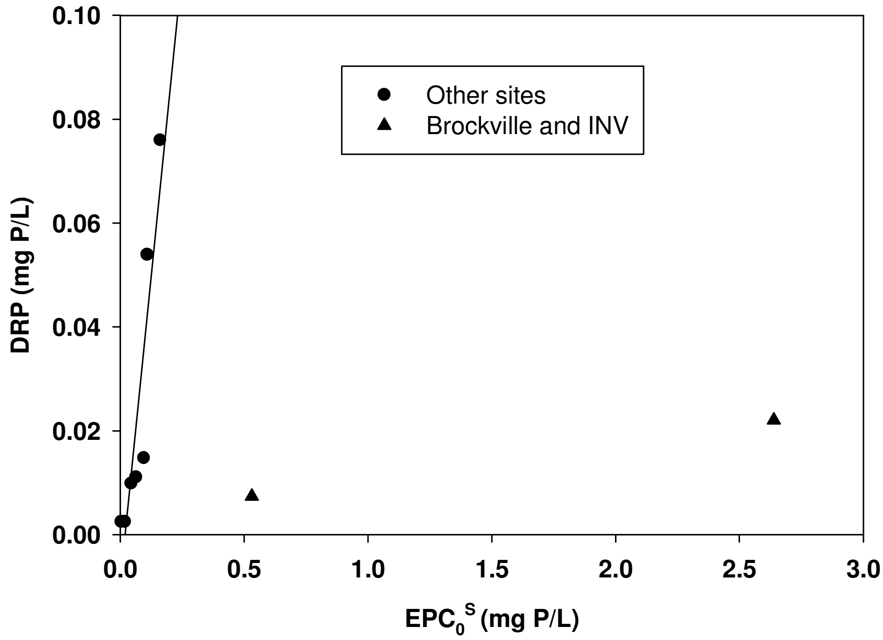

Figure 17 Linear regression of dissolved reactive phosphorus (DRP) and the equilibrium phosphorus concentration measured using stream water as a base solution (EPC_0^S). The Brockville and INV sites are not included in the regression because stream water P concentration and sediment are not in equilibrium. Regression equation: DRP = 0.473 EPC_0^S – 0.009; R^2 = 0.79, P< 0.01.

Unique sediment characteristics will affect measurements from site to site, but one conclusion that can be drawn from all of these studies is that sediments are rarely in equilibrium with the water column. Jarvie et al. (2005) measured the kinetics of DRP uptake by sediments and found that most of the uptake occurred in less than 1 hour which would suggest that equilibrium can be swift. However these sediment samples were continuously shaken together with the P solution, a situation which is unlike stream conditions. That experiment was intended to simulate the mixing with the 'boundary layer' which is in immediate contact with sediments, and the speed of equilibrium of this layer with the rest of the water column was not estimated. This raises two important questions. Firstly, how much importance should be placed on EPC values when equilibrium is rarely achieved in the field? Secondly, do the sampling and measurement techniques, as wide and varied as they are, adequately measure the true EPC_0? For example, the Invermay site had the greatest EPC_0^S (2.64 mg P/L). However, monthly monitoring has shown that the maximum DRP measured at the site has not been > 0.04 mg DRP/L in the last two years.

For additional insight into how sediments affect DRP concentrations, a linear regression was performed plotting DRP against EPC_0^S (Figure 17). Excluding the INV and Brockville sites, there was a positive relationship between EPC_0^S and DRP concentration (DRP = 0.473 EPC_0^S – 0.009; *R2* = 0.79, *P*< 0.01). The two excluded sites have EPC_0^S values much greater than stream water concentrations, and from this analysis it can be assumed that sediments were either not in equilibrium with the stream water (as suggested above for the INV site) or that these sediments play a minor role in regulating stream water P concentrations. Other studies have found stronger (Klotz, 1988; Klotz, 1991; McDowell et al., 2003e; Palmer-Felgate et al., 2009) or weaker (Jarvie et al., 2005; Haggard et al., 2007) relationships between EPC and DRP in stream water (or no relationships (House & Denison, 2002; Smith et al., 2006)). It is thus difficult to draw conclusions between EPC_0 and DRP because sampling methods and EPC_0 determinations are so different and, in some cases, do not reflect stream conditions.

3.4 Conclusions

The EPC_0 is a potentially useful measure of the relative capacity of sediments to act as sinks or sources of P to the overlaying water during base flows. The results showed that the choice of method for determining EPC_0 values had a marked effect on results and the utility of conclusions. Using a standardized 0.01*M* $CaCl_2$ solution, as proposed by Nair et al. (1984) and used by many others, resulted in lower EPC_0 than if stream water was used. This would lead to overestimation of the sediment's capacity to remove P from overlying water and would therefore be an ineffective indicator of how sediment affects in-stream processing. A more standardized method of sampling and analysis should be developed to allow comparison between studies, and even draw conclusions within studies as to the fluxes of P to and from sediments. It is suggested that unless variation in stream water chemistry can be proven to be small when sampled, then it should become standard practice to measure stream water Ca concentration before deciding on an appropriate base solution from which to derive EPC_0 values. Our initial suggestion is to use a base solution of 0.0005*M* $CaCl_2$ if the mean Ca concentration at base flow is 5-55 mg/L otherwise D.I. may be suitable for low Ca (< 5 mg/L) stream waters.

Chapter 4 - The effects of scale and in-stream processes on phosphorus and sediment export from the Silver Stream Catchment

In the previous chapters P loss was assessed at a limited range of scales, from boxes to plots and up to paddock. However, transformations and potential sources and sinks of P increase with increasing scale. Therefore the findings of the previous experiments need to be put in context with greater scale measurements to elucidate the importance of small scale sources. This chapter explores variations in connectivity with time and at increasing spatial scales. The experiments were designed to meet objectives III and IV. The method for determining the EPC_0 described in Chapter 3 was used this chapter which presents the results of a two year investigation of P and SS at different scales in the Silver Stream catchment.

4.1 Introduction

Factors that affect P and sediment loading to waterways have been researched at many different scales, from plot to catchment. As spatial scales increase, so do the number of potential sources, hydrological processes and in-stream processes which add to the complexity of the results observed. For instance, while small scale plot studies may examine specific pathways and mechanisms, large catchment studies commonly represent the cumulative effects of flow pathways, land use and best management practices. Several studies have attempted to compare scales in the same catchment; however many of them come from the UK where there is greater point source loading than in many catchments in New Zealand. New Zealand farming systems also have unique P and sediment loading patterns that are dominated by diffuse sources, due to lower nutrient inputs and year round outdoor grazing regimes.

Annual loads are the sum of base flow and storm flow loads. Storm flows are the focus of many studies because they make up the bulk of P and sediment exports (Gburek & Sharpley, 1998). Haygarth et al. (2005) gathered data from a variety of scales and flow conditions. At small scales, most P was transported during storm events, while at larger scales base flow gained importance. Forms of P also tended to become dominated by dissolved fractions in base flow and with increasing catchment scale.

In-stream physical, chemical and biological processes further affect P concentrations (Green et al., 1978). Much of these processes are controlled by bed sediments. These sediments are derived from local parent material and deposited particulates eroded from surrounding land uses during different flow regimes (Kändler & Seidler, 2009). Hence, they reflect land use and hydrological processes. However, it is the ability of bed sediment to influence P at base flow, when most interaction between sediment and the water column occurs, that is of most interest. This is because this tends to occur during summer and autumn when, collectively, stable flows and warm conditions promote algal growth. The ability of the sediment to control base flow P concentration can be estimated by determining the equilibrium phosphorus concentration (EPC_0) and, in turn, whether the sediment acts as a sink or source of P (Klotz, 1988).

The aim of this experiment was to measure P concentrations and loads as part of a nested study (5 - 8,100 ha) of the Silver Stream catchment, to understand the sources and flow regimes responsible for stream P inputs at different scales, and to make some inferences about connectivity and potential management. A second aim was to examine the role of sediments, both suspended in stream flow and from the stream bed, to investigate their role in controlling P concentration, especially when the growth of toxic algae is likely.

4.1.1 Site description

The Silver Stream near Dunedin, New Zealand is a tributary of the Taieri River, the fourth longest river in New Zealand. Algal growth is P-limited (McDowell et al., 2009a), but sufficient P is available for the Silver Stream to become eutrophic in summer and autumn. Recently, a toxic algae (*Phormidium)* was identified in the waters of the Silver Stream and highlighted as the cause of canine deaths (Threlfall, 2010). Land use in the catchment is highly varied with native forest, production forestry and sheep/beef or deer farming dominating the uplands, and dairy farming and the township of Mosgiel (population 10,000) dominating the lowlands (Table 13). The hydrology of the smallest scale catchment (farm 1-3 + Spring) has been described in depth by Srinivasan & McDowell (2009), while a general overview of the soils and land use of the entire catchment can be found in Srinivasan & McDowell (2007).

The effects of scale and land use were investigated using a nested catchment approach in the Silver Stream catchment. Eight sampling sites were set up in the catchment (Figure 18). There were four farm catchment sites and one spring site where 90% of the land use comprised grazing of sheep and beef. The other sites, forest, bridge and outlet were dominated by forest (a mixture of native bush and plantation forestry; Table 13).

4.2 Materials and methods

4.2.1 Stream measurements

Flow was measured in the smallest catchment (farm 1 & 2) using H-flumes linked via a stilling well to a datalogger which recorded the flow every 20 minutes. At the larger scale (farm 4 and forest), pressure transducers were used to measure the flow of water. The area of water was determined by measuring the cross-section of the stream at the monitoring site. A rating curve was generated by regularly measuring the height and flow velocity of the stream at the site and using the cross-section data to determine the volume of water which was then related to the output of the pressure transducers. The site at the outlet is maintained by the Otago Regional Council (ORC, 2010). Due to high flows causing logger malfunction, some flow measurements were lost during this period. The two loggers in the smallest catchment (farm 1 & 2) were missing data for eight months in 2008 and 2009, and four weeks in 2010. The flow was periodically measured manually during this time and related to the logger at farm 4 (1 km downstream). The linear regression between the sites accounted for 85% and 83% of the variation respectively, and was used to interpolate missing data at sites 1 and 2.

Water sampling was carried out every three to four weeks, with additional samples taken during storm flows. Samples were taken from all sites on the same day, usually within two hours. A total of 55 samples were collected during both storm flow and base flows from May 2008 – June 2010. Water samples were taken back to the lab for analysis immediately after collection. Suspended sediment was determined by passing a known volume of sample through standard glass-fiber filter paper and weighing the oven-dried residue. Dissolved reactive P (DRP) was determined in a sample that had been filtered (<0.45μm), while total P (TP) and total dissolved P (TDP) involved a persulphate digestion, with and without filtration (< 0.45 μm), respectively. Concentrations of P in the solutions were measured via the colorimetric method of Watanabe and Olsen (1965). Dissolved unreactive P (DUP) was calculated as the difference between TDP and DRP, while particulate P (PP) was calculated as the difference between the dissolved fraction (TDP) and TP.

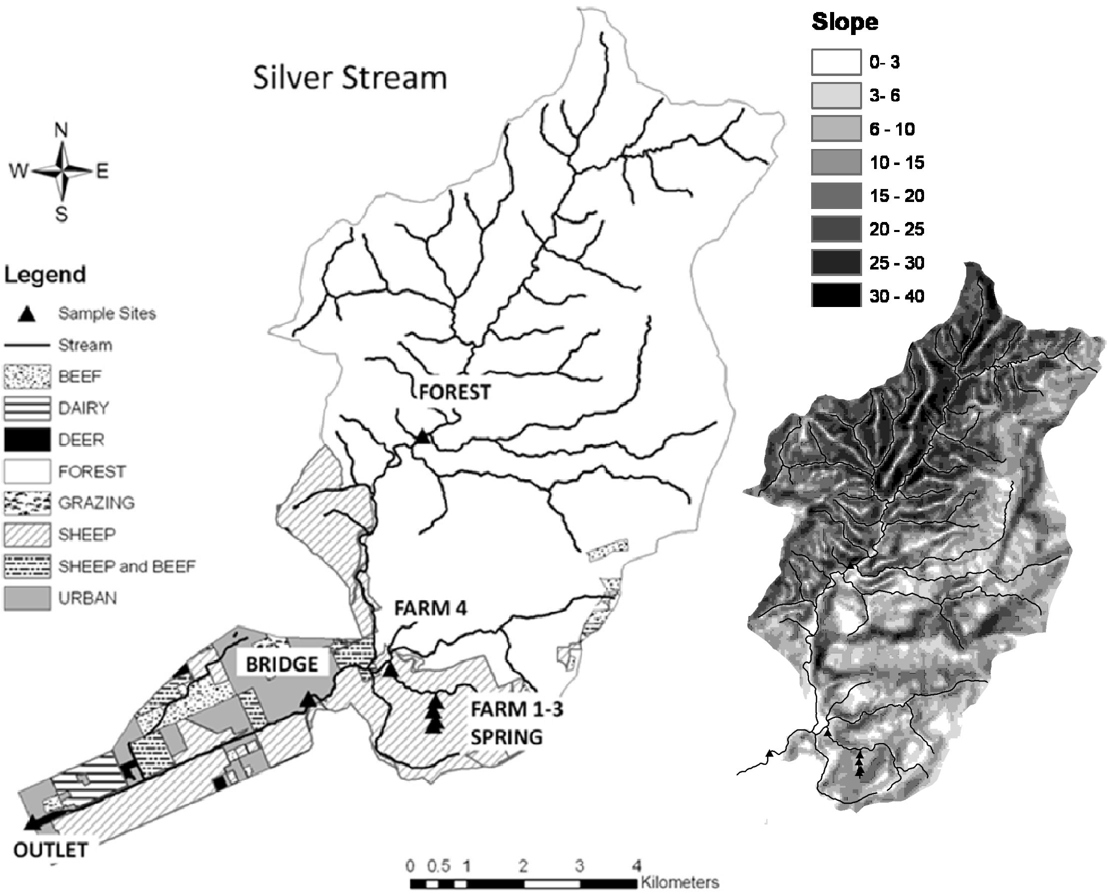

Figure 18 Land use and sampling and recording sites in the Silver Stream catchment (left) and catchment slope (%) with flat (white) to steep (black) terrain. Note scale is for land use map on left.

To estimate load, extrapolation methods could not be used due to the lack of a relationship between flow and P or SS concentration. Instead, daily concentrations of P and SS were derived by linear interpolation between sampling points (Rekolainen et al. 1991; Kronvang & Bruhn, 1996). The load was calculated as per Method 1 used by Rekolainen et al. (1991) or Method 2 used by Johnes (2007):

$$\text{Load} = K \sum_{i=1}^{n} (C_{di} Q_{di})$$

where; C_{di} is the concentration of sample (mg/L), Q_{di} is daily mean flow (L/s); K is a conversion factor to convert mg/s to kg/d (0.0864); and n is the number of days in the relevant measurement period.

4.2.2 Sediment analysis

Bed sediment samples were taken during base flow conditions in spring, autumn and summer when the sediments would be most likely to be in equilibrium with the overlying water column. Sediment from the 0-4 cm depth was collected with a plastic scoop. Approximately 10 L of stream water was collected just before the sediment, to be used as a base solution and for P and SS analysis above.

The method for measuring EPC_0 was the same method as used in Chapter 3. In that study it was found that the base solutions gave varying EPC_0 results, some not at all consistent with measurements made with stream water. Therefore, the stream water was used as a base solution. Sediments were wet-sieved through a 2 mm sieve, centrifuged for 15 minutes at 1100 × *g* and the supernatant discarded. The residual slurry was kept refrigerated until analysis the next day. A sub-sample of sediment slurry was oven dried (105°C) to determine moisture content and to calculate dry weight equivalents.

One gram (dry weight equivalent) of slurry was measured into a 30 mL centrifuge tube and 20 mL filtered (<0.7µm, glass fiber) stream water containing initial P concentrations of native background P, and native + 0.5, 1, 2, 5, 10, 20, or 50 mg P L^{-1} (as KH_2PO_4). Fresh sediments were used because drying sediment has been shown to increase its adsorption capacity (Klotz, 1988). The samples were then mixed using an end-over-end shaker for 20 hours. Afterwards, the samples were centrifuged at 3600 × *g* for 10 minutes. An aliquot of the supernatant was used for P determination, using the same methods as DRP analysis above.

Suspended sediment EPC was measured in a different way from bed sediments to better simulate stream conditions. During a storm event, two sets of four 200 mL containers were filled with stream water. The containers were taken back to the lab, weighed to estimate the volume and DRP was determined. Container 1 received no P additions while containers 2-4 were spiked with a 100 mg P/L solution to make final concentrations of 0.1, 0.5 and 1 mg P/L. The same additions were made to containers filled with deionised water as a blank. The containers were then shaken for one minute and kept at 4°C overnight to maintain a similar temperature to that of the stream. In the morning the samples were taken out, shaken again, left to sit for approximately 30 minutes and then shaken one last time before DRP was determined.

At low concentrations there is typically a linear relationship between P adsorbed or desorbed, (S^1; mg/kg) and the concentration of P in solution after equilibrium (C; mg/L) (House & Denison, 2000). The resulting equilibrium P concentration at zero net sorption or desorption (EPC_0) was calculated where $S^1 = 0$. The terms m = regression slope and b = y axis intercept were obtained from the regression.

The collected data was first tested for normality, and then statistical significance between sites was determined using ANOVA. The differences between storm and base flow were then compared using a paired T-test, both in Genstat (11th Edition, 2008).

4.3 Results and discussion

4.3.1 Hydrology

The discharge of water for each site, normalized for land area (mm^3/mm^2), showed general correlation with the rainfall for each month (Figure 19). There were almost identical trends in discharge at the forest site and at the catchment outlet. A linear regression of the monthly water export from the forest site and the catchment outlet showed that flow at the forest site contributed 32% of catchment discharge, but made up *c*. 50% of the catchment area (Table 13) (forest = 0.32* catchment outlet; $r^2 = 0.99$, $P<0.001$). Another regression showed that the farm 4 site accounted for 4% of discharge in the wider catchment (farm 4 = 0.04 *catchment outlet; $r^2 = 87\%$, $P<0.001$), in line with the size of the farm 4 catchment relative to the outlet (3.7%). It is well documented that forested catchments generate less discharge than catchments draining other land uses such as pasture (Newson, 1994).

Table 13 Land use, size, median flow rates and annual mean discharge with storm flow in parentheses at each sampling site in the Silver Stream catchment. For the location of sampling sites refer to Figure 18.

Site	Forestry	Sheep	Beef	Urban	Catchment size	Median flow	Discharge
	--------------- (%)	---------------			(ha)	(L/s)	(mm/yr)
farm 1	0	90	10	0	5	0.6	460 (153)
farm 2	0	90	10	0	10	0.8	602 (180)
farm 3	0	90	10	0	24	-[a]	-[a]
spring	0	90	10	0	-[a]	-[a]	-[a]
farm 4	8	82	10	0	303	15	458 (246)
forest	100	0	0	0	4,065	111	292 (126)
bridge	85	11	<1	3	7,128	-[a]	-[a]
outlet	76	14	2	5	8,130	297	446 (197)

[a]Not measured

The percentages of discharge attributed to storm flow were 32, 32, 54, 43 and 44 for farm 1, 2, 4, forest and outlet sites, respectively (Table 13). On most occasions, discharge was greater at the farm 4 site than at the forest site (Figure 19), which suggests that there is some storage of water in the forest catchment during low flow periods. More water storage would imply greater contact time with the soil and therefore expression of soil P concentrations. Although no measurements were made, much work, including some in the Silver Stream catchment by Srinivasan and McDowell (2007), indicates that soil P concentrations in forested areas are less than in grazed pasture. This would mean that the greater proportion of base flow in the forested sub-catchment would result in much lower P concentrations compared to other sub-catchments.

Interestingly, the mean concentration of DRP during base flow at the farm sites (0.014 mg/L) was more than double that at the forest site (0.006 mg/L; Table 14).

4.3.2 Concentrations and loads of P fractions and suspended sediment

Although highlighted in some studies (e.g. Wilcock et al., 1999; McDowell & Wilcock, 2004), little evidence was found for a seasonal difference in concentrations of P fractions or SS. One exception, DURP or organic P, was significantly greater during the summer months than during other times of the year at five out of the 8 sites monitored (Table 15). Organic P can originate from faeces (Dils & Heathwaite, 1996), which explains the enriched concentrations found at the farm sites and the low concentrations in the forest. Also, increased biological activity, due to land use or season, has been found to increase organic P turnover, resulting in a greater potential loss from the soil system (Turner, 2005). Dissolved unreactive P can also be found in the first flushes of an event after a dry period, as observed by Turner & Haygarth (2000) and Stutter et al. (2008). Most wetting and drying of soils occurs in the summer and it was expected that this effect would only be evident at small scales. However, enriched concentrations of DURP were found at most sites during the summer (Table 15), and even at the outlet of the catchment. The time it takes for these organic forms to hydrolyse into reactive P forms is variable, but can be rapid (Turner et al., 2002), acting as an important source of P for algal growth in the summer months.

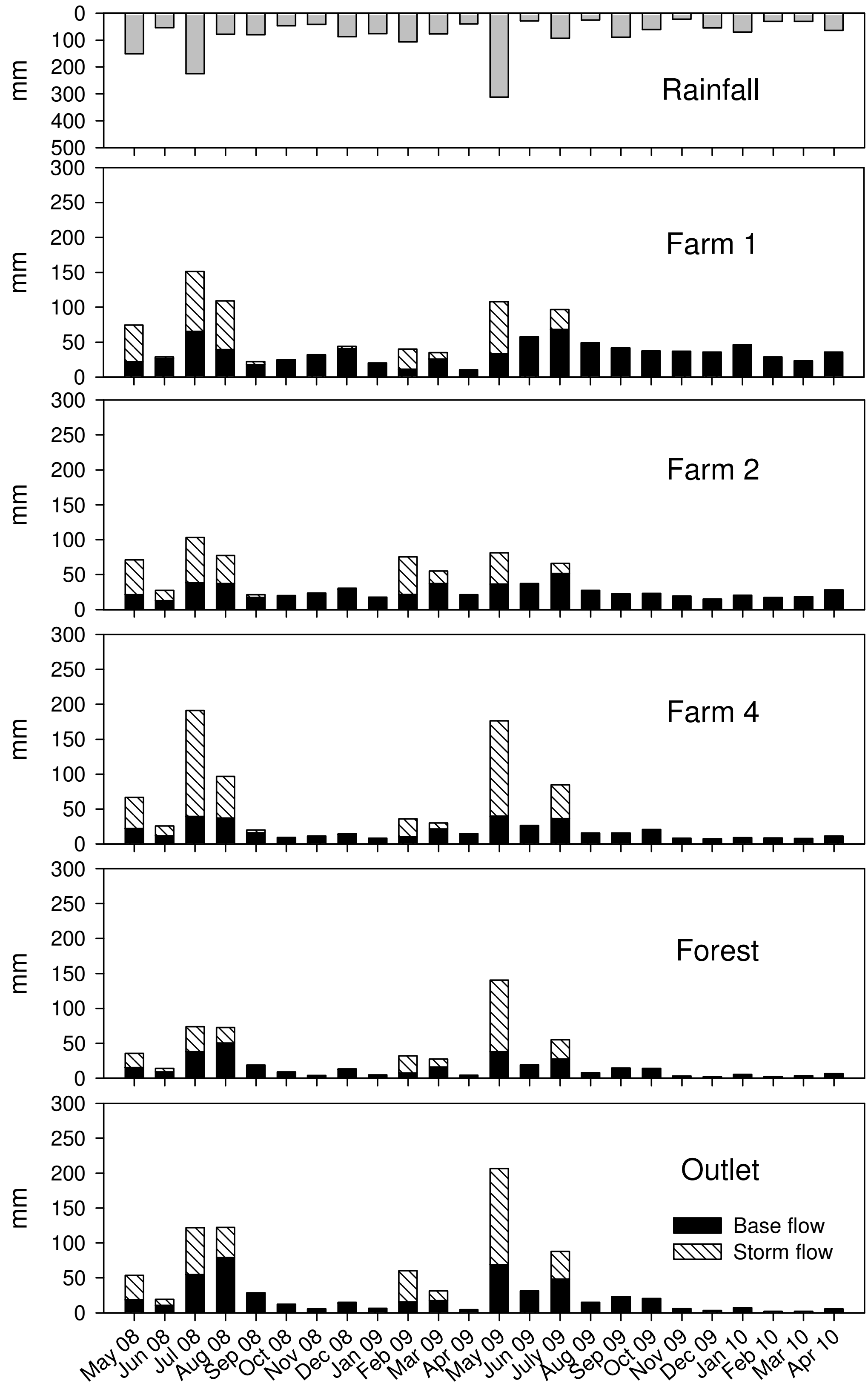

Figure 19 Rainfall and runoff (split into base flow [black bars] and storm flow [hatched bars]) by month for each site in the catchment.

Table 14 Mean concentrations of P fractions (dissolved reactive phosphorus (DRP), dissolved unreactive P (DUP), particulate P (PP) and total P (TP)) and suspended sediment (SS) concentrations taken during base and storm flow. The range of concentrations is given below in parenthesis and at the bottom of the table. The least significant difference (l.s.d.) at 5% level is given for the comparison of means between sites. A paired T-test was used to compare base and storm flow concentrations, the statistical significance (* = *P*<0.05; ** = *P*<0.01; * = *P*<0.001) is given with the mean storm concentrations.**

Site	Base (mg/L)					Storm (mg/L)				
	DRP	DUP	PP	TP	SS	DRP	DUP	PP	TP	SS
Farm 1	0.017	0.014	0.082	0.113	63	0.030*	0.030	0.267	0.327	315
	(0-0.06)	(0-0.07)	(0.01-0.26)	(0.02-0.38)	(5-611)	(0.01-0.06)	(0.01-0.10)	(0.05-1.54)	(0.08-1.68)	(38-1357)
Farm 2	0.010	0.017	0.042	0.067	14	0.028***	0.028	0.127*	0.183**	83*
	(0-0.06)	(0-0.06)	(0-0.15)	(0.01-0.20)	(0-71)	(0.01-0.04)	(0.02-0.08)	(0.04-0.35)	(0.07-0.48)	(21-268)
Farm 3	0.009	0.015	0.041	0.063	13	0.040**	0.029**	0.149	0.218*	111
	(0-0.04)	(0-0.05)	(0-0.12)	(0.02-0.15)	(0-76)	(0-0.08)	(0.01-0.07)	(0.04-0.48)	(0.07-0.61)	(14-439)
Farm 4	0.019	0.010	0.034	0.062	14	0.023	0.017**	0.130	0.169	110
	(0.01-0.07)	(0-0.02)	(0.01-0.10)	(0.03-0.13)	(0-275)	(0.01-0.04)	(0.01-0.03)	(0.03-0.44)	(0.06-0.50)	(18-451)
Spring	0.050	0.009	0.035	0.094	12	0.154***	0.030**	0.047	0.231***	25*
	(0-0.29)	(0-0.03)	(0-0.20)	(0.01-0.36)	(0-58)	(0.07-1.67)	(0.01-0.13)	(0.02-0.67)	(0.11-2.49)	(5-38)
Forest	0.006	0.007	0.026	0.038	7	0.011**	0.008	0.194*	0.212*	365
	(0-0.02)	(0-0.02)	(0-0.10)	(0.02-0.12)	(0-85)	(0.01-0.01)	(0-0.3)	(0.04-0.35)	(0.06-0.39)	(33-1220)
Bridge	0.005	0.007	0.026	0.038	10	0.013***	0.014	0.168	0.194*	321
	(0-0.02)	(0-0.03)	(0-0.09)	(0.02-0.11)	(0-121)	(0.01-0.02)	(0-0.05)	(0.01-0.59)	(0.06-0.61)	(38-1514)
Outlet	0.006	0.007	0.031	0.044	8	0.016***	0.019	0.205*	0.240*	242*
	(0-0.02)	(0-0.02)	(0.01-0.08)	(0.01-0.12)	(0-95)	(0.01-0.03)	(0-0.07)	(0.04-0.57)	(0.07-0.60	(37-769)
l.s.d. (5%)	0.009	0.004	0.013	0.018	17	0.026	0.018	0.209	0.224	292

Table 15 Mean seasonal concentrations of dissolved unreactive P (mg/L) and the least significant difference (l.s.d.) at 5% level for the comparison of mean seasonal concentrations. Concentrations in bold text are significantly greater than at least one of the other concentrations.

Site	Summer	Autumn (mg DURP/L)	Winter	Spring	l.s.d. 5%
Farm 1	**0.022**	0.015	0.006	0.011	0.010
Farm 2	**0.032**	0.012	0.007	0.013	0.011
Farm 3	**0.023**	0.007	0.008	0.016	0.009
Farm 4	0.013	0.007	0.009	0.010	0.005
Spring	**0.013**	0.007	0.006	0.011	0.006
Forest	0.009	0.005	0.006	0.007	0.004
Bridge	0.010	0.005	0.008	0.008	0.005
Outlet	**0.010**	0.003	0.007	**0.010**	0.004

Mean concentrations of sediment and P fractions were generally greater in storm than in base flows at all sites (Table 14). However, among P fractions at both farm 1 and 4 sites, only one significant increase in concentration occurred in storm flow compared to base flow. Particulate P made up 37-72% of TP during base flow and 20-91% during storm flow (Table 14). Consequently, DRP concentration as a percentage of TP was generally less in storm (5-18%) than in base flow (13-56%). Among sites, mean base flow concentrations of DRP were greater from the grazed farm 1, spring and farm 4 sites than from the forest site and the catchment outlet (Table 14). Between farm 1 and farm 2 sites there is a stretch of the stream channel overgrown by weeds which slows the flow of water and allows for sedimentation, retention and transformation of P. Our work and others (Haggard et al., 2001) shows that these areas act as a sink during low flows, and as a potential source during storm events, as shown by increases in PP, TP and SS concentrations in storm compared to base flow (Table 14). The low proportion of storm flow at this site (Table 13) acts to decrease the overall load of P and SS lost compared to other sites (Table 16).

In summary, the data in Table 16 suggests that more P, particularly dissolved P, was lost from the smaller scale sites at base flow, while storm flow derived particulate P dominated loads at larger scales. For example, while the mean annual loading rate of P and SS by area (or export coefficient) was, like the discharge, greater at smaller scales (e.g. farm 1; Table 16), loads and discharge during specific events or times of year (e.g. May 2009) were greater at the forest and outlet than at other sites (Figures 19 & 20). The percentage load of DRP and DURP in storm flow compared to total flow appeared similar across all scales, and within the range of 32 studies for forested and mixed catchments (including sheep and beef) by McDowell and Wilcock (2008). Interestingly, storm flow accounted for 80-95% of PP, TP and SS at the forest, outlet and farm 4 site, while at smaller scales only 51-74% was exported during storm flow (Table 16). This is contrary to the findings of Haygarth et al. (2005) who showed that most P was transported during storm events, particularly at small scales. However, Cooper & Thomsen (1988) found a greater proportion of TP and DRP export occurred during storm flow from forested sites compared with pasture sites. Mosley (1981) described how organic debris in forested streams creates temporary damns in the stream which are broken up and release sediment when a storm event occurs.

Table 16 Annual mean loads (kg/ha/yr) of P fractions and suspended sediment from 5 catchment sites for 5/2008-5/2010. Numbers in parentheses refer to the load generated during storm flow.

Site	DRP	DURP	PP (kg/ha/yr)	TP	SS
Farm 1	0.12 (0.05)	0.10 (0.04)	0.66 (0.36)	0.88[1](0.45)	870 (584)
Farm 2	0.07(0.04)	0.09 (0.03)	0.35 (0.20)	0.52 (0.27)	225 (168)
Farm 4	0.10 (0.06)	0.08 (0.05)	0.62 (0.53)	0.80 (0.64)	529 (492)
Forest	0.02 (0.01)	0.03 (0.02)	0.35 (0.31)	0.40 (0.34)	692 (664)
Outlet	0.05(0.03)	0.06 (0.03)	0.66 (0.58)	0.76 (0.64)	745 (703)

[1] note that due to rounding, TP may not equal the sum of DRP, DURP and PP.

The difference in pasture loads during storm flow could therefore lie in the generation and loss of P from different zones. For example, previous work has identified critical source areas that contributed P and SS during small and large storms in the farm catchment (McDowell & Srinivasan, 2009). These included direct stock access to the stream, deer wallows and tracks, fence lines and gateways, as well as saturation areas close to the stream. In contrast, Srinivasan & McDowell (2007) hypothesized that only saturated areas were likely to contribute P. These areas would only contribute once they were fully active (i.e. saturated) and this, together with the possibility of increased storage (as mentioned above), supports their activity during large storms.

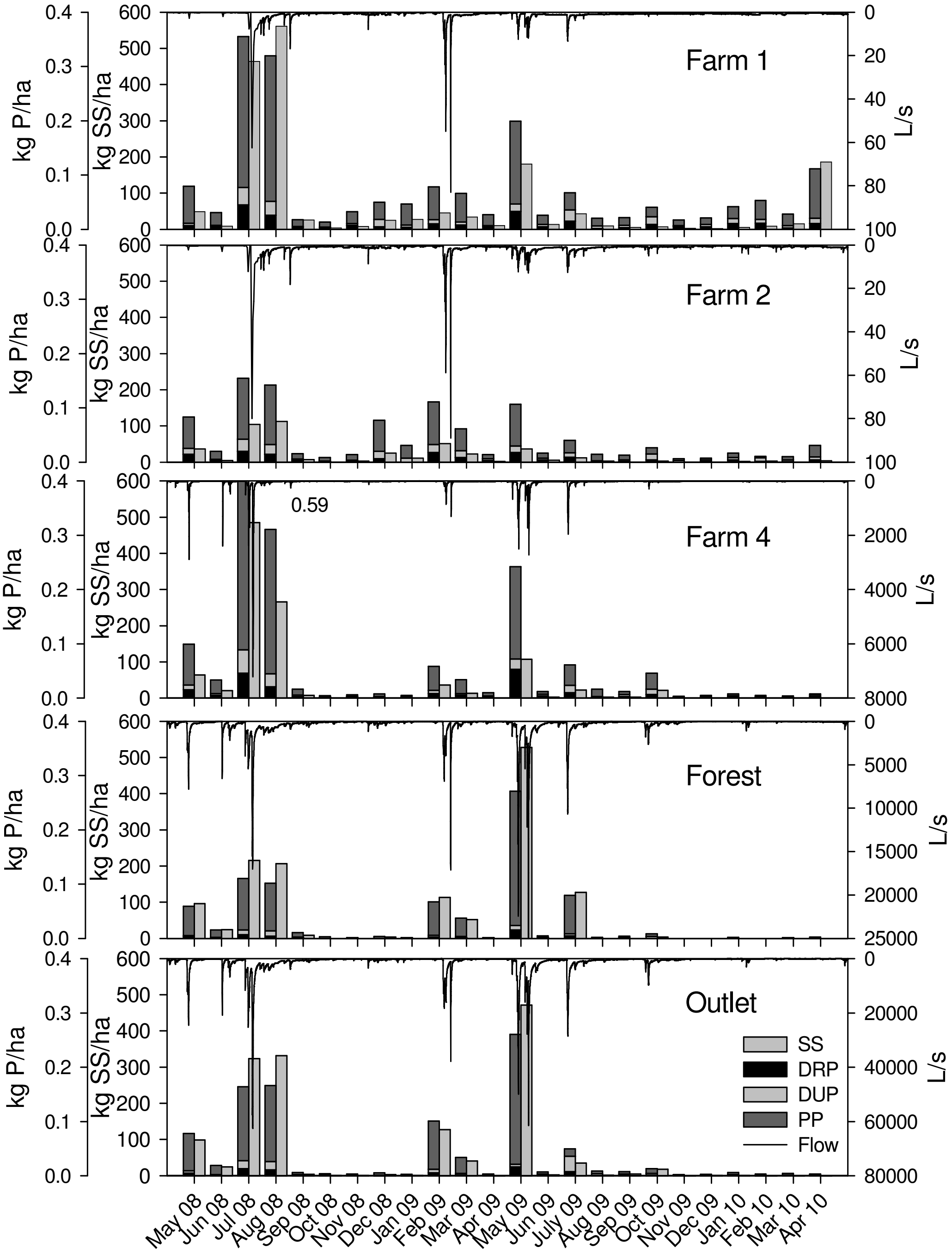

Fig. 19 Export coefficients (kg P/ha) of dissolved reactive phosphorus (DRP), dissolved unreactive P (DURP), particulate P (PP) and suspended sediment (SS) for each site by month (2008 – 2010). The 0.59 value refers to the load of P in July 2008.

4.3.3 Influence of sediments

During base flow, none of the measured sediment EPC_0 concentrations were in equilibrium (± 20%) with DRP concentrations measured at the same time, nor was there any relationship between EPC_0 and DRP concentrations with time or between sites (Table 17). However, the fact that the EPC_0 measured was much greater than the DRP measured indicates a potential release of P from sediments to water. The absence of equilibrium between sediments and stream P concentrations was also found in an earlier investigation (Chapter 3) and could be due to in-stream sediment residence being too short to reach equilibrium (Jarvie et al., 2005) with the overlying water column. The highest measured EPC_0 was found in the sediments at the farm 2 site (1.719 mg/L) which concurs with the study by Stutter et al. (2008) where sediments from headwaters were more enriched in P, while further downstream biological processing decreased EPC_0 concentrations. In addition, the EPC_0 of sediment from the farm 2 site was found to increase with increasing base flow rates (data not shown), which supports the hypothesis that at high base flows the sediment is composed of newer, P enriched sediment.

Table 17 Equilibrium P concentration at zero net sorption and desorption (EPC0) and dissolved reactive P (DRP) concentrations measured at different sites and at different dates during base flow.

Date	Farm 2		Forest		Bridge	
	EPC_0	DRP	EPC_0	DRP	EPC_0	DRP
			(mg/L)			
16/04/2009	1.053	0.017	0.043	0.007	0.023	0.005
30/10/2009	-[1]	-	0.025	0.010	0.112	0.005
3/02/2010	0.187	0.014	0.093	0.007	0.108	0.005
13/05/2010	1.719	0.012	0.103	0.002	0.051	0.002

[1] data not available

Measurements of EPC_0 were made on SS collected at each site during a storm event on September 29, 2009 (Table 18). The fact that EPC_0 concentrations were 6-26 times greater than measured DRP shows that there is potential for substantial desorption of P into the surrounding water during storm flow. When the EPC_0 was weighted to take account of the concentration of SS in the water, a significant (r^2=0.48, P<0.05) relationship was found with DRP concentration (DRP = 0.9096 (EPC_0 * SS) + 0.004) suggesting a role for SS in determining DRP in storm flow. Sharpley & Syers (1979) estimated that 33% of the annual DRP load from another New Zealand stream was released from suspended material in the stream.

Table 18 Equilibrium P concentration at zero net sorption and desorption (EPC0) of suspended sediment taken from stream during a storm event (29/9/2009).

Site	EPC	DRP (mg/L)	SS
Farm 2	0.481	0.018	12
Farm 4	0.221	0.033	107
Forest	0.134	0.005	30
Bridge	0.112	0.008	58
Outlet	0.187	0.009	95

4.3.4 Scale and connectivity

To assess the connectivity and changes between scales and sources, linear regressions were made between sites using the concentrations of DRP, PP, TP and SS samples taken on the same day (within 2 hours) (Tables 19 & 20). The matrices show the degree of association between near and far sites in a nested design under different flow rates. Measured DRP concentrations from farm sites 1-4 (5-10 ha) were related during both storm and base flow, and in most cases the significance (and r^2) increased during storms. Haygarth et al. (2005) also found that connectivity, as evidenced by greater DRP concentrations, was greatest at scales < 20 ha. The significance between sites was similarly greater for PP and SS for farm sites 1-3 during storm flows than base flows. In general, the closer the site was to the outlet, the higher the likelihood that there would be a significant relationship in P or SS fractions, especially for PP and SS during storm flow. For example, farm 1 was not related to the outlet, but other farm sites were, with the relationship between farm 4 and the outlet being highly significant for storm flow (r^2 = 0.96, P<0.001; Table 19).

Significant relationships between the forest, outlet and bridge sites were found more often during base flow than storm flow for DRP and PP, evidence of consistent in-stream processes at work during low flows. However, during storm flow, differences in topography and land use which influence erosion processes, lead to dissimilar rates of runoff and erosion (Tables 19 & 20). The exception was between the bridge and the outlet. Between these sites the stream channel is uniform with similar hydraulic energy, while the surrounding flat land contributes little direct runoff. During base flow there were significant relationships between the farm 4 and forest sites, which were not connected, but may provide evidence of similar processes at work.

Table 19 The r^2 values and significance of linear regressions between the given sites for DRP, PP and TP samples collected during base and (storm) events on the same day.

	Farm 1	Spring	Farm 2	Farm 3	Farm 4	Forest	Bridge
				DRP			
Spring	8.5* (51*)	-					
Farm 2	51*** (74**)	9* (54*)	-				
Farm 3	21** (80***)	ns (ns)	60*** (73**)	-			
Farm 4	44*** (60*)	11* (ns)	73*** (84**)	52*** (ns)	-		
Forest	9* (ns)	ns (ns)	ns (ns)	ns (ns)	10* (ns)	-	
Bridge	ns (ns)	ns (ns)	ns (ns)	ns (ns)	ns (ns)	23*** (ns)	-
Outlet	17** (ns)	ns (ns)	17** (ns)	34*** (ns)	12* (ns)	ns (ns)	ns (62**)
				PP			
Spring	9* (ns)	-					
Farm 2	38*** (73**)	ns (ns)	-				
Farm 3	23*** (44*)	ns (ns)	54*** (71**)	-			
Farm 4	ns (97***)	ns (ns)	ns (92**)	ns (99***)	-		
Forest	8* (ns)	ns (ns)	ns (72*)	15** (ns)	21** (ns)	-	
Bridge	ns (ns)	ns (ns)	ns (ns)	ns (ns)	37*** (96***)	17** (ns)	-
Outlet	ns (ns)	ns (ns)	ns (45*)	26*** (ns)	ns (95***)	27*** (62*)	10* (90***)
				TP			
Spring	33*** (ns)	-					
Farm 2	49*** (80***)	45*** (ns)	-				
Farm 3	27*** (61**)	46*** (ns)	80*** (81***)	-			
Farm 4	19** (98***)	ns (ns)	28*** (93**)	20** (99***)	-		
Forest	11* (ns)	ns (ns)	14** (76*)	14** (ns)	33*** (ns)	-	
Bridge	ns (ns)	ns (ns)	ns (ns)	ns (ns)	33*** (99***)	25*** (ns)	-
Outlet	ns (ns)	27*** (ns)	24*** (59*)	48*** (51*)	19** (96***)	32*** (66*)	14** (91***)

Table 20 The r^2 values and significance of linear regressions between the given sites for SS samples collected during base and (storm) events on the same day.

	Farm 1	Spring	Farm 2	Farm 3	Farm 4	Forest	Bridge
			1.	*SS*			
Spring	ns (ns)	-					
Farm 2	ns (66**)	ns (ns)	-				
Farm 3	ns (47*)	ns (ns)	48*** (95***)	-			
Farm 4	ns (66*)	ns (ns)	ns (98***)	ns (99***)	-		
Forest	ns (ns)	ns (ns)	ns (ns)	ns (ns)	87*** (99***)	-	
Bridge	ns (ns)	ns (ns)	ns (ns)	ns (ns)	20** (96***)	20** (91**)	-
Outlet	ns (ns)	ns (ns)	ns (51*)	ns (49*)	9* (99***)	52*** (89**)	8* (69**)

Total P at the outlet represents the integral of processes and P fractions at all scales and land uses. It was possible to estimate the average change in TP concentration from the smallest farm 1 catchment down to the outlet by forcing the intercept of the regression between connected sites through zero (without significantly decreasing the coefficient of determination) and comparing the slope (Figure 21). For storm flow, TP concentration decreased by 32% between farm 1 and 2, commensurate with a decrease in slope (Figure 21) and the presence of a riparian area in the stream channel (McDowell, 2008b). From farm 2 to 3 there was an increase in TP (19%), commensurate with access to the stream channel by grazing animals (McDowell & Srinivasan, 2009), but still a net decrease from farm 1. From farm 3 to farm 4, a distance of just over 1km, there was little change in TP concentrations despite an increase in slope, but this could be due to a shift from grazed pasture to exotic forest around the stream channel. Where the farm catchment joins up with the main catchment at the bridge, there was a 15% decrease in TP which is likely to be the result of dilution from forest flow. However, between the bridge and the outlet, there was a further 14% increase in TP during storm events which may be related to the artificially straightened channel between these sites (Figure 18).

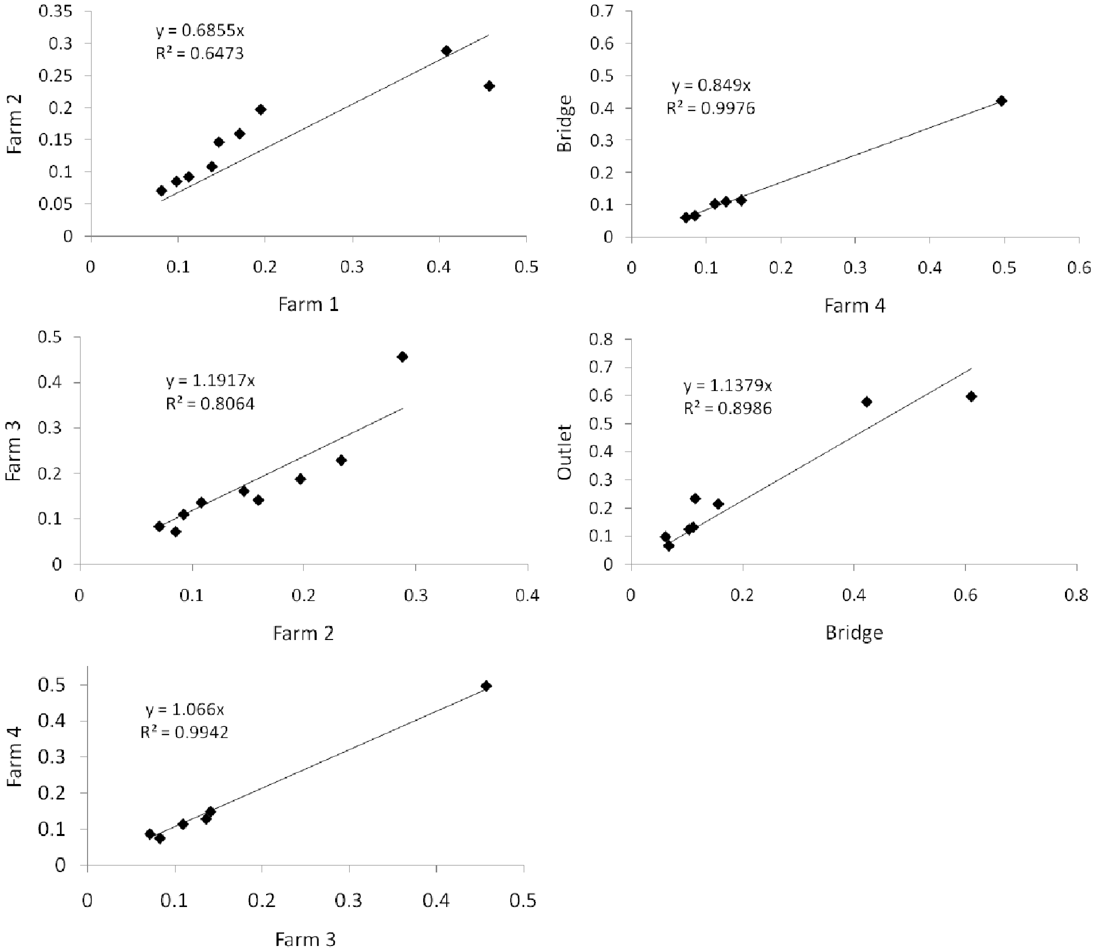

Figure 21 Linear regressions of total P (mg/L) between given sites measured during storm events. The scale of the sites increases from top to bottom by column. The regressions were forced through the origin with no significant decrease in r^2 values.

Another way of plotting the change in mean concentrations is shown in Figure 22 for base flow DRP concentrations. This data shows that enriched concentrations from the farm sites were diltuted by low DRP water from the forest site. Sharpley & Tunney (2000) measured dissolved P concentrations at the catchment outlet that were much less than those in measured overland flow. They attributed this reduction to dilution by subsurface flow or in-stream processes. The data from this study would suggest that during base flow, DRP concentration at the outlet is controlled by the supply of DRP from the forest site (itself a function of low soil P and good contact time with draining water) with little connectivity to the pasture-dominated farm sites. However, connectivity, and to some degree control of DRP concentrations, during storm flow occurs at the farm sites via the loss of SS and its effect on EPC_0. Therefore, although during base flows the contribution of P from the small scale farm sites is minimal, a considerable amount of dissolved and particulate P may originate from these areas during storm flows which highlight the need to manage fertilizer and soil P levels.

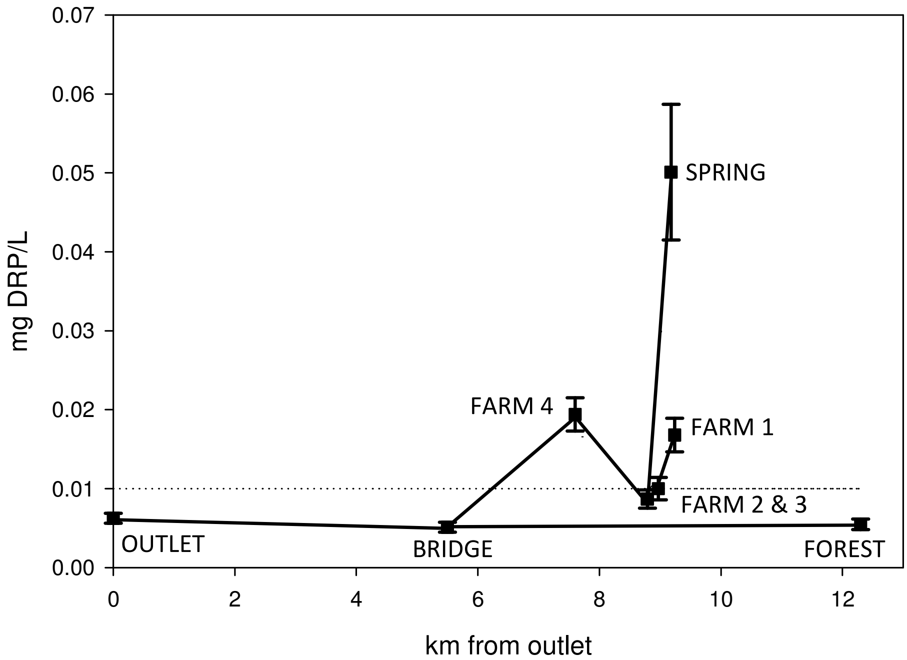

Figure 22 Mean base flow dissolved reactive P (DR) concentration at each site. The connecting lines show which sites were hydrologically connected. The dotted line represents the ANZECC (ANZECC-ARMCANZ, 2000) trigger DRP concentration in slightly disturbed New Zealand lowland streams (0.010 mg DRP/L).

In this catchment the majority of the P load is PP. It is uncertain how quickly PP becomes available, but there are numerous places for it to settle out and become part of the sediment where it then has the potential to become a source of P to stream biota. The exclusion of stock from all waterways in the grazed catchments is one way to eliminate sediment and PP loss from treading, as well as direct faecal deposits in the stream. However, grazed grasslands have been shown to export a high level of sediment from overland flow, even after stock access to streams is excluded (Bilotta et al., 2010). This highlights the need for appropriate grazing management strategies that restrict the hard grazing of pastures, leaving more cover to intercept particles in runoff and protect the soil from raindrop impacts. When the soil is saturated and sensitive to damage, stock should be kept on well drained paddocks distant from wetlands and waterways. For dairy or beef cattle, purpose-built standoff pads or herd homes may be a sensible long term investment (Longhurst et al., 2006). The high sediment loads from the forest are a result of the steep terrain (Figure 18) and high erosive energy. Forest roads are also known to contribute to sediment loading from forestry operations (Fransen et al., 2001), but few of these roads are present in the forested catchment here as it is predominantly native bush.

4.4 Conclusions

The results from this chapter have shown that P and sediment measured at small scales do not reflect catchment scale trends. Catchments where pastoral farming is combined with less intensive land use resulted in lower total P loading. However, the concentrations were still great enough in the summer to potentially trigger the growth of toxic algal blooms. A portion of this P may originate from DURP released from feces and from wetting and drying events during summer. The bed and suspended sediment examined here highlighted the potential for P desorption during storm and base flows, but it is still unclear how much of a contribution sediments make. A greater proportion of P, particularly dissolved P, was lost from the smaller scale sites at base flow, while storm flow derived particulate P dominated loads at larger scales. Evidence of connectivity was apparent at small scales (<10 ha) during base flow and storm events. However, at larger scales the different combinations of sources, hydrology and in-stream processes restricted connectivity between sites to storm events. The processes for particulate P and SS transport, and the potential control of dissolved P via desorption, were better aligned during storm flow than at base flow. This focuses management options for decreasing P in the stream towards those that minimize sediment and particulate P loss, particularly fencing off the stream from animal disturbance.

Chapter 5 - Bayesian Network for point and diffuse source phosphorus transfer from dairy pastures

This chapter combines the results of the previous experiments to help meet the final objective of the thesis: to develop rules for assessing P losses and connectivity in different landscapes. This chapter describes the development of a Bayesian Network (BN) that combines knowledge from previous experiments, literature and expert opinion. This network was used to understand the important sources and transport processes operating in a south Otago dairy catchment, and the management strategies that are most effective in mitigating TP loading to streams.

5.1 Introduction: Bayesian Networks

As previously mentioned, the mechanisms for the export of P have been studied extensively and much is already known (Sharpley, 1985; Simard et al., 2000; Wood et al., 2005). However, the combination of sources, site specific effects, and changing weather and transport pathways means that the integration of source area knowledge and transport mechanisms is challenging. This is evidenced by the number of studies which have shown a poor correlation between measurements made at the plot scale and at larger scales (Haygarth et al., 2005; Jordan et al., 2005; Wood et al., 2005). In spite of years of research, much uncertainty remains.

One way of dealing with uncertainty is to use a Bayesian Network which graphically represents complex interactions within systems. BNs are an alternative to conventional modelling that accommodate spatial and temporal variation (Pearl, 1988) in complex multi-factor problems (Varis, 1997; Varis & Kuikka, 1999; Ames & Neilson, 2001; Ames, 2002). These networks represent uncertainty in knowledge as a probability distribution and use conventional probability theory to manage that uncertainty. The conditional dependencies between the different knowledge components (prior probability distributions or a priori beliefs) are explicitly represented in the conditional probability tables that underpin the network and are always retained. Therefore, in a BN the exact values of the variables do not need to be known, and a range of probable values is acceptable and, in some cases, more realistic. When observational data (i.e. evidence) are added to one or more nodes (i.e. variables) in a network, the a priori beliefs are modified through belief propagation to yield posteriori beliefs. This enables BN to be used for both forwards and backwards reasoning, with the output commonly represented in an intuitive graphical visualization of the system (i.e. a cause and effect diagram). Consequently, BN provides a natural framework for the inclusion of expert opinions and thereby a natural framework for updating parameters as more evidence and new knowledge becomes available. This approach has been applied in modelling complex ecosystem interactions, such as the bleaching of coral reefs

(Wooldridge & Done, 2004), and in more practical applications, such as deciding which irrigation systems would be most appropriate (Robertson & Wang, 2004). They have also been shown to be good at comparing management options to mitigate N (Nash et al., 2010) and P losses (McDowell et al., 2009b).

The flexibility of BNs is particularly useful for modeling farming systems where the complexity of the interactions may preclude the collection of large data sets upon which parameterization and testing of a conventional model could be based. Each step or node in a BN is visible, all nodes are interdependent and the propagation of each decision can be seen throughout the model. The straightforward nature of BNs makes model validation relatively simple, though not necessarily rigorous, especially in data poor environments.

There are, however, some limitations of BNs which need to be understood before working with or using them. The predictions made by a BN are meant to aid in decision-making, rather than to actually make the decisions. Restrictions on network size and on the construction of complex conditional probability tables (primarily as a result of computer storage) are a disadvantage of BNs but can be addressed using Monte Carlo type methods where deterministic relationships are defined. However, another limitation on BN use is the need to convert the continuous values of the variables into discrete intervals (i.e. discretise) (Smid et al., 2010), and determining the most appropriate discretisation is rather an ad hoc process (Castelletti & Soncini-Sessa, 2006). Temporal and spatial relationships are difficult to capture in a BN and therefore a specific time and scale must be decided on before model development.

In this study, a BN of P exports for a hypothetical dairy farm in the south Otago region of New Zealand has been developed and tested. The network is used to integrate previous work and investigate the effects of different management options under contrasting rainfall and drainage regimes.

5.2 Methods

5.2.1 Network development

In this study NETICA® (Norsys Software Corp., Vancouver, Canada) software was used for developing the BN. The network was developed to represent annual total P exports from a typical dairy farm in the south Otago region of New Zealand. The south Otago region was selected because of its importance to dairying (73,000 t milk solids/yr) (LIC, 2010) and proximity to important environmental assets such as lakes, rivers and wetlands (e.g. Clutha and Tokomoairiro Rivers, Lake Tuakitoto and associated wetlands).

The process for network development outlined by Nash et al. (2010) was used as the basis for this study. In summary, a literature search was conducted to collate research data relating to total P losses from dairy farming systems. From these data, and interviews with industry specialists, the most important sources of P were identified, as well as mobilization and transport processes from these sources. With this knowledge a preliminary "cause-and-effect" diagram was created to describe the key P sources and transport processes. The diagram was presented to a specially convened meeting of five researchers familiar with dairy farm systems who specialized in P losses, water quality, dairy effluent management and soil physical properties. Much of the discussion was around the partitioning of overland and subsurface flow, and the accompanying effects on P transport. There was also feedback on the handling of dairy effluent which led to changes in the structure of the network.

Once comments from the specialist workshop had been incorporated, the "cause-and-effect" diagram consisted of nodes which represented variables. These nodes contained preliminary states, which were the initial values it was believed a particular node could take, and the nodes were linked together with arrows to indicate causality. Published literature was again consulted to locate specific experiments and data with which to quantify relationships between the variables. Preference was given to data and relationships derived from the Otago region but, by necessity, in many instances information from the other regions was used. Where possible, deterministic relationships (i.e. equations) were used to develop the conditional probability tables (CPTs) that underlie the BN using the "Equation to Table" function in Netica. Continuous Nodes (i.e. variables) were discretised with the States revised appropriately. Where deterministic relationships could not be found, expert opinion was used to populate the CPTs using the best current understanding of processes. This was the case for the following nodes: Slope, Track Design and Drainage Class.

Once the CPTs were complete the network was considered functional. As there were no data which examined all of the factors (i.e. nodes and states) in the network at the correct scale, initial validation was undertaken by comparing the network output to the limited experimental data that were available (Table 21). It is of note that data from the North Island was included in this process because of the lack of South Island data from effluent loads. In each case TP exports estimated from the network were within the ranges of the values in Table 21. For example, the network suggested the loads of 0.772 (± 1.1) kg TP /ha for overland flow and 1.36 (± 1.3) kg TP/ha for subsurface flow, in line with the overland flow (0 to 0.98 kg TP/ha) and drainage results

(0.9 and 2.2 kg TP/ha) in Table 21. As an additional step in the validation process the network was presented to five researchers specializing in nutrient exports from dairy farming systems. In each case the network was found to reasonably approximate expected system behavior. In some cases, where information was either deficient or conflicting, additional physical experiments were undertaken.

Table 21 Ranges of reported TP loads in drainage and overland flow measured at the plot and field scale, as well as catchment loads from four catchments of varying size and land use.

Management	Location & soil type	Load (kg/ha)	Reference
		Drainage	
No effluent	Edendale & Tapanui pallic and brown soils	0.15-1.2	(Monaghan et al., 2002; Monaghan & Smith, 2004; McDowell et al., 2005a)
Effluent applied	Tapanui Pallic	0.9-2.2	(Monaghan & Smith, 2004; McDowell et al., 2005a)
		Overland flow	
Drained	Edendale & Hillend pallic soils	0-0.05	(Monaghan et al., 2000; Smith & Monaghan, 2003)
Undrained	Edendale & Hillend pallic soils	0-0.98	(Monaghan et al., 2000; Smith & Monaghan, 2003; McDowell et al., 2006) + own data
Drained with effluent applied	Massey tokamaru	1.1	(Houlbrooke et al., 2004)

The final BN is presented in Figure 23. A complete description, and the references of each node, is presented in Appendix I. A brief description of each node is also given in the following section.

5.2.2 Network description

The variables used in the network were divided into five categories after Nash et al. (2010): 1) Management Variables are practices which are under the control of the land manager; 2) Site Variables are the physical attributes of the site which are not under the control of the land manager (e.g. slope, soil type); 3) Year Variables depend on chance events like weather, while; 4) Intermediate variables combine the effects of the Management, Site and Year Variables using either empirical relationships or expert knowledge, and; 5) The Outcome Variables estimate the ultimate effects of various scenarios on P exports. This division was not always clear as some variables fit into more than one category. For example, Olsen P is categorized site variable but could also be seen as a management variable since it can be manipulated with management over time. Therefore, it should be understood that there is some flexibility around these categories.

Table 22 Description of assumptions used for key nodes. For a more complete description of node and network structure refer to the supplementary information in Appendix I.

Node	Assumption
Sub-Surface Flow	Drainage system sets drainage to improve drainage by two classes.
Deferred Application	Deferred Application means that there is adequate storage to practice Deferred Application, and storage variable is not needed
Feed pad or Herd home	That effluent from Feed pad and Herd home is roughly the same and can be calculated in the same way.
Paddocks with access	Based on a 30 day grazing rotation and 20 hrs spent in paddock (4 milking), over 10 months of the year (i.e. # of paddocks *10 months)
Additional winter loss	10% of the farm is put into crops, crops are grazed 2 months out of the year
Background TP (mg/L)	if Olsen P < 8 mg/kg then background TP is 0.01 mg/L
Surface Diffuse Load (kg/ha)	Incidental fertilizer load calculated assuming risk in one month and therefore uses 1/12 of the runoff (average % of annual rainfall in OCT-NOV, but rainfall is generally evenly distributed throughout the year so should be appropriate for each month)
TP Load Track (kg/ha)	Assumes that 0.5% of farm area is covered by lanes and total runoff is used because lanes and tracks are impervious. Poor track design means 25 -50% of lane area will contribute, fair means 0-25% will contribute and Good means 0% will contribute. Load also adjusted for stocking rate, light means half as much, while heavy means 50% more than average stocking rates.
Effluent Load	Effluent loss is partitioned between overland and subsurface loss according to drainage class

In a previous network describing P exports from dairying in Australia, the number of parent nodes and states for each node were generally limited to three (McDowell et al., 2009b). This was necessitated by the process of expert opinion that was used to develop the many CPTs. The highly deterministic nature of the network in this study lent itself to more than three states for each node and therefore increased sensitivity compared to previous networks. In addition, wherever possible ranges of values were used instead of discrete values to include the uncertainty of states. Importantly, the states and state values were discretised to reflect the associated deterministic equations and effects on subsequent estimates. Structurally this network is also different to that developed previously. These differences are most notable for effluent management, which is treated as one of three diffuse P sources, rather than a point source. In the network by McDowell et al. (2009b) the application of effluent is assumed to have no impact on subsurface P loadings, contrary to some studies (Monaghan & Smith, 2004; McDowell et al., 2005a), while in this network the effluent load is partitioned between surface and subsurface loads. Structurally, this enabled the effects of different effluent management strategies to be investigated more thoroughly

5.2.2.1 Transport

Annual Rainfall (mm*)* is the root node used to estimate the annual volume of water available for the transport of P. Total Runoff (mm) was estimated from a linear regression using the annual rainfall and the runoff calculated using the water balance model (i.e. bucket model as described on NIWAs website http://cliflo.niwa.co.nz/) for 30 years of data from three weather stations around the south Otago Region (Balclutha, Lovells flat, and Baverstock) from the National Climate Database (NIWA).

Total Runoff is partitioned between Overland Flow (mm) and Subsurface Flow (mm) conditioned on Amended Drainage Class (Hewitt, 1993). If Artificial Drainage is present the drainage class is moved two classes toward Well Drained, based on overland flow data from drained and undrained plots. Smith & Monaghan (2003) measured runoff from a poor to imperfectly drained soil in Southland, with and without artificial drainage. The overland flow during the measurement period averaged 46.5 mm, while the average from the drained plots was 9.4 mm. If it is assumed that 50% of runoff is overland flow (according to the classification used here) then the total runoff is 93mm. The measured overland flow in the artificially drained plot was reduced to 10% of the total runoff which equates to a change of two drainage classes (i.e. Well Drained).

5.2.2.2 Diffuse sources

The export of TP in this network is divided into diffuse and point sources. The point sources feed directly into the Total Load, while the diffuse sources are partitioned between overland and subsurface flow before being summed in the Total Potential Load (kg TP/ ha/yr).

The diffuse load was deemed to have originated from three sources. The Background Soil Load (kg/ha/yr*)* from soil was derived using relationships developed between the Olsen P of the three main soil types in south Otago, and TP concentrations in overland flow (McDowell & Condron, 2004). In the absence of appropriate field data, these relationships were used despite having been derived using simulation rainfall on bare, repacked soil trays. Phosphorus mobilization direct from plants (and through the effects of treading on plants) and faeces were estimated to be 0.3 mg/L (McDowell et al., 2007) and added to the soil P background concentration. It is of note that similar studies using soils from elsewhere in New Zealand show similar results, e.g. Sharpley & Syers (1976) found an increase of 0.2 mg/L from grazing.

The incidental fertilizer load (Incidental Fert Load, kg/ha/yr) depends on the interval between fertilizer application and runoff. Both field and simulation studies suggest that this relationship can be represented by a power function (McDowell et al., 2003b; Nash et al., 2005). The following equation incorporates both the interval between fertilizer application and runoff, and fertilizer application rate (McDowell et al., 2003b):

Incidental Fert (mg/L) =

$$(1.07 \ast \text{Fert Application Rate} - 10.3) \ast \text{Days between application and runoff}^{(-0.0076 \ast \text{Application Rate} - 0.55)}$$

It is assumed that the risk of P loss from fertilizer is only in the overland flow component. This is supported by a lysimeter study by McDowell (2008a) who found no significant difference between leachate from soils with no fertilizer versus soils that received 30 kg P/ha. Phosphate fertilizer applied in excess of plant requirements raises the Olsen P in the soil and this is covered in the subsurface component of the network. Phosphate fertilizer is usually applied by dairy farmers in October or November. The concentration of TP calculated by the equation above is then multiplied by the average percentage of overland flow in October or November (5% of the annual runoff).

The Winter ON or OFF management variable specifies the wintering strategy for the farm. Three options have been included in this network which cover the main wintering strategies in this region: wintering off (OFF Farm), wintering on but using a standoff pad or similar (e.g. Herd Home®) to minimize treading damage by stock (ON Standoff Pad), or wintering on using a forage crop such as *Brassicas* (ON Crop). Wintering off adds no further inputs over the winter period. Wintering on with crops is calculated in Additional Winter Loss (kg/ha): seven times the Background Soil Load if the Total Runoff is greater than 300mm, three times as high as the Background Soil Load if the runoff is between 100 and 300 mm, and zero if the Total Runoff is less than 100 mm (McDowell, 2006b; McDowell et al., 2006; McDowell & Houlbrooke, 2009). This load is then divided by 6 as it is assumed that the wintering will take place for two months out of the year, and divided by 10 assuming that 10% of the farm is cropped (McDowell & Houlbrooke, 2009).

The Total Effluent Load (kg/ha) is the sum of the Dairy Shed Effluent (kg/ha) and Standoff Pad Effluent (kg/ha). Theses loads are based on the Stocking Rate (cow/ha), the Lactation Days and the Standoff Pad Use (weeks) during the year, with the volume of effluent produced (Vanderholm, 1984) and the concentration of TP in effluent (Longhurst et al., 2000; Longhurst et al., 2006):

Dairy Shed Effluent = 50L/cow/day * 70 mg/L * Stocking Rate * Lactation Days

Standoff Pad Effluent = 0.01 kg TP/ cow/day * Stocking Rate * Standoff Pad Use.

The effluent irrigation management options in the BN are: the percentage of the farm in which effluent is spread (Effluent Area (%)); the Application Intensity (or instantaneous application rate; mm/h) of the irrigated effluent; and the use of Deferred Application (Houlbrooke et al., 2004). The size of effluent storage (days) was omitted because if deferred irrigation is used, it is assumed that there is a pond large enough for storage. In the network, the practice of Deferred Application results in annual losses of 3% of the Total Effluent Load (Houlbrooke et al., 2003; Houlbrooke et al., 2008). If Deferred Application is not used then the load is 60% of the Total Effluent Load (McDowell et al., 2005a); however, a further reduction can be made if low rate effluent spreaders are used (K-line, 4 mm/hr) to half the TP lost (30% of Total).

It has been shown that significant amounts of P can be lost in both subsurface and overland flow from effluent applied to land. Therefore the Effluent Load was partitioned between the Subsurface Diffuse Load and the Overland Diffuse Load according to the proportion of subsurface and surface flow using the amended drainage class. This was after Houlbrooke *et al.* (2004) who found the proportion of overland and subsurface loading was similar to the proportion of overland and subsurface flow.

5.2.2.3 Point sources

Stock access (kg/ha/yr) to streams was estimated using a study which counted the number of dung deposits made by cows in an unfenced stream in a paddock (James et al., 2006). This was then expressed as the number of deposits per cow per hour, and the amount of P per deposit was calculated from the average weight and concentration of TP in a dung pat (Davies-Colley et al., 2004; McDowell et al., 2006). To upscale this paddock measurement to the whole farm, a node was created specifying the number of paddocks on the farm with stream access (paddocks with access). For simplification it was assumed that the farm was on a 30 day rotation and that each paddock would be used for 20 hours (4 hours for milking), once a month for 10 months:

Stock access = 0.03 dung/hr/cow * 20 hr * 7.8 g TP/ kg * 0.92 kg/ dung * Stocking Rate * Paddocks with access* 10 (4)

The load from streams and tracks (TP Load Track (kg/ha/yr)) was calculated using the average runoff concentrations from a lane in the selected region (5.5 mg/L Chapter 2). The assumptions used here were that 0.5% of the farm area was covered by lanes. The node Track Design is a modifier of the load where Poor track design means that 100% of the lane will contribute due to poor maintenance or design and direct drainage to waterways. Fair track design means that half of the load will enter waterways, while Good means that all drainage is diverted away from waterways (McDowell et al., 2009b). The total runoff was used to calculate the load because previous lane measurements have shown infiltration rates of < 0.01 cm^3/h (Chapter 2) indicating almost all of the runoff will be as overland flow.

5.2.2.4 Output variables

The sum of the inputs from Incidental Fert Load (kg/ha/yr), Background Soil Load (kg/ha/yr*)* and Effluent Load make up the Overland Diffuse Load (kg/ha/yr). The Subsurface Diffuse Load (kg/ha/yr) is the sum of the SS Background Soil Load (kg/ha/yr) and the Effluent Load. These diffuse sources were then summed to arrive at the Total Potential Load (kg/ha/yr*)* which is, as the name implies, the total load of P available for transport. This load was then modified by the site variable Slope*,* which gives steep slopes more weight than undulating landscape (McDowell et al., 2005c). The point sources were added directly to the Total Load (kg/ha/yr) because direct deposition by cows is not modified by catchment slope and lane runoff (TP Load Track) is already conditioned with the Track Design node. High loads estimates will be biased due to curve linear relationships between nodes. The higher values (> 5 kg/ha/yr) can be considered as “high” rather than the actual estimate.

5.3 Application and analyses of the network

A sensitivity analysis of the Overland and Subsurface Diffuse Load nodes, as well as the final Total Load node, was the first step of analyzing the network. This analysis is built into the Netica software used to construct the model (Pearl, 1988). The results of the analysis are presented in terms of Variance Reduction. The Variance Reduction is a measure of the difference in variances of the target node before and after the value of a linked node (one that is connected in some way with the target node) is known. The sensitivity of a linked node decreases with distance from the target node due to the intermediate variables. The sensitivity analysis includes all of the nodes that are linked to the target node in any way, even indirectly through common intermediate variables, going both with and against the direction of the linking arrows (Figure 24). For example, the point source node Stock Access is not directly linked to the Subsurface Diffuse Load, but since both parts of the network use Stocking Rate, Stock Access is part of the variance reduction analysis for Subsurface Diffuse Load.

Table 23 Results of the sensitivity analysis performed for the Total Load. Site, Year or Management variables are in bold.

Node	Variance reduction
Total_Load	2.6450
Total_Potential_load	1.4940
Subsurface_Diffuse_Load	0.8419
Overland_Diffuse_Load	0.6057
Effluent_Load	0.5503
Slope	0.4181
Deferred_Application	0.3793
SS_Background_Soil_Load	0.2979
Background_Soil_Load	0.2968
Background_TP	0.2068
Total_Runoff	0.1436
Annual_Rainfall	0.1382
Overland_Flow	0.1167
Olsen_P	0.1063
Fertiliser_Load	0.1002
Dominant_Soil_Type	0.0767
Sub_Surface_Flow	0.0580
Total_Effluent_Load	0.0563
Standoff_Pad_Effluent	0.0297
Dairy_Shed_Effluent	0.0275
Additional_Winter_Load	0.0274
TP_Load_Track	0.0266
Stocking_Rate	0.0201
Winter_ON_or_OFF	0.0190
Amended_Drainage_Class	0.0181
Incidental_Fert	0.0119
Application_Intensity	0.0107
Artificial_Drainage	0.0078
Primary_Soil_Drainage_Class	0.0055
Days_between_application_and_runoff	0.0050
Stock_Access	0.0050
Paddocks_Access	0.0037
Fert_Application_Rate	0.0026
Effluent_Area	0.0013
Standoff_Pad_Use	0.0004
Lactation_Days	0.0003
Track_Design	0.0000

Table 24 Results of the sensitivity analysis performed for the Overland Diffuse Load and Subsurface Diffuse Load. Site, Year or Management variables are in bold.

Node	Variance reduction	Node	Variance reduction
Overland_Diffuse_Load	1.0600	Subsurface_Diffuse_Load	1.7140
Background_Soil_Load	0.6131	Total_Potential_load	0.9913
Overland_Flow	0.4541	Effluent_Load	0.6960
Total_Potential_load	0.4093	SS_Background_Soil_Load	0.6589
Fertiliser_Load	0.3278	Total_Load	0.5495
Amended_Drainage_Class	0.3019	**Deferred_Application**	0.4369
Total_Load	0.2292	**Sub_Surface_Flow**	0.2503
Artificial_Drainage	0.1729	Background_TP	0.2234
Effluent_Load	0.0888	Amended_Drainage_Class	0.1278
Primary_Soil_Drainage_Class	0.0874	**Olsen_P**	0.1233
Deferred_Application	0.0519	**Total_Runoff**	0.1215
Background_TP	0.0516	**Annual_Rainfall**	0.1175
Total_Runoff	0.0480	**Dominant_Soil_Type**	0.0895
Annual_Rainfall	0.0454	Overland_Diffuse_Load	0.0863
Sub_Surface_Flow	0.0406	Total_Effluent_Load	0.0827
Additional_Winter_Load	0.0372	**Artificial_Drainage**	0.0823
Incidental_Fert	0.0280	Standoff_Pad_Effluent	0.0463
Subsurface_Diffuse_Load	0.0226	Dairy_Shed_Effluent	0.0373
Olsen_P	0.0224	**Primary_Soil_Drainage_Class**	0.0341
Dominant_Soil_Type	0.0156	**Winter_ON_or_OFF**	0.0298
Days_between_application_and_runoff	0.0117	**Overland_Flow**	0.0293
Total_Effluent_Load	0.0113	**Stocking_Rate**	0.0266
TP_Load_Track	0.0069	TP_Load_Track	0.0229
Standoff_Pad_Effluent	0.0065	**Application_Intensity**	0.0147
Fert_Application_Rate	0.0058	Fertiliser_Load	0.0102
Dairy_Shed_Effluent	0.0049	Background_Soil_Load	0.0025
Winter_ON_or_OFF	0.0042	**Effluent_Area**	0.0010
SS_Background_Soil_Load	0.0038	**Standoff_Pad_Use**	0.0006
Stocking_Rate	0.0035	**Lactation_Days**	0.0005
Application_Intensity	0.0019	Additional_Winter_Load	0.0004
Effluent_Area	0.0004	Stock_Access	0.0001
Standoff_Pad_Use	0.0001		
Lactation_Days	0.0001		

The calculated intermediate loads (Total Potential Load, Subsurface Diffuse Load, and Overland Diffuse Load) accounted for most of the variance reduction in the Total Load (Table 23). Site and Year transport-related variables like Slope, Annual Rainfall and Total Runoff were also important in reducing the Total Load variance, which is reasonable considering the variation in rainfall and runoff are the greatest sources of load variation in actual systems. The management variable responsible for the greatest reduction in variance for the Total Load was Deferred Application. The least reduction in variance came from supplying information to the Track Design node. This is probably due to the fact that the most probable loads calculated from the TP Load Track are so small that conditioning the loads with the Track Design make little difference.

A sensitivity analysis was also performed on the Overland Diffuse Load and the Subsurface Diffuse Load (Table 24). The most important sources of variance in the Overland Diffuse Load were the Overland Flow along with the related Artificial Drainage and Primary Drainage Class. Olsen P and Dominant Soil Type were more important in reducing the variance of the Subsurface Diffuse Load than the Overland Diffuse Load. Deferred Application was again the most important Management variable for reducing the variance of both the Overland and Subsurface Diffuse Loads. After Deferred Application, the next most important management variable in terms of reducing variance, were similar to both Overland and Subsurface Loads: Fert Application Rate, Winter On or Off, Stocking Rate, and Application.

The network was then used to compare different management variables with transport factors and their effects on the Total Load, Overland Diffuse Load and Subsurface Diffuse Load (Table 25). A high Annual Rainfall (985 mm) resulted in a Total Load 2.5 times greater than a low Annual Rainfall (544 mm). In addition, the loads reaching receiving waters were four times greater from the farms on hilly slopes (15-25°) than on undulating slopes (0-7°), evidence that mitigation in these environments is of great importance. When Artificial Drainage was present the Total Load was lower than for an undrained farm (1.42 vs. 1.61 kg TP/ha). The main difference was that 56% of the diffuse load came from the overland component in the undrained catchment while 22% came from the overland component in the drained catchment. This affirms that the installation of artificial drainage shifts most of the diffuse TP load from overland to subsurface flow without a great reduction the Total Load. Therefore, in artificially drained farms, mitigation strategies need to be focused on reducing the subsurface loads from tile drain outlets.

Table 25 Results of interrogating the TP export network.

---------Transport Factors--------			-----------------------------Management variables-----------------------------							---------Node values--------		
Annual Rainfall	Artificial drainage	Days between application and runoff	Winter On or Off	Deferred irrigation	Effluent area	Application Intensity	Track design	Paddocks with access	Stocking density	Overland Diffuse Load	Subsurface Diffuse Load	Total load
Default	Default	Default	Default	Default	Default	Default	Default	Default	Default	0.772	1.36	1.5
High[a]	Default	Default	Default	Default	Default	Default	Default	Default	Default	1.27	2.08	2.27
Low[a]	Default	Default	Default	Default	Default	Default	Default	Default	Default	0.48	0.838	0.929
Default	True	Default	Default	Default	Default	Default	Default	Default	Default	0.448	1.58	1.42
Default	False	Default	Default	Default	Default	Default	Default	Default	Default	1.32	0.996	1.62
Default	Default	Undulating	Default	Default	Default	Default	Default	Default	Default	0.772	1.36	0.416
Default	Default	Hilly	Default	Default	Default	Default	Default	Default	Default	0.772	1.36	2.04
Default	Default	Default	ON Crop	True	Default	Default	Default	Default	Default	0.575	0.795	0.968
Default	Default	Default	ON Crop	False	Default	Default	Default	Default	Default	0.951	1.9	2.06
Default	Default	Default	ON Crop	Default	Default	Default	Default	Default	Default	0.736	1.27	1.44
Default	Default	Default	ON Standoff	Default	Default	Default	Default	Default	Default	0.906	1.72	1.77
Default	Default	Default	ON Standoff	True	Default	Default	Default	Default	Default	0.592	0.848	1
Default	Default	Default	ON Standoff	False	Good	Low	Default	Default	Default	0.91	1.83	1.97
Default	Default	Default	ON Standoff	False	Poor	High	Default	Default	Default	1.42	3.1	2.98
Default	Default	Default	OFF	Default	Default	Default	Default	Default	Default	0.736	1.27	1.42
High[a]	Default	Default	OFF	Default	Default	Default	Default	Default	Default	1.23	1.98	2.2
High[a]	Default	Default	ON Crop	Default	Default	Default	Default	Default	Default	1.23	1.98	2.24
Default	Default	Default	Default	Default	Default	Default	Good	0	Light	0.676	1.1	1.26
Default	Default	Default	Default	Default	Default	Default	Poor	Default	Heavy	0.855	1.59	1.66
Default	Default	Default	Default	Default	Default	Default	Default	4-5	Heavy	0.855	1.59	2.37
Default	Default	1-2	Default	Default	Default	Default	Default	Default	Default	1.18	1.36	1.76
Default	Default	20-30	Default	Default	Default	Default	Default	Default	Default	0.679	1.36	1.43

[a]High -= 985 mm; Low = 544 mm.

The benefits of different wintering strategies are closely related to the management of effluent used. When effluent was managed according to current practice (i.e. default; Table 25), the total load when wintering on crop (1.44 kg TP/ha/yr) was lower than when cows are wintered on a standoff pad or similar (1.77 kg TP/ha/yr). However, when the effluent from the standoff pad from the dairy shed was applied using the principles of deferred irrigation, then wintering on a standoff pad resulted in the lowest export of TP (1 kg TP/ha/yr). The lowest TP loads were predicted when Deferred Application was used, and from the sensitivity analysis (Tables 23 and 24) it was seen that, out of all the management variables, the loads are most sensitive to Deferred Application. If Deferred Application was not used, but the next best management strategy (Effluent Area: Good & Application Intensity: Low), the resulting load was twice as great (1.97 kg TP/ha/yr) as when Deferred Application was used. Most of the load increase was due to Subsurface Diffuse Loads. When there were no good practices in place (Deferred Application: False; Effluent Area: Poor; Application Intensity: High) the load was three times as high as when Deferred Application was used (2.98 kg TP/ha/yr). These results show the importance of proper effluent management, even when effluent is only applied over 10-20% of the farm. The practice of deferred application is of great benefit to reducing TP loads at the farm scale. In addition to crop or standoff pads, cows may be wintered off farm as is common in this area (Kira et al., 2008). According to the network, when animals are wintered off farm the Total Load is only slightly lower than if cows are wintered on crops, but this difference increases if more rainfall and therefore more runoff is received.

Table 26 Diagnostic analysis of the Management variables linked to the Effluent Load (Assuming that the rainfall is average (764mm), on an imperfectly drained Pallic soil, and that a standoff pad is used). The most probable state of the given node is given along with the degree of probability expressed as a percentage of all the states in parenthesis (from 100% = most probable, 0% not probable).

Effluent Load (kg/ha)	Deferred application	Effluent Area	Application Intensity	Standoff pad use	Lactation days	Stocking rate
0.15	True (100%)	Good (99%)	High (85%)	8- 10 (31%)	260-270 (48%)	Light (100%)
90	False (100%)	Poor (100%)	High (100%)	10-12 (32%)	260-270 (51%)	Heavy (74%)

The importance of the common point sources of dairy farms in this region was also investigated. When Track Design is Poor and the Stocking Rate is High the annual TP load is 30% higher than when best practices are in place (1.66 vs. 1.26 kg TP/ha/yr) (Track Design: Good; Stocking Rate: Low; Paddocks With Access: 0). When the Stocking Rate is High and stock are allowed access to streams in 4-5 paddocks, the Total Load is almost 90% higher than when best practices are used (2.37 kg TP/ha/yr). This implies that these point sources can be of great importance to the Total Load of TP exported at the farm scale. Mitigation of these sources can also reduce the default, or current practice, load by 16%. This may not seem like a large amount, but it should be noted that the TP load from these point sources often has a high proportion of dissolved reactive P, which is directly available for biotic uptake.

The Effluent Load is calculated using a total of nine nodes including five management variables (Figure 23). Since the Effluent Load is an important part of both the Overland and Subsurface Diffuse Load, this part of the network was used to answer the question: what management variable states will lead to low vs. high effluent loads? The network was used diagnostically to accomplish this. For the purposes of this analysis is was assumed that the Annual Rainfall was average (764 mm), the Dominant Soil Type was Pallic, the soil was Imperfectly Drained and a standoff pad was used. When the Effluent Load was at its lowest possible (0.15 kg/ha), the most probable states of the management nodes were that Deferred Application was True, the Effluent Area was Good and the Stocking Rate was Low (Table 26). The states of the remaining nodes were no different from their default states. When the highest Effluent Load was selected (90 kg/ha), the network predicted that Deferred Application was False, the Effluent Area was Poor, and that High Application Intensity and Heavy Stocking Rate were used. The Standoff Pad Use and the Lactation Days nodes showed almost no change for these two extreme scenarios. These nodes were also among the least effective at reducing the variance of the load variables in the sensitivity analysis (Tables 23 and 24). This would imply that using the average values for these variables would probably be just as effective as including the nodes in the network.

5.4 Conclusions

The response of this network to contrasting scenarios was consistent with experimental results. Annual rainfall, overland and subsurface flow, and slope have a great effect on the total phosphorus loads exported from dairy farms. These variables are beyond our control, but using the network to manipulate them helps us to understand when and where the greatest risk of loading occurs. Hilly to steep slopes need more attention than undulating landscapes as they make a sizeable contribution to loads, and in artificially drained catchments, the subsurface loads are important contributors to the total TP loads.

Under current practice, loading from diffuse sources is more important than from point sources, and of the diffuse sources, the effluent load was the greatest potential contributor to diffuse loading. Point sources have the potential to significantly affect TP loads at the farm scale, but the results showed that stock access to waterways had the potential to contribute greater loads than runoff from tracks.

The mitigation options explored here showed varying degrees of effectiveness in reducing TP loads. However, it was clear that the practice of deferred application of effluent was very effective at reducing the Total TP Load. Deferred application of effluent reduced whole farm TP loads by one third. Through the network it was possible to compare the effects of different wintering strategies and effluent management options. Using a standoff or wintering pad reduced annual loads compared with wintering on a crop, but only when effluent was applied according to the practice of deferred application.

The main shortcoming of the network was that only the most important sources and processes were included, and those processes were averaged out over a year. However, in certain situations there may be other important sources and processes at work that are not included in this network. An alternative strategy for making a BN could be to focus on a critical time period, for example when the risk of eutrophication is greatest, instead of using a yearly time step. This network has nevertheless proved to be a helpful tool for combining the findings of different aspects of P loading and understanding the interactions and relative importance of each.

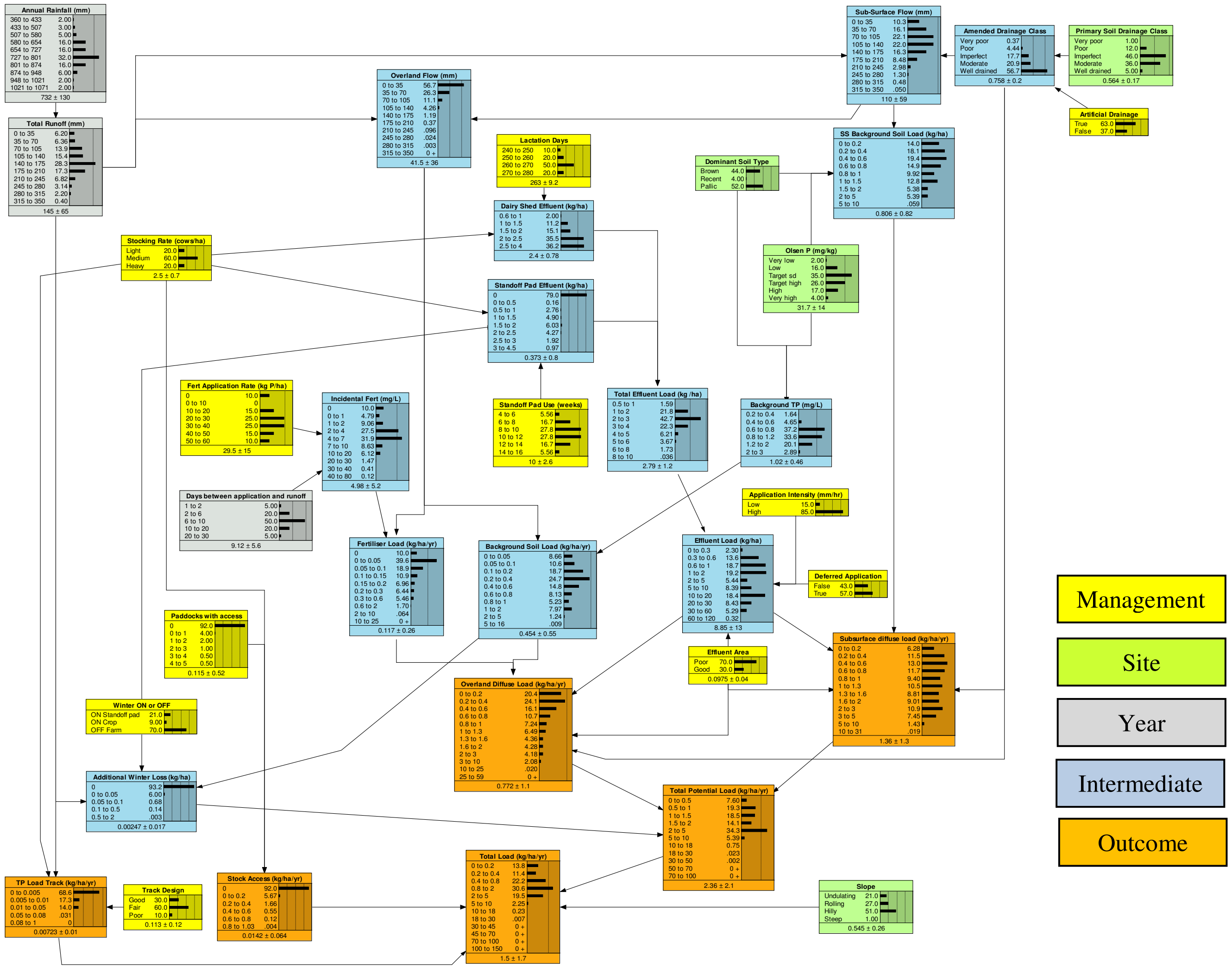

Figure 23 The Bayesian network of TP export from dairy farms in Otago, New Zealand. Note that only arrows intersecting at right angles are linked.

Chapter 6 – Conclusions and future research priorities

This chapter begins with meeting the final objective of the thesis, which was to develop rules for assessing connectivity in different landscapes. This section is a synthesis of all the knowledge and insights gained in the previous experiments. Section 6.2 summarizes the key findings and conclusions from this work and Section 6.3 describes some priorities for future research.

6.1 Rules for assessing connectivity in different landscapes

Given the wide variety of possible weather and site effects which impact on connectivity, formulating absolute rules for assessment is difficult. Often the best answer that can be given to the question of what impact a source will have at a given scale is that it depends on the specific circumstances. However, there are some guidelines that can be drawn from the results gathered from the dairy catchment in Chapter 2 and the nested catchment study in Chapter 4. The results from the dairy catchment showed that there was variation in loading by season, and also in the dominant fraction of P exported. The results from the nested catchment study showed that the connectivity between sites for particulate P was evident at all scales. In contrast, connectivity between sites for dissolved P was only found at smaller scales.

To combine these results into rules/guidelines, a lookup-table was developed that classified the potential contribution of P loading from the three sites examined in Chapter 2, based on season and the main form of P lost (Table 27). The contribution was classified as low, medium or high and the export from pasture ("Medium") was used as the basis for comparison. From the analysis of the Bayesian Network in the previous chapter, it was found that the annual loading from grazed pasture is the greatest source of total P due to its spatial dominance. Other point sources also have the potential to contribute significantly, as was shown in the modelling exercise in Chapter 2 (Section 2.3.3.5), but this was only shown during specific conditions and not over the whole year. Simply put, "Low" indicates that the P load contribution from the source will be lower than from the pasture, "High" indicates that the loading from the source is higher than that of the pasture, and "Medium" indicates that the load from the source is on par with that of the pasture.

This table also classifies the likelihood of connectivity at different scales. Connectivity in this context is when the load from a source is observed at a scale greater than the immediate area of the source. This likelihood of connectivity was classified as "Low" or "High" for the Field (< 10 ha), Farm (10-500 ha) and Catchment (> 500 ha) scales.

Table 27 Guidelines for the contribution of different sources of P loading to storm events, the dominant fraction of P lost and connectivity at different scales by season. Pasture contributions are set to "Medium" and the contribution from other sites are relative to its loading. (Field:< 10 ha; Farm:10-500 ha; Catchment: > 500 ha).

		Summer	Autumn	Winter	Spring
		Pasture			
P Fraction		Particulate	Particulate	Particulate	Particulate
Contribution		Medium	Medium	Medium	Medium
Connectivity:	Field	High	High	High	High
	Farm	High	High	High	High
	Catchment	High	High	High	High
		Trough			
P Fraction		Dissolved	Dissolved	Particulate	Particulate
Contribution		Low	Low	Low	Low
Connectivity:	Field	Low	Low	Low	Low
	Farm	Low	Low	Low	Low
	Catchment	Low	Low	Low	Low
		Lane			
P Fraction		Dissolved	Dissolved	Dissolved	Dissolved
Contribution		Low	High	Medium	Low
Connectivity:	Field	Low	High	High	Low
	Farm	Low	Low	Low	Low
	Catchment	Low	Low	Low	Low

Using this table it is evident that, for example, that during spring and summer, the majority of the loading comes from the pasture, that it is in particulate form (Table 27) and that loading in the particulate form has a high connectivity during storm flow through all scales. During the autumn, the load of dissolved P from the lane can be greater than from the pasture, but during storm flows this load will only be connected at the paddock scale.

This table does not include connectivity between sites during base flows because it is assumed that connectivity between the source areas investigated (trough and lane) will only occur when the sites are hydrologically active. However, dissolved P may potentially desorb from sediments during base flows. This source, or other point sources of dissolved P present, is likely to be connected at the farm scale. Similarly, particulate P loads found during base flow (e.g. from stock access to streams) will be localized to the field scale as was found in the small scale farm catchments in Chapter 4.

These guidelines are meant to be used when evaluating the risk from an area comparable to the one that was studied here, which is with similar soil types, topography and land use. From this table the relative magnitude of the source in relation to pasture loading can be seen, as well as the form of loading and at which scale the effects of these sources will be observed. Of course, this table represents average conditions and source areas similar to the ones measured in this study.

6.2 Conclusions

In summary, a review of the literature identified a clear gap in the research regarding highly trafficked areas on dairy farms that could potentially contribute to P loading. It has also identified trends in loading at different scales and land uses. This thesis has addressed these knowledge gaps by exploring how and when the lanes and watering troughs found on dairy farms contribute to total P loads. Additional analysis of P fractions and different flow pathways and regimes revealed that the loading of P is dependent not only on scale, but also on the fraction of P being transported. In an effort to elucidate some of the processes controlling the availability of P in streams this thesis also explored different methods of measuring EPC_0. Finally, a Bayesian Network has been constructed to link together the findings of this work and others, and to produce a working model capable of assisting in a addressing many remaining questions.

The research set out to meet the five objectives outlined in Chapter 1. Objective I was to quantify the potential source areas within a dairy catchment and this was accomplished through the experiments detailed in Chapter 2. These experiments proved that the loading from a trough and from a laneway were greater than from pasture. Scaling up runoff plot data from 8 storm events suggested that the laneway, which made up < 2% of the catchment, could potentially contribute as much as 89% of the DRP load. From the Bayesian Network it was shown that poorly designed laneways made a significant contribution to farm scale P loading, and that by adopting good track design and lowering the stocking rate, TP exported from this source could be reduced considerably. However diffuse sources still had the potential to make even greater contribution to loads when effluent was poorly managed.

Objective II was to identify and quantify the potential pathways of P loss from the source area. The potential pathways investigated included overland flow and subsurface flow, as detailed in Chapter 2. Concentrations of dissolved P were at times two orders of magnitude greater in overland flow from the lane than from the pasture. Subsurface flow was also found to be an important pathway for P loss. The dissolved reactive P concentrations measured in the adjacent stream were greater during storm events than during base flow which indicated loading from a source that is not active during base flow. However, it remained unclear as to whether this load originated from one of the point sources investigated or from desorbed P in eroded sediment.

Objective III investigated the spatial and temporal variations in loads from source areas. This objective was met using sampling results from a dairy catchment (Chapter 2) and a mixed-use catchment (Chapter 4). From these investigations it was found that the amount, and also the major fraction of P loading from the lane and trough, varied throughout the year, while from the pasture the differences were not as pronounced. At the catchment scale there were no significant variations with season except for the DURP fraction. With increasing scale, the importance of localized inputs and sources decreased, and in-stream cycling and transformations seemed to be of greater importance.

Objective IV was to determine the connectivity between identified source areas and water flow pathways. This objective was especially challenging because of the uncertainty surrounding the water flow pathways. However, evidence of connectivity between sources and flow paths was found at different scales, with different land uses and for different P fractions. Evidence of connectivity was apparent at small scales (<10 ha) during base flow and storm events, and during storm flow there was a clear connection between sites for particulate P and SS. Stream P loads were dominated by particulate P in both streams, and runoff from pasture was also primarily particulate P. The dissolved fractions were more difficult to trace from sources to flow paths, but it is concluded that the processes occurring during transport have a greater effect on the dissolved fractions than the particulate fractions.

Objective V was to develop rules for assessing P losses and connectivity in different landscapes. This objective was met by combining the results and knowledge gained from the experiments into a simple table that can be used to estimate the risk of loading from different sources and the connectivity with increasing scale.

The overall hypothesis for this research was that potential source areas in grassland catchments become critical when they are in hydrologically active areas of the catchment and are connected by water flow pathways to receiving waters, though their importance as critical sources diminishes with increasing scale. The potential source areas investigated supported this hypothesis, and their importance was shown to depend on the dominant fraction of P being transported, the flow regime and the surrounding land use. Particulate fractions were more likely to be transported from small to large scales during storm flow, while dissolved fractions were more likely to undergo in-stream processing that altered the dissolved P concentrations between scales.

An underlying theme has been to suggest management to mitigate or minimize the source areas investigated. Some of the proposed strategies were to use alum to adsorb dissolved P before it is lost in runoff and vegetated buffer strips that could intercept nutrients in overland flow. While there is merit in these management strategies, the simplest advice is to position the source areas well away from streams, or other routes that could connect them to streams, wherever possible.

Management options were also investigated using the Bayesian Network in Chapter 5. The Network confirmed the importance of good track design, and also showed that management can have a major impact on the diffuse loading of TP. In the case of laneways, runoff should be treated as a source of P and not directed into waterways. Laneway runoff should instead be directed to land or collected and used for irrigation.

6.3 Priorities for future research

Based on the findings of this research future study should include the following:

6.3.1 Quantifying flow paths and transformations

The quantification of flow paths was carried out indirectly in this study, but it was found that flow paths and the connectivity of sources at different scales depend on the fractions of P studied. Particulate P was found to be the dominant fraction of P exported from pastures, though other studies have found the dissolved fraction to be of greater importance. During storm events sediment was more easily traced than phosphorous from small to large scales because sediment is only acted on by physical processes and can be entrained or resuspended, but not taken up or transformed like P. This contrast highlights how P transformations occur even during storm flow when transport is rapid.

Flow paths affect the form of P lost and the potential transformations which occur. The shallow subsurface flow path is the least understood. Because the bulk of precipitation leaves the system via this pathway, more detailed investigations are needed which quantify the interactions and transformations between flow paths and different forms of P. It was proven here that there are smaller areas that can act as sources, but the methods used here were inadequate to measure exactly what path was used for transport from source to receiving waters and the transformations that took place.

At greater scales more complex interactions come into play, therefore looking at only one nutrient and its transformations will only yield partial answers. Biological transformations in the soil are linked to inputs and availability of carbon and nitrogen, and consideration for carbon and nitrogen dynamics should be integrated with P dynamics as well. Might measuring carbon, nitrogen and phosphorus together better explain the results observed?

6.3.2 Bed sediment sorption of P

This work has highlighted the importance of in-stream processes to the loading and connectivity of dissolved P fractions. However, the measurement of EPC_0, both by standard calcium chloride solutions and stream water, has proven to be a poor predictor of dissolved P concentrations in the environments studied. In addition, evidence in Chapter 3 suggested that overlaying water was not in equilibrium with the sediments and therefore how much importance should be placed on EPC values when equilibrium is rarely achieved in the field? The ability of sediments to act as a source or a sink depends on the sediment composition as well as the conditions in the stream bed such as redox potential, sediment aeration and pH, which are time-consuming to replicate in the lab.

An alternative method for measuring the sorption capacities of sediment is needed that would better predict the sorption or desorption of dissolved P during base flows. This method would probably use the same principles as the EPC_0 methodology (i.e. equilibrating sediments with solutions spiked with P), but measured *in situ*, possibly by isolating a core of sediment in a PVC pipe and adding P solutions while the sediment remains in the stream bed. This is, of course, made difficult being underwater, and would only work during sustained low flows, otherwise a storm would likely tear out any instillations on the stream bed.

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Appendix I

Primary Soil Drainage Class - Site Variable

Node Name	*States* [*State value*]	*Conditional probability table*	*Reference*
Primary_Soil_Drainage_Class ***Variable Name*** Primary Soil Drainage Class	Very_poor [0-0.2] Poor [0.2-0.4] Imperfect [0.4-0.6] Moderate [0.6-0.8] Well_drained [0.8-1]	Prior probabilities: Very_poor 0.01 Poor 0.12 Imperfect 0.46 Moderate 0.36 Well_drained 0.05	(Hewitt, 1993) Probabilities taken from % area from NZ soil Map

The value equates to the fraction that passes into subsurface flow (ie 0 = impermeable, 1 = gravel), from NZ soil Classification. (From New Zealand Land Resource Inventory survey-NZLRI)

Dominant Soil Type - Site Variable

Node Name	*States*	*Conditional probability table*	*Reference*
Dominant_Soil_Type ***Variable Name*** Dominant Soil Type	Brown Recent Pallic	Brown 44 Recent 4 Pallic 52	From NZ soil map (Hewitt, 1998) The 3 soil types make up 95% of the area

These 3 soil types make up 95% of area. (From New Zealand Land Resource Inventory survey-NZLRI)

Olsen P (mg/kg) - Site Variable

Node Name	*States [State value]*	*Conditional probability table*	*Reference*
Olsen_P ***Variable Name*** Olsen P (mg/kg)	Very_low [0-10] Low [10-20] Target_Sd [20-30] Target_high [30-40] High [40-60] Very_high [60-80]	Very low 2 Low 16 Target Sd 35 Target high 26 high 17 Very high 4	(Roberts & Morton, 2009) (Wheeler et al., 2004)

Olsen P levels Classified according to the target ranges in Roberts & Morton 2009: 20-30 mg/kg for dairy; 30-40 mg/kg for high production dairy. Prior probabilities from SFS and 500 soils database as described by Wheeler 2004.

Slope – Site Variable

Node Name	*States [State value]*	*Conditional probability table*	*Reference*
Slope ***Variable Name*** Slope	Undulating [0-0.3] Rolling [0.3-0.6] Hilly [0.6-0.9] Steep [0.9-1.1]	Undulating 21 Rolling 27 Hilly 51 Steep 1	Approximately same weighting as (McDowell et al., 2005c) (Classes from New Zealand Land Resource Inventory survey-NZLRI)

A greater weighting and influence on P loss is given to steep slopes, ie all potential runoff will reach waterways. Weighting converted to ranges for Bayesian network. Steep areas have the potential to export greater loads due to erosion.

Annual Rainfall – Year Variable

Node Name	***States Continuous***	***Conditional probability table***	***Reference***
Annual_Rainfall	360-		CliFlo: NIWA's National Climate Database on the Web. Retrieved DATE: February 28, 2011. http://cliflo.niwa.co.nz/ (NIWA, 2011)
	433-	2	
Variable Name	507-	3	
Annual Rainfall (mm)	580-	5	
	654-	16	
	727-	16	
	801-	32	
	874-	16	
	948-	6	
	1021-	2	
	1071	2	

Data is taken from NIWA annual values from 1980-2010 from 3 sites around south Otago: Balclutha (5866); Lovells flat (5872); Baverstock (5849)

Probabilities based on histogram of data set

Total Runoff – Year Variable

Node Name	***States continuous***	***Equation***	***Reference***
Total_Runoff	0-	Deterministic equation:	CliFlo: NIWA's National Climate Database on the Web. Retrieved DATE: February 28, 2011. http://cliflo.niwa.co.nz/ (NIWA, 2011)
	35-	Total_Runoff (Annual_Rainfall) =	
Variable Name	70-	Annual_Rainfall < 460 ? 0 :	
	105-	0.5217*Annual_Rainfall-238.63	
Total Runoff (mm)	140-		
	175-		
	210-		
	245-		
	280		

The equation estimates total runoff from linear regression based on NIWA dataset for 1980 – 2010 for 3 sites around south Otago (R^2 = 0.76). No runoff when annual rainfall was below 460 mm. (Runoff is calculated from a water balance model described on NIWA website in reference)

Days between application and runoff - Year Variable

Node Name	***States continuous***	***Conditional probability table***	***Reference***
Days_between_application	1- 2- 6- 10- 20- 30	Normally distributed	
Variable Name			
Days between application and runoff			

Stocking Rate (cows/ha) – Management variable

Node Name	*States* *[State value]*	*Conditional probability table*	*Reference*
Stocking_Rate ***Variable Name*** Stocking Rate	Light [1-2] Medium [2-3] Heavy [3-4]	normally distributed around average	(LIC, 2009)

Average stocking rate for Otago region is 2.88 cows/ha. The range is from 2.71-3.2 in Otago. Stocking rates based on 2008-9 Dairy statistics: (LIC, 2009)

Lactation Days - Management variable

Node Name	*States* *[State value]*	*Conditional probability table*	*Reference*
Lactation_Days ***Variable Name*** Lactation Days	[240-250] [250-260] [260-270] [270-280]	normally distributed around average	(LIC, 2009)

Average for south island is 264 days (Days in Milk, production; LIC 2008/09). To estimate the time spent in the milking shed.

Effluent area - Management variable

Node Name	*States* *[State value]*	*Conditional probability table*	*Reference*
Effluent_Area ***Variable Name*** Effluent Area	Poor [0.05 – 0.1] Good [0.1-0.2]	Good 30 Poor 70	

Good > 15-20%, Poor <10% from Houlbrooke, D.J., personal communication, 1/3/2011. Prior probabilities from Otago Regional Council representative Dylan Robertson (personal communication 15/3/2011).

Application Intensity - Management variable

Node Name	***States [State value]***	***Conditional probability table***	***Reference***
Application_Intensity	Low	low 15	(Kira et al., 2008)
Variable Name	High	high 85	
Application Intensity (mm/hr)			

low = sprinkler or K-line type (4 mm/hr); high = travelling irrigator type (60mm/hr)

Deferred Application- Management variable

Node Name	***States***	***Conditional probability table***	***Reference***
Deferred_Application	True	True 57	(Kira et al., 2008)
Variable Name	False	False 43	
Deferred Application			

Deferred application = storage because one is not possible without the other

From Kira 2008 <7 days = no deferred storage 38%; > 30 days = deferred 52% (added 5% to each to make up to 100). Principles of deferred irrigation found in (Houlbrooke et al., 2004)

Winter ON or OFF– Management variable

Node Name	***States [State value]***	***Conditional probability table***	***Reference***
Winter_ON_or_OFF	ON_Standoff_pad	ON_Standoff_pad 21	(Kira et al., 2008)
Variable Name	ON_Crop	ON_Crop 9	
Winter ON or OFF	OFF_Farm	OFF_Farm 70	

Fert Application rate (kg P/ha) - Management variable

Node Name	***States [State value]***	***Conditional probability table***	***Reference***
Fert_Application_Rate ***Variable Name*** Fert Application Rate (kg P/ha)	[0] [0-10] [10-20] [20-30] [30-40] [40-50]	[0] 10 [0-10] 0 [10-20] 15 [20-30] 25 [30-40] 25 [40-50] 10	(Nguyen et al., 1989; Roberts & Morton, 1999)

Stocking rate range is from 2.71-3.2 in Otago which corresponds to maintenance P requirements of 27-55 kg P/ha (From Fertilizer use on NZ Dairy Farms, 1999) = 30kg/ha for average stocking rate (2.6). Probability of applying 0-10 kg/ha set to 0 because that application rate is not used. State was included because the states are continuous.

Track Design - Management variable

Node Name	***States [State value]***	***Conditional probability table***	***Reference***
Track_Design ***Variable Name*** Track Design	Good [0] Fair [0-0.25] Poor [0.25-0.5]	Good = 30 Fair = 60 Poor = 10	(McDowell et al., 2009b)

Probabilities from ORC (Dylan Robertson, personal communication 15/3/2011).

Paddocks with access – Management variable

Node Name	*States* *Continuous*	*Equation/conditional probability table*	*Reference*
Paddocks_Access	[0] 0- 1- 2- 3- 4- 5	0 92 0-1 4 1-2 2 2-3 1 3-4 0.5 4-5 0.5	(Kira et al., 2008)
Variable Name			
Paddocks with access			

If stock has access then duration is based on a 30 day grazing rotation and 24 hrs spent in paddock, over 10 months of the year. 92% of waterways are fenced off.

Artificial Drainage – Management Variable

Node Name	*States* *[State value]*	*Equation/conditional probability table*	*Reference*
Artificial_Drainage	True False	True 63 False 37	(Kira et al., 2008)
Variable Name			
Artificial Drainage			

Standoff Pad use (weeks)- – Management Variable

Node Name	*States* *[State value]*	*Equation*	*Reference*
Standoff_Pad_Use	[4-6] [6-8] [8-10] [10-12] [12-14] [14-16]	P (Standoff_Pad_Use |) = (TriangularEnd3Dist(Standoff_Pad_Use, 10, 4, 16))	(Longhurst et al., 2006)
Variable Name			
Standoff Pad Use (weeks)			

Stand off pad or herd home. From Longhurst et al, 2006 use <= 25% 0f year. Used up to 4 months usually 10 weeks average (+/- 2 weeks) from Bob Longhurst, personal communication, 2/3/2011. CPT normally distributed about the mean

Amended Drainage Class- Intermediate Variable

Node Name	***States [State value]***	***Equation***	***Reference***
Amended_Drainage_Class ***Variable Name*** Amended Drainage Class	Very_poor [0-0.2] Poor [0.2-0.4] Imperfect [0.4-0.6] Moderate [0.6-0.8] Well_drained [0.8-1]	Amended_Drainage_Class (Artificial_Drainage, Primary_Soil_Drainage_Class) = Artificial_Drainage == True && Primary_Soil_Drainage_Class == Very_poor? Primary_Soil_Drainage_Class + 0.4: Artificial_Drainage == True && Primary_Soil_Drainage_Class == Poor? Primary_Soil_Drainage_Class + 0.4: Artificial_Drainage == True && Primary_Soil_Drainage_Class == Imperfect? Primary_Soil_Drainage_Class + 0.4: Artificial_Drainage == True && Primary_Soil_Drainage_Class == Moderate? Primary_Soil_Drainage_Class + 0.2: Primary_Soil_Drainage_Class	Same classification as primary drainage class. (Sharpley & Syers, 1976; Smith & Monaghan, 2003a).

This node increases the proportion of runoff in subsurface flow if artificial drainage is present. The drainage is increased by two classes according to given references.

Sub-Surface Flow – Intermediate Variable

Node Name	*States* *Continuous*	*Equation*	*Reference*
Sub_Surface_Flow	0- 35- 70- 105- 140- 175- 210- 245- 280- 315- 350	Sub_Surface_Flow (Total_Runoff, Amended_Drainage_Class) = Total_Runoff * Amended_Drainage_Class	CliFlo: NIWA's National Climate Database on the Web. Retrieved DATE: February 28, 2011. http://cliflo.niwa.co.nz/:http://cliflo.niwa.co.nz/ (NIWA, 2011)
Variable Name			
Sub-Surface Flow (mm)			

This node uses the percentage of secondary soil class to determine the percent primary soil class and then uses the drainages classes of each to work out the amount of subsurface flow.

Overland Flow – Intermediate Variable

Node Name	*States* *continuous*	*Equation*	*Reference*
Overland_Flow ***Variable Name*** Overland Flow (mm)	0- 35- 70- 105- 140- 175- 210- 245- 280- 315- 350-	Overland_Flow (Total_Runoff, Sub_Surface_Flow) = Total_Runoff - Sub_Surface_Flow < 0? 0: Total_Runoff - Sub_Surface_Flow	

Simply calculates surface flow as the difference between subsurface flow and total runoff (and therefore dependent on subsurface runoff)

Background TP (mg/L) - Intermediate Variable

Node Name	*States* *Continuous*	*Equation*	*Reference*
Background_TP ***Variable Name*** Background TP (mg/L)	0.2- 0.4- 0.6- 0.8- 1.2- 2- 3-	Background_TP (Dominant_Soil_Type, Olsen_P) = Olsen_P > 8 ? (Dominant_Soil_Type == Brown? (0.0071 * Olsen_P + 0.236) + 0.3: Dominant_Soil_Type == Recent? (0.025 * Olsen_P + 0.655) +0.3: (0.0307* Olsen_P - 0.1573)+0.3) : 0.3	(McDowell et al., 2003a) (McDowell et al., 2007a)

Equations take from paper and averaged for both soils/soil type.

0.3 mg TP/L was added to the final concentration to take into account the effects of treading and dung deposition. If Olsen P values were below 8 then the result was negative so concentrations when Olsen P below 8 is given the value of 0.01 mg/l.

Incidental Fert (mg/L) - Intermediate Variable

Node Name	***States*** ***continuous***	***Equation***	***Reference***
Incidental_Fert	0 0- 1- 2- 4- 7- 10- 20- 30- 40- 80	Incidental_Fert (Days_between_application, Fert_Application_Rate) = Fert_Application_Rate == 0 && Days_between_application < 30? 0: (1.07*Fert_Application_Rate-10.3)* Days_between_application^ (-0.0076*Fert_Application_Rate-0.55)	(McDowell et al., 2003b)
Variable Name			
Incidental Fert (mg/L)			

Decay rate equation adapted from equations in McDowell *et al.*, 2003b to suit variable application rates.

Dairy shed effluent (kg/ha) - Intermediate Variable

Node Name	***States*** ***Continuous***	***Equation***	***Reference***
Dairy_Shed_Effluent	0.6 1- 1.5- 2- 2.5- 4	Dairy_Shed_Effluent (Stocking_Rate, Lactation_Days) = (Stocking_Rate* Lactation_Days*50*70)/1000000	(Vanderholm, 1984) (Longhurst et al., 2000)
Variable Name			
Dairy Shed Effluent (kg)			

From Vanderholm 1984 50L/animal/day; from longhurst 2000 average concentration of effluent 70 mg/l

Standoff Pad Effluent (kg/ha) - Intermediate Variable

Node Name	*States* *Continuous*	*Equation*	*Reference*
Standoff_Pad_Effluent ***Variable Name*** Standoff Pad Effluent (kg/ha)	0 0- 0.5- 1- 1.5- 2- 2.5- 3- 4.5	Standoff_Pad_Effluent (Stocking_Rate, Standoff_Pad_Use, FeedPad_Herd_Home) = FeedPad_Herd_Home == False ? 0: (Stocking_Rate * Standoff_Pad_Use*7*0.01)	(Longhurst et al., 2006)

Standoff pad or Herd home, both have similar forms of management (stockpiled until soil conditions are suitable for land application) and concentrations of P. From Longhurst et al. 2006:in southland 150 cows, using pad 90 days/year accumulated 400 m^3 of manure (BD 0.92t/m^3)The manure contained 0.6kg P/t Therefore manure P is estimated at 0.016 kg P/cow/day and rounded down to 0.01 kg P/cow/day.

Total Effluent Load (kg/ha) - Intermediate Variable

Node Name	*States* *continuous*	*Equation*	*Reference*
Total_Effluent_Load ***Variable Name*** Total Effluent Load (kg/ha)	0.5- 1- 2- 3- 4- 5- 6- 8- 10	Total_Effluent_Load (Dairy_Shed_Effluent, Standoff_Pad_Effluent) = Dairy_Shed_Effluent + Standoff_Pad_Effluent	

Sum of effluent loads

Effluent Load (kg/ha) - Intermediate Variable

Node Name	*States* continuous	*Equation*	*Reference*
Effluent_Load ***Variable Name*** Effluent Load (kg/ha)	0- 0.3- 0.6- 1- 2- 5- 10- 20- 30- 60- 120	Effluent_Load (Total_Effluent_Load, Deferred_Application, Application_Intensity, Effluent_Area) = Deferred_Application == True ? Total_Effluent_Load/Effluent_Area * 0.03: Application_Intensity == Low? Total_Effluent_Load/Effluent_Area * 0.30: Total_Effluent_Load/Effluent_Area * 0.60	(Houlbrooke et al., 2003; Houlbrooke et al., 2004; McDowell et al., 2005a; Houlbrooke et al., 2008)

SS Background Soil Load (kg/ha)- Intermediate Variable

Node Name	*States* continuous	*Equation*	*Reference*
SS_Background_Soil_Load ***Variable Name*** SS Background Soil Load (kg/ha)	0- 0.2- 0.4- 0.6- 0.8- 1- 1.5- 2- 5- 10	SS_Background_Soil_Load (Dominant_Soil_Type, Olsen_P, Sub_Surface_Flow) = ((Olsen_P < 8? 0.01: Dominant_Soil_Type == Brown? 0.0071 * Olsen_P + 0.236: Dominant_Soil_Type == Recent? 0.025 * Olsen_P + 0.655: 0.0307* Olsen_P - 0.1573) *Sub_Surface_Flow/100)	(McDowell et al., 2003a)

Background Soil Load (kg/ha/yr)- Intermediate Variable

Node Name	*States continuous*	*Equation*	*Reference*
Background_Soil_Load ***Variable Name*** Background Soil Load (kg/ha/yr)	0- 0.05- 0.1- 0.2- 0.4- 0.6- 0.8- 1- 2- 5- 16	Background_Soil_Load (Background_TP, Overland_Flow) = Background_TP *Overland_Flow/100	

Concentration from Background TP multiplied by the overland flow to get the load (kg/ha/yr)

Additional Winter Loss (kg/ha) - Intermediate Variable

Node Name	*States continuous*	*Equation*	*Reference*
Additional_Winter_Loss ***Variable Name*** Additional Winter Loss (kg/ha)	[0] 0- 0.05- 0.1- 0.5- 2	Additional_Winter_Loss (Winter_ON_or_OFF, Total_Runoff, Background_Soil_Load) = Winter_ON_or_OFF == ON_Crop? (Total_Runoff < 100? 0: Total_Runoff > 300? Background_Soil_Load * 7 /(6*10) : Background_Soil_Load * 3/(6* 10)) : 0	(McDowell, 2006b; McDowell et al., 2006; McDowell & Houlbrooke, 2009)

From references: Loads ranged from 0 – 10 times the pasture. Where loads were 7 times higher, the rainfall was 25% greater that year. The load is relative to background load so that loads are not counted twice. The load is divided by 6 because cropping takes place 2 months out of the year and divided by 10 assuming cropping is used on 10% of the farm.

Fertiliser Load (kg/ha/yr) - Intermediate Variable

Node Name	***States continuous***	***Equation***	***Reference***
Fertiliser_Load ***Variable Name*** Fertiliser Load (kg/ha)	[0] 0- 0.05- 0.1- 0.15- 0.2- 0.3- 0.6- 2- 10- 25-	Fertiliser_Load (Incidental_Fert, Overland_Flow) = (Incidental_Fert*Overland_Flow/20/100))	

The surface runoff divided by 20 because the average runoff in October or November is 5% of the annual runoff. It is assumed that this effect is only valid for the month that fertiliser is applied

Overland Diffuse Load (kg/ha/yr) – Outcome Variable

Node Name	*States continuous*	*Equation*	*Reference*
Overland_Diffuse_Load ***Variable Name*** Overland Diffuse Load (kg/ha/yr)	0- 0.2- 0.4- 0.6- 0.8- 1- 1.3- 1.6- 2- 3- 10- 25- 59	Overland_Diffuse_Load (Effluent_Area, Effluent_Load, Fertiliser_Load, Background_Soil_Load, Amended_Drainage_Class) = Background_Soil_Load * (1-Effluent_Area)+ (Effluent_Load* (1-Amended_Drainage_Class) * Effluent_Area) + Fertiliser_Load	

Incidental effluent + Incidental fert + Background. Effluent Load takes into account the % of farm that the effluent is applied to and the % of overland flow out of total runoff. The rest of the farm area is background

Subsurface Diffuse load (kg/ha) - Outcome Variable

Node Name	***States*** ***continuous***	***Equation***	***Reference***
Subsurface_Diffuse_Load	0 0.2- 0.4- 0.6- 0.8- 1- 1.3- 1.6- 2- 3- 5- 10- 31	Subsurface_Diffuse_Load (SS_Background_Soil_Load, Effluent_Load, Effluent_Area, Amended_Drainage_Class) = SS_Background_Soil_Load * (1-Effluent_Area) + Effluent_Load * Amended_Drainage_Class * Effluent_Area	
Variable Name			
Subsurface diffuse load (kg/ha)			

= background + incidental effluent; background is calculated as the farm area not used for effluent plus the proportion of subsurface flow times the load of effluent load, also by the area used for effluent spreading.

TP Load Track (kg/ha/yr) - Outcome Variable

Node Name	***States*** ***continuous***	***Equation***	***Reference***
TP_Load_Track ***Variable Name*** TP Load Track (kg/ha)	0- 0.005- 0.01- 0.05- 0.08- 1	TP_Load_Track (Track_Design, Total_Runoff, Stocking_Rate) = 5.5 * Total_Runoff/100*0.005*Track_Design * (Stocking_Rate == light ? 0.5: Stocking_Rate == heavy? 1.5: 1)	Own data (section 2) 5.5 mg/l average concentration from lane at Hillend (McDowell et al., 2009b)

The average concentration from lane (5.5 mg/L) at Hillend. Weighting from (McDowell et al., 2009b). Total runoff (not just surface) is used to calculate load because lanes are impervious surfaces. Load also adjusted for stocking rate: light means half as much while heavy means 50% more than average stocking rates.

Stock Access (kg/ha) - Outcome Variable

Node Name	***States*** ***continuous***	***Equation/conditional probability table***	***Reference***
TPoint_Source_Access ***Variable Name*** Point Source Access (kg/ha)	[0] 0- 0.2- 0.4- 0.6- 0.8- 1.03	Stock_Access (Paddocks_Access, Stocking_Rate) = Paddocks_Access == 0 ? 0: (0.03* 20* (7.8*.92) *Stocking_Rate * Paddocks_Access*10)/1000	(James et al., 2006) (McDowell et al., 2006) (Davies-Colley et al., 2004)

0.03 Poos/cow/hr from (James et al., 2006); 7.8g/kg concentration from (McDowell et al., 2006); 0.92 kg weight from Davies-Colley RJ, Nagels JW, Smith RA, Young RG, Phillips CJ (2004)

Total Potential Load (kg/ha/yr) - Outcome Variable

Node Name	*States* *continuous*	*Equation*	*Reference*
Total_Potential_load	0- 0.5- 1- 1.5- 2- 5- 10- 18- 30- 50- 70- 100	Total_Potential_load (Overland_Diffuse_Load, Subsurface_Diffuse_Load, Additional_Winter_Loss) = Overland_Diffuse_Load + Subsurface_Diffuse_Load + Additional_Winter_Loss	
Variable Name			
Total Potential Load (kg/ha/yr)			

Sum of sources

Total Load (kg/ha/yr) - Point Source Load

Node Name	*States* *continuous*	*Equation*	*Reference*
Total_Load	0- 0.2- 0.4- 0.8- 2- 5- 10- 18- 30- 45- 70- 100- 150-	Total_Load (Slope, Total_Potential_load, TP_Load_Track, Stock_Access) = (Slope * Total_Potential_load) + TP_Load_Track + Stock_Access	(McDowell et al., 2005c)
Variable Name			
Total Load (kg/ha/yr)			