Impacts of livestock grazing on soil physical quality and phosphorus and suspended sediment losses in surface runoff

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We confirm that the following papers have been accepted or submitted for publication in advance of submission of the thesis for examination:

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(2) Curran-Cournane, F., McDowell, R.W. and Condron, L.M. 2010. Effects of cattle treading and soil moisture on phosphorus and sediment losses in surface runoff from four pastoral soils in southern New Zealand. **New Zealand Journal of Agricultural Research** (in press).

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Abstract

Livestock grazing is considered a major contributor in the loss of phosphorus (P) and suspended sediment (SS) from soil in surface runoff, which in turn can have adverse impacts on aquatic ecosystems. Recent and ongoing increases in the intensity of pastoral farming in New Zealand represent an enhanced risk of P and SS loss.

The objective of this research was to assess and quantify the impacts of livestock grazing on soil physical quality (measured by macroporosity, soil bulk density, and saturated hydraulic conductivity K_{sat}) and associated losses of P and SS in surface runoff. The hypothesis was that livestock grazing would have detrimental impacts of soil physical quality which would increase P and SS loss. A mechanical cattle hoof and rainfall simulator were used to examine the influence of cattle treading and soil type on short-term P and SS losses. Field experiments were also carried out to determine if stock type (sheep, beef cattle and deer) influenced P and SS losses in runoff and the last experiment examined the potential to mitigate losses in runoff via sub-soiling in cattle-grazed pasture.

Results revealed that soil type, soil moisture, and land management practices have a major influence on soil physical quality and losses of P and SS in runoff under livestock grazing. Losses of P and SS in runoff were greater from the more compacted Pallic and Recent Gley soils with mean macroporosity values of 12 and 17% v/v, respectively, compared with Brown (23% v/v) and Melanic (37% v/v) soils. Data from field trials at Invermay and Windsor showed that soil physical quality differed between the Brown-Pallic soil at Invermay and the Pallic soil at Windsor. For example, mean macroporosities for cattle grazed plots were 16 and 8% v/v, respectively. At Invermay, significant relationships were found between loads and concentrations of P and SS with changes in macroporosity and K_{sat}; an increase in compaction increased surface runoff losses. These findings are comparable to results obtained from the rainfall simulation studies with greater P and SS losses occurring from the heavier compacted Pallic and Recent Gley soils than the better-structured Melanic and Brown soils.

Soil moisture had a major influence on P fractions lost and runoff processes. Increasing soil moisture from 10% to 90% available water holding capacity increased particulate P concentration from 0.028 mg/L to 0.161 mg/L, while SS concentrations increased from 0.009 g/L to 0.169 g/L. These changes were mainly attributed to deeper hoof penetration and soil disturbance with increased soil moisture. On the other hand, dissolved reactive P concentrations decreased with increased soil moisture from 0.423 mg/L to 0.127 mg/L, respectively, which was mainly attributed to a combination of P release from soil

microbial biomass and lack of dilution. Results from the field study carried out at Invermay demonstrated how soil moisture can influence seasonal P dynamics and runoff processes. Winter runoff occurred as a consequence of saturation-excess conditions that accounted for most P and SS loss (mean 0.015 kg TP/ha and 17.61 kg SS/ha). In summer, infiltration-excess runoff dominated and although loads (mean 0.007 kg TP/ha and 1.23 kg SS/ha) were not as great as those occurring in winter, concentrations were greater (summer mean 1.585 mg TP/L versus winter mean 0.015 mg TP/L). The enrichment of P concentrations in summer could pose a significant algal growth risk to surface water quality due to increased light and warmth.

Management practices were confirmed as having a major influence on soil physical quality and losses of P and SS in runoff. Results from the rainfall simulator study showed that cattle dung was the main source of TP and SS in runoff (0.511 mg/L and 0.092 g/L, respectively). Field results revealed that P and SS losses in runoff decreased with time since grazing. Treading by cattle had greater negative impacts on all soil physical properties (macroporosity, soil bulk density and K_{sat}) than sheep or deer treading. For example, mean macroporosity for cattle grazed plots was 16% v/v followed by 22 and 23% v/v for deer and sheep grazed plots, respectively. However, stock type did not affect P and SS losses in runoff. To mitigate runoff losses of P and SS from a Pallic soil at Windsor, the site was sub-soiled (aerated) to 20 cm soil depth. Results revealed an increase in dissolved reactive P (2.24 kg P/ha/yr) for aerated soil compared with non-aerated (control) soil (1.20 kg P/ha/yr), which was attributed to mechanical soil disturbance causing the desorption of P from soil during the last storm event, which had a return period of 40 years. Six months after soil aeration soil physical quality was similar between aerated and control treatments.

The collective findings of this research demonstrated how livestock grazing negatively impact soil physical quality and P and SS losses in runoff. It was clear that the best management practice to effectively decrease these losses is to restrict or avoid grazing when soil moisture has reached field capacity, particularly for vulnerable soil types, regardless of stock type.

Keywords: water quality; animal treading; soil compaction; soil type; soil moisture; stock type (cattle, sheep, deer)

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Chapter 1- Introduction

1.1 Background

In recent years, there has been worldwide concern regarding agriculture as a source of phosphorus (P) and suspended sediment (SS) loss causing a dramatic reduction in the quality of aquatic ecosystems (Daniel *et al.* 1994; Ulen *et al.* 2007). The ecological significance of increased excessive nutrient loss to aquatic ecosystems will be described in detail in a later section but examples include the outbreak of toxic algal blooms and loss of biodiversity (Carpenter *et al.* 1998; Yule *et al.* 2010). Livestock grazing has been identified as a major contributor of increased nutrient losses to aquatic systems (Carpenter *et al.* 1998). According to the Ministry for the Environment (2007), 30% of lakes >1 ha (n=1000) within New Zealand have poor to extremely poor water quality which is largely a result of intensive land use causing excessive losses of nutrients from land. With increasing land being cleared for agriculture (Walling 2006), the intensity of livestock grazing (duration, density, stock type) is likely to increase and the objective of this research is to advance understanding the impact livestock grazing have on soil physical quality and P and SS losses in surface runoff from pastoral soils. To gain a true understanding of the processes involved in P and SS loss we must consider the role soil plays in the natural cycling of P.

1.2 Phosphorus in soil

Most phosphorus (P) that is present in the terrestrial ecosystem is stored within the soil (Condron and Tiessen 2005). Phosphorus is essential in the soil to produce crops and provide sufficient pasture growth (Sharpley and Tunney 2000). The forms of P present in soil may be divided into two broad categories; organic P and inorganic P. Inorganic forms of P originate from the weathering of rocks containing apatite mineral, supplemented with mineral fertilisers. Organic P forms originate from compounds formed in the cells of plants, animals (and animal excreta) and micro-organisms which are released in the soil after the decomposition of dead biota (Figure 1.1). These forms of P are held in different soil P pools and a combination of physio-chemical, biological and physical processes either remove or release P into soil solution (Pierzynski *et al.* 2000).

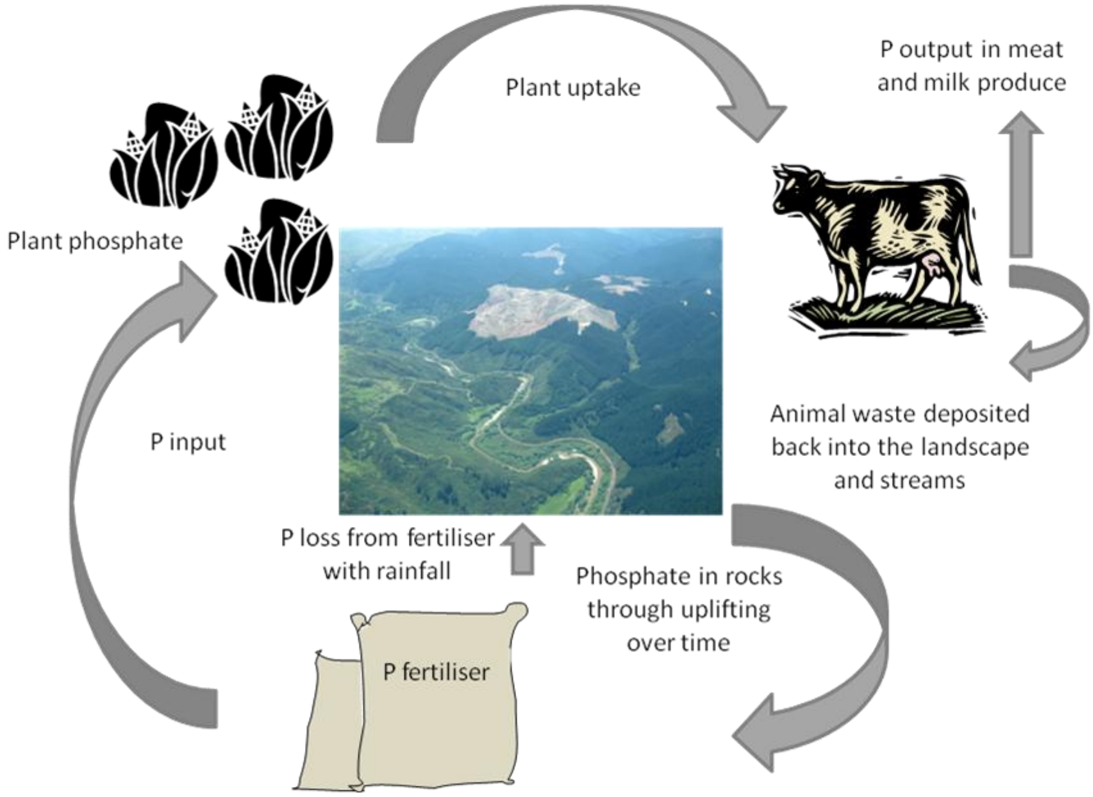

Figure 1.1. Simplistic diagram of the phosphorus cycle (refer to Pierzynski *et al.* (2000) for a detailed illustration).

Processes that remove P from soil solution- The processes that remove P from soil solution include P sorption or precipitation and microbial and plant uptake (Frossard *et al.* 1995). The term sorption combines two processes: the adsorption of phosphate ions onto iron (Fe) and aluminium (Al) compounds that coat the surface of mineral particles; and the physical/chemical sorption of phosphate ions below the surface of the particles (Pierzynski *et al.* 2000). Precipitation is the formation of discrete, insoluble compounds in soils (Cornforth 1998). Phosphorus is retained to Ca carbonates and Al and Fe hydrous oxides in soils during precipitation.

Processes that release P into soil solution- The processes that release P into soil solution include desorption/dissolution, fertiliser addition, organic P mineralisation and the release of P from decomposition of plant residue and microbial biomass turnover. Desorption describes the release of P from the solid phase into the soil solution for available plant uptake but can also be lost in surface runoff and leaching (Sharpley *et al.* 1981). Biological processes include microbial mediated mineralization which converts plant nutrients from organic to available P forms (Heathwaite *et al.* 2005). Earthworm channelling is also a biological process that aids the transport of P from soil surfaces (Haygarth and Condron 2003).

Other sources of soil P can be the direct result of the accumulation of available P in soil as a result of the build up of fertilisers and other land management practices (Frossard *et al.* 2000). Before the industrial revolution, P was applied to agricultural soils by 'recycling animal manure, crushed animal bones, human and bird excreta, city waste and ash' (Van Vuuren *et al.* 2010). This is now replaced predominantly by phosphate rock, a non-renewable P resource. Regardless of P source, an increase in soil P status enhances the risk of significant P loss from soil to water. While there is little economic advantage in extending soil P levels beyond the established agronomic optima, it can take many years for a soil that is already P-enriched to decline to optimum agronomic levels (Aye *et al.* 2006; McCollum 1991; Monaghan *et al.* 2007a; Power *et al.* 2005). The overuse of P fertiliser has also raised issues regarding the possibility of a depletion of phosphate rock as a source of P fertiliser and the demand for P fertiliser is likely to increase to support food production for the rising global population (Gilbert 2009). Although some studies have reported that such claims of short- to medium-term depletion are unwarranted (Van Vuuren *et al.* 2010), reductions in the use of P fertiliser can be achieved by recovering and recycling phosphates from livestock waste and by improving plant nutrient management (Gilbert 2009; Van Vuuren *et al.* 2010).

1.3 Background to non-point agricultural phosphorus

Excessive losses of phosphorus (P) to rivers, lakes, and oceans that impair water quality originate from either point (e.g. municipal and industrial sewage treatment plants, runoff from industrial/commercial sites) or non point/diffuse sources (e.g. agriculture) (Carpenter *et al.* 1998). The former tends to be continuous over time and can be assessed by analysing waste and chemical concentrations sporadically at the same location; hence regulation of point sources is relatively easy. In contrast, non-point sources are more difficult to monitor as they can be transported overland, underground or through the atmosphere to receiving aquatic ecosystems.

The losses of non-point agricultural P are enhanced during storm and heavy rainfall events when it can be mobilised, detached and transported in runoff to water bodies (McDowell and Srinivsan 2009). Although there is still concern regarding point sources of water pollution (Jarvie *et al.* 2005), in many cases, losses have been reduced due to their easier identification and control, but losses of non-point source pollutants have increased. Since the 1990's there has been increased interest in non-point source agricultural P because of the environmental issues associated with their export to surface waters (Powlson 1998).

The increased interest in non-point agricultural P losses arises from three factors; the environmental significance of P loss, the importance of soil erosion and associated losses of P in surface runoff and hydrological P pathways (Powlson 1998).

1.3.1. Environmental significance of phosphorus loss

Generally, pristine freshwater systems are deficient in P and nitrogen. However, as little as 20 µg P/l is sufficient for accelerated algal and plant growth resulting in eutrophication, while some algal blooms can become toxic (Powlson 1998). Phosphorus in surface and subsurface runoff is made up of both dissolved (<0.45 µm) and particulate forms (>0.45 µm). Dissolved inorganic or reactive forms of P (DRP) are readily bioavailable for algal aquatic plant uptake which can lead to the excessive growth of algal blooms and toxicity. Although the presence of DRP in fresh waters is a very algal available P species, particulate P is also a long term source of P to waters. The mitigation of particulate P forms is vital as it can be solubilised through enzymatic or physical processes making it bioavailable for uptake (McDowell *et al.* 2004a).

As well as encouraging outbreaks of toxic algal blooms, the consequences of P entering aquatic ecosystems also include decreased oxygen levels causing fish loss, reductions in biodiversity and disruptions to food chains (Carpenter *et al.* 1998). For example, in the south west coast of Ireland the outbreak of the harmful dinoflagellate, *Karenia mikimotoi*, caused mass mortality of farmed rainbow trout in 1976 (Jenkinson and Connors 1980) and clams in 1992 (Joyce 1995), threatening the livelihood of many fishermen in that area. As well as impairing fish and shellfish populations, harmful toxic algal blooms can also cause the mortality of marine mammals. Over a 10 week period in 1982, 39 Florida manatees (*Trichechus manatus latirostris*) were found dead in the lower Caloosahatchee River and nearby waters of southwestern Florida. The majority of their deaths were associated with the exposure to the outbreak of a harmful red tide dinoflagellate, *Gymnodinium breve*, which was linked to enhanced eutrophication (O'Shea *et al.* 2006).

1.3.2. Soil erosion and associated losses of phosphorus in surface runoff

Soil erosion is the process involving the detachment, transport and deposition of soil particles (Mwendera *et al.* 1997). Suspended sediment (SS) originates from soil erosion and includes soil particles < 0.7 and > 0.7 µm (Bilotta *et al.* 2007b). Losses of SS from soil to water increases the turbidity of fresh water systems which adversely affects the activity and diversity of many biota (Bilotta and Brazier 2008; Yule *et al.* 2010).

The global loss of soil by erosion is estimated at 25 billion tonnes of sediment through rivers every year (Steinfeld *et al.* 2006). Sediment-P (i.e. particulate-P, PP) has been found to be the major contributor of TP load in rivers for New Zealand (Parfitt *et al.* 2008). Generally, one third of PP which is associated with SS is biologically available (Ryding and Rast 1989) and the degree of bioavailability depends on the adsorption of P to soil particles (Heathwaite 1997). Livestock grazing is a major cause of accelerated SS loss. Suspended sediment and PP losses are exacerbated through land disturbance such as the contact of hooves on soil particularly when soils are wet (McDowell *et al.* 2003a; Smith and Monaghan 2003a). Furthermore, dung contains a considerable proportion of TP as PP (McDowell *et al.* 2006a) and SS (Curran Cournane *et al.* 2010a) lost in surface runoff.

The transport of non-point agricultural pollutants is complex and challenging to mitigate because of difficulties associated with identifying and quantifying their sources and transfer pathways.

1.3.3. *Hydrological pathways for phosphorus and suspended sediment*

Losses of P and SS from agricultural soil to surface waters occurs from several hydrological pathways (Figure 1.2) which include the downward movement of P in subsurface flow or the surface movement of P in surface runoff.

Previously, the downward movement of P was considered minimal because unlike nitrogen, P adsorbs tightly to soil particles and organic matter (Powlson 1998). However, it was discovered that an environmentally significant amount of P can be lost downwards through matrix or preferential flow in soil cracks or large macropores (Powlson 1998). The downward movement of P is further encouraged in sandy soils and soils with a low P retention (McDowell *et al.* 2004a). Soils in New Zealand that are slow draining or prone to water-logged conditions, such as Pallic and Gley soils (Hewitt 1998), are commonly drained to encourage the rapid movement of excess rainfall through preferential flow. However, contaminants such as farm dairy effluent (FDE) can be transported through the mole tiles to the aquatic environment deteriorating water quality (Monaghan and Smith 2004). Although volumes of effluent transported in mole tiles can be small, concentrations of TP can be significant. It has been suggested that strategies such as ‘deferred irrigation’ and a low rate application of FDE have the potential to reduce such losses in drainage (Houlbrooke *et al.* 2003; Monaghan and Smith 2004).

Surface runoff is generated by either saturation-excess or infiltration-excess conditions. Saturation-excess runoff is the process whereby a soil has reached saturation and any excess rainfall is lost as runoff and infiltration-excess runoff is the process whereby the intensity of rainfall exceeds the infiltration rate of the soil. Surface runoff has been identified as a major hydrological pathway for the transport and delivery of P and sediment to surface waters (Ballantine *et al.* 2009), as much soil P is associated with soil particle surfaces (Quinton *et al.* 2001). Surface runoff has been found to be an important pathway for P and SS losses in Southland and Otago, New Zealand (Smith and Monaghan 2003a), particularly for poorly draining soils with high slaking and dispersion attributes resulting in surface ponding and saturation-excess conditions (Hewitt 1998).

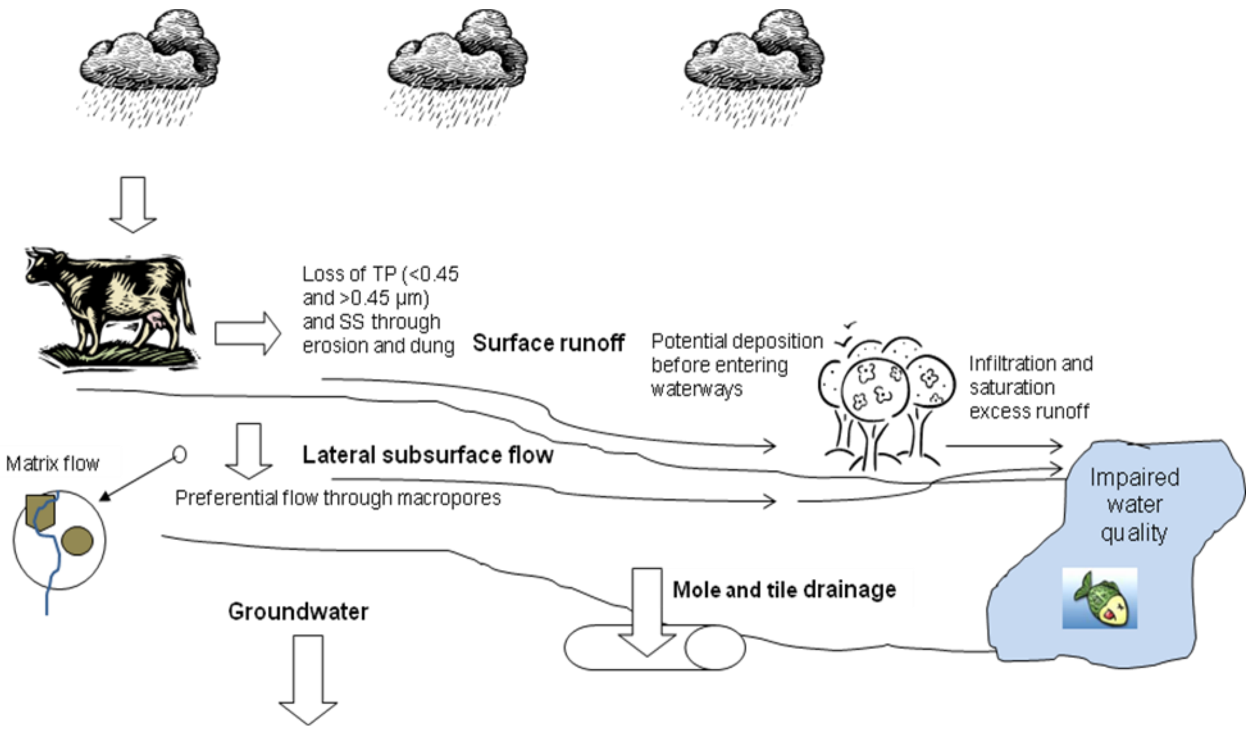

Figure 1.2. Hydrological pathways for phosphorus and suspended sediment transfer from agricultural land to surface freshwaters.

1.4 Changes in pastoral land-use in New Zealand

Globally, the intensification of livestock operations are one of the most significant contributors to environmental issues such as reductions in species diversity and disruptions to aquatic ecosystems (Steinfeld *et al.* 2006). Land use for the livestock sector occupies 70% of agricultural land and about 30% of total ice-free terrestrial land (Steinfeld *et al.* 2006). In New Zealand, the agricultural sector accounts for half of the total export income. In contrast to many regions (e.g. Europe, USA), New Zealand pastoral systems have stock outside all year. Stock type and management also varies, for example, deer farming occurs either alone or in combination with intensive or extensive sheep and beef operations.

Recently, within New Zealand there have been significant changes in land use in several areas with the conversion of many sheep farms to dairy farms with the aim to increase farm profitability (Monaghan *et al.* 2005). According to Statistics New Zealand (2010), national sheep counts were estimated at 32.3M in 2010, half the 70M recorded in the early 1980s, which has been partly attributed to dairy expansion. Dairy cow numbers have increased from 2.9M in 1981 to a record 5.9M last year. Although one third of dairy farms are located in the South Island of New Zealand, the farms are on average larger than those in the North Island in terms of both farm area and cow numbers. Figures for stock counts in Southland follow similar national trends. Currently in Southland, sheep numbers have fallen by half the 9M recorded in the early 1980s and dairy cow numbers have increased by about 19%, according to Statistics New Zealand figures for the year to June 2009. The New Zealand deer sector is very important for the economy as it is now the world's largest producer and exporter of farmed venison (MAF 2003). The population of red deer (*Cervus elaphus*) and their hybrids with wapiti (*Cervus elaphus sp.*) that are farmed in New Zealand increased from about 42,000 in 1979 to 1.4M in 2007 (MAF 2008). As a result of dairy farm expansion, deer farms are tending to move off the plains and back into the hillslopes. The expansion of dairy cow numbers in Southland and consequent pastoral land-use intensity, particularly for soils vulnerable to cattle treading, have begun to equate to environmental issues such as water and soil physical quality deterioration (Houlbrooke *et al.* 2009a; Monaghan *et al.* 2007a).

It has been estimated that between 0.42-5 kg/ha/yr of TP is lost from dairy farming followed by deer and mixed stock (sheep and beef) at 0.22-3 and 0.3-2.37 kg/ha/yr, respectively (McDowell *et al.* 2008). The contribution of large P losses from dairy may be explained by the greater loading pressure (*c.* 138 kPa) exerted on soil surfaces compared to that of sheep (*c.* 66 kPa) (Greenwood and McKenzie 2001). Intensification is known to double when animals are walking (Willatt and Pullar 1983). Such pressures can have negative impacts on soil physical quality promoting increased P and SS losses in surface water e.g. (Batey 2009). In contrast to TP losses, it has been estimated that deer farming is the greatest contributor of SS losses to surface waters (20-4480 kg/ha/yr), followed by mixed stock and dairy farming (600-2000 and 142-883 kg/ha/yr), respectively (McDowell *et al.* 2008). Deer exhibit behavioural characteristics such as fence-line pacing and wallowing which can lead to the depletion of topsoil resulting to excessive losses of SS. For example, Thorrold and Trolove (1996) reported up to 22 t/ha/year of sediment was lost from a paddock due to fence line pacing. Such P and SS losses will vary between stock type, land-use management, soil type and controls such as livestock loading, grazing intensity and duration (Monaghan *et al.* 2007a).

1.4 Objectives, hypotheses and thesis structure

New Zealand pastoral land-use is intensifying with the conversion of sheep to dairy farms and the growing deer sector, all of which is likely to continue to expand. This will have major influences on soil physical quality which in turn will affect the potential for P and SS loss in surface runoff.

The overall aim of this research project is to advance understanding the impact livestock grazing have on soil physical quality and losses of P and SS in surface runoff from pastoral soils, including recommendation of potential mitigation. To this end, a series of four hypotheses are proposed and linked to specific objectives and experiments carried out under both controlled environmental and field conditions as shown in Figure 1.3.

Objectives 1 and 2 (Chapters 3 and 4, respectively, see Figure 1.3) were carried out under controlled simulated and lab-based conditions. A mechanical cattle hoof was employed to simulate cattle treading and a rainfall simulator to generate surface runoff to analyse P and SS losses from the various treatments and soil types proposed. A cattle hoof was chosen to simulate treading to compliment, objective 3, i.e. that cattle will have a greater detrimental impact than sheep or deer. Field scale *in situ* plot studies (for objectives 3 and 4) were also used to test some of the hypotheses in objectives 1 and 2; e.g. how the impacts of grazing and seasonal changes in soil moisture regimes influence soil physical quality and P and SS losses.

The field scale *in situ* plot studies employed surface runoff plots with runoff analysed for P and SS from the various treatments proposed. Objectives 3 and 4 (Chapters 5 and 6, respectively) were undertaken at Invermay and Windsor, in Otago, and were established for 2-

and 1-years, respectively. The former is a sheep, beef cattle and deer farm and the impacts all three livestock have on soil physical quality and losses of P and SS were investigated. Windsor is a sheep and beef cattle farm, but only cattle grazing were investigated. Extensive soil physical quality sampling was undertaken in both field trials which will be discussed in detail in the appropriate chapters.

Each experimental research Chapter (3-6) provides a detailed account of the proposed objectives, hypothesis and experimental design and description. A summarised version is illustrated Figure 1.3.

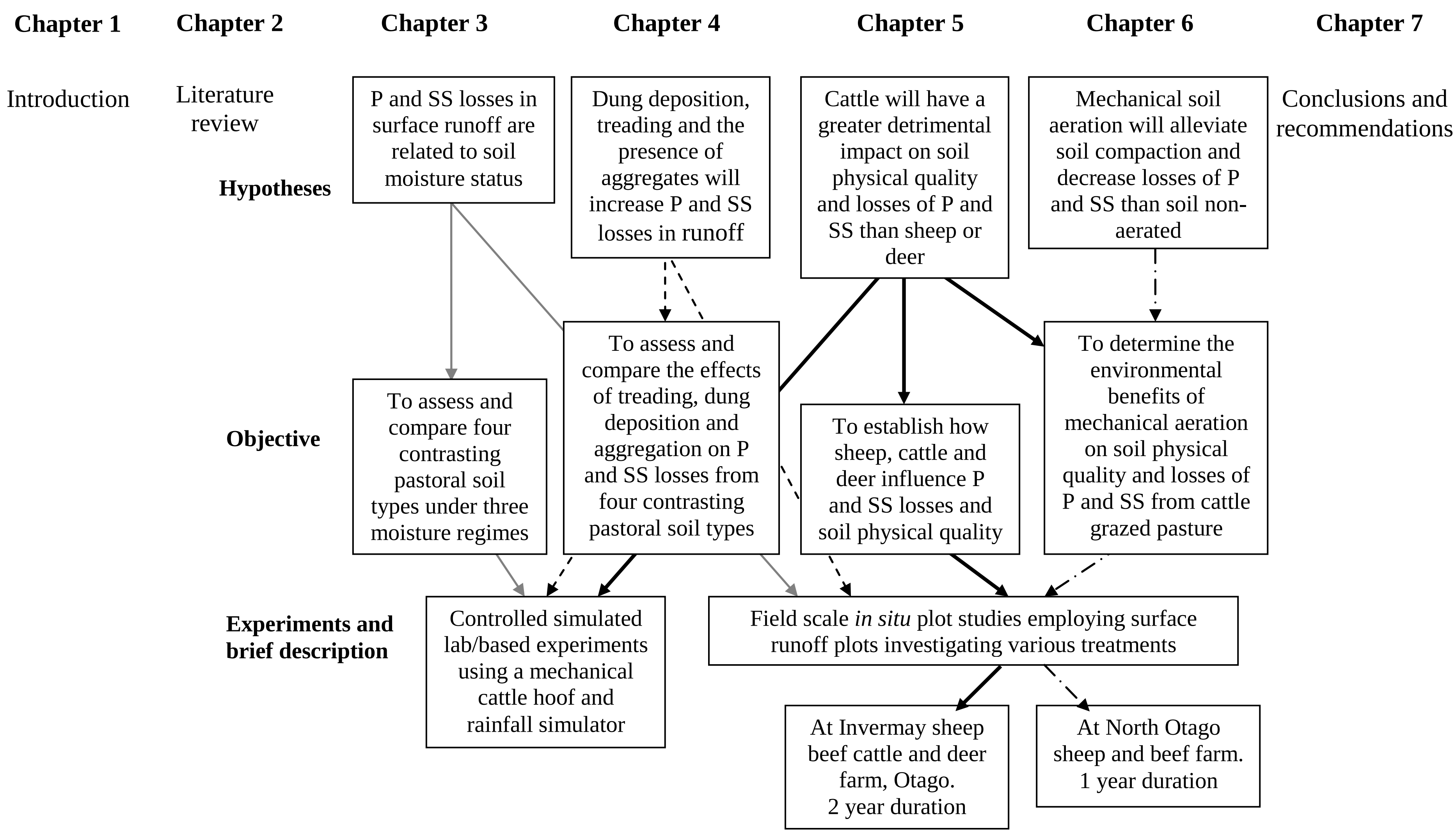

Figure 1.3. Format and structure of chapters and research experiments of the thesis.

Chapter 2- Literature Review- Impacts of livestock grazing on soil physical quality and losses of phosphorus and suspended sediment in surface runoff

2.1 Introduction

2.1.1 Phosphorus

As mentioned in Chapter 1, some forms of P that enter aquatic ecosystems are more readily available for aquatic uptake than others. Dissolved P forms are more bioavailable than particulate P forms promoting eutrophication and thereby affecting surface water quality. However, the latter form can become solubilised for aquatic uptake through enzymatic process and the desorption of P particulates to overlying waters (McDowell *et al.* 2004a) (see section 1.2). This review will report on the losses of both dissolved and particulate P forms influenced by livestock grazing. Although P can be transported in both surface and subsurface runoff (Figure 1.2) (Melland *et al.* 2008; Ulen *et al.* 2007), this review will focus on surface runoff processes.

2.1.2 Suspended sediment

Soil erosion is the process involving the removal of soil particles including organic matter, dung excreta, plant and pasture debris, colloids, sorbed contaminants such as P and any other component of soil that can be removed by the processes of erosion (Bilotta *et al.* 2007b). Suspended sediment is an important vector for the transport of contaminants and, like P, poses an environmental risk to water quality (Bilotta and Brazier 2008; Yule *et al.* 2010). An increase in SS to freshwater systems will increase the turbidity of water and interfere with primary production. The various impacts stock type have on SS losses will be discussed in detail.

2.1.3 Livestock treading, soil physical quality and losses of phosphorus and suspended sediment

Livestock treading, in the form of pugging or compaction of soil (see below and Figure 2.1), has been reported to be a major attribute encouraging soil erosion and P loss (Evans *et al.* 2006). As mentioned in Chapter 1, different stock types play very different roles on soil physical quality and losses of P and SS. For example, cattle have a greater bearing capacity than sheep and deer (Fleming 2003). Further, deer exhibit specific behavioural characteristics such as fence line pacing and wallowing promoting soil damage and P and SS losses

(Thorrold and Trolove 1996), while potential sources of P and SS for cattle and sheep include camping activity (Niu *et al.* 2009; Sheath and Carlson 1998; Van Groenigen *et al.* 2005). This variety of stock type coupled with a large variation in soil and climate magnifies the number of permutations that may lead to grazing and surface water quality problems via P and SS loss.

The purpose of this review is to highlight work done mainly in New Zealand given the extensive changes in land-use intensity (see section 1.4), and pertinent international studies, on the factors that influence soil physical quality and P and SS loss within grazed pastoral systems. In most areas of New Zealand, stock graze outside year-round and research has been carried out to investigate environmental issues associated with winter grazing in particular. As mentioned in section 1.4, the New Zealand deer sector is the world's largest exporter of farmed venison and consequently, many environmental studies regarding farmed deer have been conducted in New Zealand. This is also reflected in the review. Lastly, the review will consider the different effects cattle, sheep and deer grazing have on soil physical quality and losses of P and SS.

2.2 Impacts of animal treading on soil physical quality

Animal treading can be detrimental on soil physical quality, particularly under wet conditions, which can lead to increased nutrient and sediment losses in surface runoff (Batey 2009; Bilotta *et al.* 2007a; Drewry 2006; Tian *et al.* 2007; Zaimes *et al.* 2008). Soil compaction and pugging are the main terms used to describe the impact of animal treading on soil. Drewry *et al.* (2006) describes soil compaction as a decrease in soil porosity [volume of pores to the total volume of soil (McLaren and Cameron 1996)], particularly the volume of large inter-aggregate pores (macropores) and to some extent this can lead to an increase in the proportion of fine pores (Baumgartl and Horn 1991). Although macroporosity has been defined as pores >30 µm up to >195 µm (Cattle and Southorn 2010; Drewry 2001; Drewry and Paton 2005; Koppi *et al.* 1992), the current review considers macroporosity to be pores with diameter >30 µm unless otherwise specified. The term consolidation is used to describe the compression of a saturated soil by squeezing out water. Bilotta *et al.* (2007a) describes soil pugging as animal treading on wet soft soil, creating hoof prints (Figure 2.1).

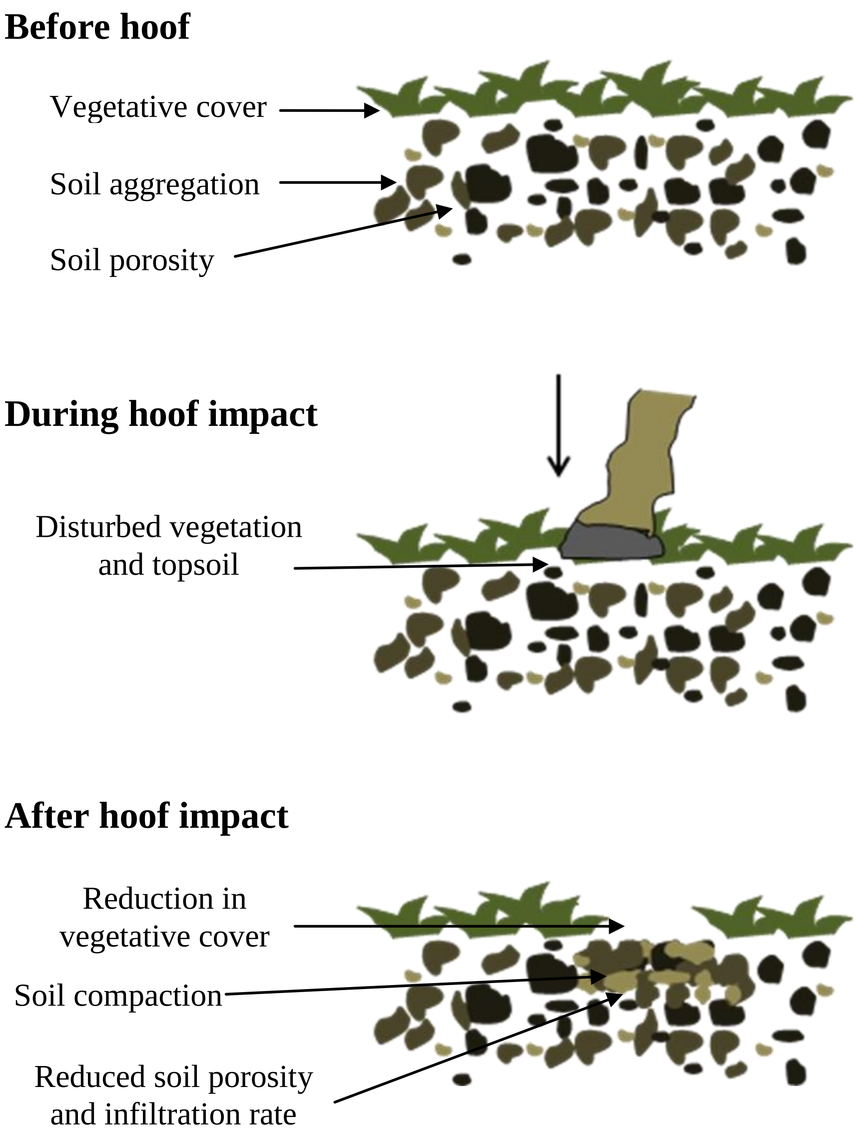

Figure 2.1 Diagram illustrating the impact of hoof treading on soil physical quality.

Hoof action can reach depths up to 20 cm when the soil is wet (Drewry 2006; McDowell *et al.* 2003b; Ward and Greenwood 2002) with greatest soil compaction occurring at the 5-10 cm soil depth according to a study that was carried out in the Southern Great Alps (Northup *et al.* 2010). When very wet, plastic deformation can arise when soil moisture is greater than the plastic limit of the soil. In this state, destruction of aggregates and rearrangement of microaggregates and soil particles occurs. This, in turn, will decrease soil infiltration (which controls the movement of water into the soil) and consequently ponding and loss of P and SS can occur from land to water bodies via surface runoff (Greenwood 1999).

Any topsoil disturbance will affect the soil's hydraulic properties which control the partition of rainfall into infiltration and runoff at the soil surface (Gonzalez-Sosa *et al.* 2010). For example, Pande and Yamamoto (2006) measured the effects light (equivalent to 300 cows/ha) and heavy (600 cows/ha) trodden sites had on saturated hydraulic conductivity (K_{sat}; drainage capacity of the soil) against non-trodden sites on two types of sloping (moderately; 4-15° and steep; 16-30°) Japanese mountain pastures. The soil type at the location was in relatively good condition with a soil bulk density between 0.7-0.8 g/cm^3 and soil porosity of about 70%. Measurements for K_{sat} for the non-, lightly- and heavily-trodden sites were 140, 0.65, 0.25 mm/hr, respectively, indicating increased compaction of topsoil with increased hoof pressure. In turn, an increase in soil compaction can increase the potential of surface runoff (Batey 2009). Surface runoff can be generated by either saturated- or infiltration-excess conditions and soil compaction can further encourage the runoff generation of both processes.

Other negative changes that occur as a result of animal treading include the reduction or depletion of pasture growth which negates the benefit that tillers provide in trapping soil particles and preventing soil erosion (Nash and Halliwell 1999). This is particularly so for dry regions such as savannas in West Africa where soil erosion is common and short-term duration grazing encouraged to avoid erosion events (Savadogo *et al.* 2007). As well as having detrimental impacts on infiltration rates and promoting surface runoff, livestock treading should never result in an effective soil bulk density above 1.7 g/cm^3, a value reported to have negative effects on microbial biomass and carbon-mineralisation (Beylich *et al.* 2010). However, not all impacts of livestock treading are negative. For example, by causing a decrease in surface roughness, compaction can decrease soil surface water repellency (Bryant *et al.* 2007).

Regarding the impact they have on soil physical quality, stock type can play very different roles and are considered separately below.

2.2.1 Impacts of cattle on soil physical quality

Results from a pasture trial in Ireland showed that unrestricted access of cattle to a pasture plot caused a 57-83% decrease in macroporosity, 8-17% greater soil bulk density and 27-50% higher resistance to penetration than areas where cattle were excluded, across a range of sandy loam and clay loam topsoils (Kurz *et al.* 2006). Similarly, soil bulk density in cattle grazed sites was reported to be 0.37-0.47 g/cm^3 greater than sites not grazed by cattle for over 6 or 26 years in Madera, California (Tate *et al.* 2004). Furthermore, there were no significant differences in soil bulk density between sites ungrazed by cattle > 26 years and sites ungrazed for > 6 years.

Seasonal dynamics in the Eastern Alps of Austria have recently been revealed for grassland ecosystems (Leitinger *et al.* 2010). For pastures, cattle trampling caused significant increases in soil bulk density of up to 0.33 g/cm^3 and more than 60% decreases in infiltration rates than abandoned areas in the alpine catchment. However, these authors suggested that soil bulk density would likely recover as a result of freeze-thaw and bioturbation cycles during the winter season when it is common practice to remove stock. Stock exclusion can also result in soil physical and pasture recovery. For example, stock-excluded pasture improved the quality of several environmental variables in southern Alberta, Canada (Miller *et al.* 2010). Improvements included increased vegetative cover (13-21%) and standing litter (38-742%) and decreased the exposure of bare soil (72-93%) and soil bulk density (6-8%).

In contrast to the study reported by Leitinger *et al.* (2010) whereby soil physical recovery was expected to occur during the winter season, soil physical damage is likely to worsen in southern regions of New Zealand, where it is common practice to graze cattle in winter when soil is wet. Although the extent of soil damage is soil type dependent, winter pugging can decrease pasture yield by 20-80% (Ledgard *et al.* 1996) and can result in 40-50% reduction in subsequent regrowth (Mickan 1996). Therefore, it has been recommended that dairy cows are removed from the milking area in winter, so pastures are not grazed or treaded upon. In New Zealand, cattle are usually winter-grazed on different farms, or on forage *Brassica* crops which can enhance surface runoff [(Drewry 2006), and see section 2.3.1]. Grazing of 'carry-in feed' in woodland can also be quite common particularly in winter to decrease the damage to pasture in paddocks. However, in a simulated cattle compaction study in Italy, intense and repeated compaction reduced organic matter content, increased soil bulk density and had a significant negative effect on root development in woodland. Compaction of all levels (light and intense) reduced water infiltration capacity and increased penetration resistance (Ferrero 1991).

When the impact of sheep and cattle treading were investigated on hill country soil in southern Hawkes Bay, New Zealand, very little surface soil disturbance was measured from sheep treading compared to the significant upward and downward movement of soil caused by cattle (Betteridge *et al.* 1999). Overall, the impact of treading damage on soil surface configuration, with soil moisture wetter than the plastic limit, was greater by cattle than by sheep stocked at the same stocking units per hectare although net disturbance (compaction) of sheep pasture was more than twice that on cattle pasture.

The combination of both cattle grazing and irrigation in New Zealand was found to be more destructive from a soil physical perspective with a lower macroporosity (9% v/v) compared to 18% v/v for dryland sheep grazing (Houlbrooke *et al.* 2006). Such values of macroporosity had no significant effect on pasture yield to date under cattle grazing but can influence runoff processes (see section 2.3.2). Furthermore, the grazing of cattle on winter forage crops caused significantly (P < 0.01) lower macroporosity than that of sheep grazing for two out of three years and greater soil bulk density measurements from cattle for all three years (Houlbrooke *et al.* 2009b).

2.2.2 Impacts of sheep on soil physical quality

In general, recovery and reappearance of damaged and buried tillers, and recovery of soil structural damage have been found to be remarkably rapid under sheep grazing (Drewry *et al.* 1999; Edmond 1973), compared to cattle grazing (Drewry 2006). In saying that, Albon *et al.* (2007) reported that when the trampling impacts of sheep, deer, and cattle on rangeland habitats in Scotland were investigated, sheep were found to exhibit the greatest negative impact. This was attributed to the greater aggregation of sheep because of their limited ranging behaviour which can be exacerbated when sheep are herded into places convenient for farmers. Furthermore, desertification has been associated with historic sheep grazing in Northern China and accompanied by severe soil erosion, soil nutrition decline and species diversity loss (Huang *et al.* 2007). The conversion of soil fractions, such as the removal of fine silts and increase in sands, resulted in a coarser soil surface and a consequent decline of soil fertility as the natural grassland shifted to a more desertified landscape. As desertification intensified soil bulk density increased and soil porosity decreased. Reductions of 94%, 89% and 69% of soil organic matter occurred in the 0-5, 5-10 and 10-30 cm soil depths. Overall, these findings were concluded to result in numerous ecological and environmental issues.

When 97 sheep and 87 dairy farms were surveyed in Southland and South Otago, New Zealand, soil macroporosity, soil bulk density, air permeability and hydraulic conductivity were compared between both stock and four contrasting soil types (Drewry *et al.* 2000a). For all soil physical properties, dairy farms were in a worse state than sheep farms. However, the degree of damage caused by each stock class was soil type dependent (Table 2.1) with the Fragic Pallic soils experiencing greater physical damage compared to the Firm Brown soils. The former have previously been regarded as having poor drainage and structural properties and the latter considered to be well structured (Greenwood 1999; Hewitt and Shepherd 1997).

Table 2.1. Classification of extensive soils in New Zealand for susceptibility to treading damage by plastic deformation and resilience to damage.

Soil	***Susceptibility***	***Resilience***
Momona (Recent)	High	Low
Mataura (Recent)	High	Low
Cargill (Brown)	Medium	High
Mokotua (Brown)	Medium	Medium
Waikiwi (Brown)	Medium-high	Medium
Warepa (Pallic)	High	Low
Pukemutu (Pallic)	Medium	High

Climo and Richardson (1984) compared three soil types under sheep grazing during heavy rainfall in New Zealand. All three soils had decreased macroporosity after compaction with the Tokomaru silt loam being most susceptible as result of poor drainage. The Ramiha silt loam was least susceptible to compaction because of its better structure and drainage properties, and the Manawatu fine sandy loam suffered intermediate damage exhibiting both good drainage but weak structural properties.

Soil bulk density in a Pukemutu silt loam soil [Mottled Fragic Pallic soil (Hewitt 1998)] in New Zealand has been shown to significantly increase after intensive sheep grazing in winter at the 0-5 cm soil depth (Drewry *et al.* 1999). Plastic deformation, rather than compaction, was found to be the major process responsible for damage. However, although the winter stocking density of 1800 sheep/ha had detrimental impacts on soil physical quality, no long term negative effects were observed after two growing seasons for pasture yield on the site.

When set-stocking and rotational grazing by sheep were compared in New South Wales, Australia, there were no significant differences between the two treatments for soil bulk density, hydraulic conductivity and organic carbon concentration. However, significant differences were found from image analysis of soil macroporosity with set-stoking resulting in a decrease in total macroporosity (Cattle and Southorn 2010). Within the study, macroporosity was partitioned into two size ranges defined with a diameter 0.065-1.5 mm and >150 mm. Structural quality was considered 'best' under pasture cages where pasture was grazed in the absence of hoof pressure. Similarly, when historic sheep grazing at Desert Steppe in Inner Mongolia was investigated grazing increased soil bulk density when compared to an ungrazed treatment (Li *et al.* 2008). In contrast, du Toit *et al.* (2009) reported that when various stocking rates of sheep (4, 8, 16 small-stock units [SSU]/ha) were compared to ungrazed rangeland, the light stocking (4 SSU/ha) treatment influenced both infiltration rates and soil bulk density most favourably. Light stocking reportedly loosened the topsoil sufficiently to increase initial infiltration rates. Infiltration rates for light stocking and heavy stocking (16 SSU/ ha) was 17% higher and 14% lower, respectively, than ungrazed rangeland over two growing seasons of Arid Namo Karoo vegetation in South Africa.

After over 13 years of rotational sheep grazing in New South Wales, Australia, results showed that soil bulk density measurements decreased as the intensity of the sheep camping increased (Niu *et al.* 2009). Soil bulk density measured at 0-5 and 5-10 cm soil depths at the camp site were 0.96 g/cm^3 and 1.34 g/cm^3, respectively, which increased to 1.14 g/cm^3 and 1.39 g/cm^3 20 m away from the camp site on perennial pastures at corresponding soil depths. Likewise, mean pasture dry matter was greatest within 5 m of the camp site and least 20 m away from the camp site. The authors concluded that the rich pasture growth and high organic matter within the camp site provided a buffer causing the soil within the camp site to be more resistant against the impacts of soil compaction thereby decreasing soil bulk density within 5 m of the camp compared with 20 m away.

After cessation of sheep grazing for 45 years in Canterbury, New Zealand, soil physical quality improved slowly at the 0-5 cm depth compared to that of grazed pasture (Basher and Lynn 1996). The exclusion of grazing allows soil properties to naturally recover, but is most likely to be restricted to the top 10 cm or at the most 15 cm (Drewry 2006).

2.2.3 Impacts of deer on soil physical quality

Only a limited number of studies have investigated the impact of deer on soil physical quality (Kumbasli *et al.* 2010; McDowell *et al.* 2004b; McDowell and Paton 2004; McDowell and Stevens 2006; McDowell and Stevens 2008). Kumbasli *et al.* (2010) investigated the effects of red deer on various properties of the 0-5 cm soil depth by comparing a breeding area with an undisturbed area in Istanbul Belgrad Forest, Turkey. Soil was significantly compacted with reduced litter mass and organic carbon content within the breeding area. The available water holding capacity of the soil in the breeding area was significantly lower and soil bulk density significantly higher as a result of long term deer grazing when compared to the undisturbed area.

Behavioural characteristics exhibited by deer include fence-line pacing and wallowing, the former occurring when deer experience periods of stress e.g. during food shortages and calving (McDowell and Stevens 2006). Soils along fence-lines have been found to have a macroporosity about one quarter (4.4% v/v) of that from soils in other areas of the paddock (16.4% v/v) (Pollard and Drewry 2002). In another study at Invermay deer farm, Mosgiel, New Zealand, macroporosity was 6.8% v/v along fencelines compared with 16.1% v/v from the 'rest of the paddock' a day after deer grazed which increased to 6.4% v/v and 18.9% v/v, respectively, six weeks after deer grazed (McDowell *et al.* 2008). Furthermore, K_{sat} was observed to be 58 mm/hr along fencelines and 692 mm/hr from the 'rest of the paddock' one day after deer grazed (McDowell *et al.* 2008). Values <1 mm/hr for K_{sat} along well established fencelines have also been recorded in an adjacent paddock at Invermay deer farm where the topsoil had been completely eroded. In contrast to deer grazing and the impacts it has on fence-line pacing, soil physical quality is likely to be at its least compacted state under fence lines in pastures with cattle grazing (Drewry 2006), with cattle preferring to walk on animal tracks/camps and easy contoured areas (<25°) (Sheath and Carlson 1998).

2.2.4 Recovery of soil physical quality

Natural recovery of soil physical quality is the amount of time it takes to improve the structural and physical damage of soil from the adverse effects of livestock compaction and pugging (Drewry 2006). Natural recovery processes include freeze-thaw, wetting and drying cycles and earthworm activity, with the extent of improvement on soil physical quality varying from weeks to years (Drewry 2006). The degree of recovery can vary from site to site depending on many factors such as previous land management practices, soil type, and climate (Drewry 2006). For example, Drewry and Paton (2000) noted an 88% improvement in macroporosity after four months from cattle grazed vs. ungrazed plots compared to a 44% improvement in macroporosity after five months as a result of reduced levels of grazing and stock exclusion (Drewry *et al.* 2004). Although soils in both studies were silt loams, soil type and structure was a major factor in determining the degree of improvement. Similarly, soil bulk density improved 32-58% after a 12 month post graze/damage interval (i.e. nil grazing) from a cattle grazed vs. ungrazed land management practice (Stephenson and Veigel 1987), compared to a 10% improvement in soil bulk density after a post graze interval of 45 years from a sheep grazed vs. ungrazed land management practice (Basher and Lynn 1996). Again, soils and sites were very different in both studies with the soil classed as a Mollisol in Idaho, USA and a Dystochrept in Canterbury, New Zealand.

Natural recovery processes are limited to soil depths of 10 cm with a maximum of 15 cm (Drewry 2006), whereas mechanical loosening of the soil, which can also off-set the adverse effects of soil compaction and pugging, can easily reach soil depths of 24 cm

(McDowell *et al.* 2008). In Southland, New Zealand, Drewry *et al.* (2000b) investigated the effects of shallow mechanical loosening, otherwise known as subsoiling or aeration, on soil physical quality of a Pallic soil. Subsoiling dramatically increased macroporosity, saturated hydraulic conductivity, and air permeability and decreased soil bulk density. Improvements were evident up to 2.5 years later at 18-24 cm soil depth, although some recompaction and settling occurred in the upper 18 cm of the soil profile. These authors recommended subsoiling on sites that compact with macroporosity values < 10% v/v. Burgess *et al.* (2000) reported that subsoiling on a Te Kowhai silt loam initially showed decreased soil bulk density and penetration resistance and increased hydraulic conductivities, total porosity, macroporosity, and the proportion of small aggregates compared with undisturbed soil after dairy cattle grazing. However, soil bulk density, proportion of small aggregates, total porosity, saturated and unsaturated hydraulic conductivity reverted back to undisturbed levels after ten months. Other work has shown that the benefits of subsoiling can be soil type dependent with some soils quickly reverting back to its original state shortly after subsoiling (Houlbrooke 1996), particularly dispersive soils (Adcock *et al.* 2007). Although informative, these studies have neglected to report upon environmental impacts such as contaminant losses associated with mechanical aeration, an area where our understanding is limited.

Results from studies in America on contaminant losses associated with mechanical aeration are variable and appear to be soil type dependent. For example, both Shah *et al.* (2004) and Franklin *et al.* (2007) have reported reductions in both surface runoff volume and P losses for aerated soil when compared to non-aerated soil for well-draining soils in West Virginia and Georgia, respectively. In contrast, soil aeration was reported to increase runoff volumes and P losses compared to non-aerated soil for poorly draining soils (Franklin *et al.* 2007). In New Zealand, Houlbrooke *et al.* (2005) reported that the loss of farm dairy effluent (FDE) in surface runoff was less from aerated soil than from non-aerated plots due to improved surface infiltration rates in the former treatment. However, studies reporting the impacts of mechanical soil aeration have failed to assess the influence soil aeration have on both soil physical quality and contaminant losses, both of which are directly related to each other (McDowell *et al.* 2003b).

2.3 Animal grazing and phosphorus losses in surface runoff

Sources of P available to surface runoff as a result of animal grazing are plant, soil, fertilizer, or manure (Nash and Halliwell 1999). Numerous studies have shown that animal grazing increases the loss of P and SS in surface runoff (Table 2.2). Similar to soil physical quality, different stock types have very different influences on P losses in surface runoff. For example, cattle and sheep have been reported to defecate 11-16 (28.4 kg/day) and 7-26 (1.6 kg/day) times a day, respectively (Haynes and Williams 1993). This equates to about 5.5 and 8 g/kg dry matter of total P for cattle and sheep dung , respectively (McDowell 2006a). Although having least total P on a dry weight basis, the risk of contaminant loss posed by cattle dung was found to be greater than that posed by either deer or sheep dung, which was attributed to having the greatest surface area of all dungs exposed to rainfall (McDowell 2006a). The following sections give an account of the impacts cattle, sheep and deer grazing have on P losses in surface runoff.

Table 2.2. Mean concentrations of total phosphorus (TP), dissolved reactive P (DRP) and suspended sediment (SS) losses determined in surface runoff from various studies predominantly in New Zealand.

Region	Stock type	Forage type	DRP (mg/L)	TP (mg/L)	SS (g/L)	Soil Olsen P (mg/ kg)	Annual P fertilizer (kg P/ha)	Reference
Mossburn, NZ[1]	Deer	pasture	0.063	0.35	0.085	50-59	20	(McDowell *et al.* 2006b)
Invermay, NZ	Deer	pasture	0.195	1.173	0.185	34-53	34-53	(McDowell and Paton 2004)
Hillend, NZ	Cattle	crop	0.579	8.078	5.26	15-34	34	(McDowell *et al.* 2003a)
Hillend, NZ	Cattle	pasture	0.181	0.593	0.013	21	34	(McDowell *et al.* 2003b)
Hillend, NZ	Cattle	crop	0.054	1.041	0.063	21	34	(McDowell *et al.* 2003b)
Edendale, NZ	Cattle	pasture	0.21	0.944		24	50	(Smith and Monaghan 2003b)
Whatawhata, NZ	Cattle	pasture		2.998	0.059	25	25	(Nguyen *et al.* 1998)
Ballantrae, NZ	Cattle	pasture		0.31	0.62	7	12-86	(Lambert *et al.* 1985)
Ballantrae, NZ	Sheep	pasture		0.161	0.339	7	12-86	(Lambert *et al.* 1985)
Waikataki, NZ	Cattle	pasture	1.258	1.49		45	25	(Carey *et al.* 2004)
Tussock Creek, NZ	Cattle	pasture	0.449	0.923	0.079	40	53	(Houlbrooke *et al.* 2009a)
Kelso, NZ	Cattle	pasture	0.018	0.103	0.015	59-65	7-50	(McDowell *et al.* 2004c)
Vic[2], Australia	Cattle	pasture		4.235		30	25	(Barlow *et al.* 2005)
Vic, Australia	Cattle	pasture	0.743	1.055		33	42	(Mundy *et al.* 2003)

[1] New Zealand

[2] Victoria

2.3.1 Impacts of cattle on losses of phosphorus

Cattle have been reported to utilize water filled features for cooling down purposes more during the warm season (summer and autumn) in Florida, USA (Pandey *et al.* 2009) and defecate *c.* 50 times more per metre of water crossing than dry pathways (Davies-Colley *et al.* 2004). The average percentage of daily time spent by cattle near/in water locations (water trough, wetland, and ditch) during the warm season was 11.45 ± 0.39% and 6.09 ± 0.69% during the cool season (winter and spring) (Pandey *et al.* 2009). This summer cattle behaviour has been reported to have detrimental impacts on water quality in the Sierra Nevada Mountain range in California as cattle dung gets washed into lakes and streams promoting eutrophication in an otherwise oligotrophic environment (Derlet *et al.* 2010). To overcome such water quality issues these authors proposed limiting summer cattle grazing on public lands to lower elevations. Results from Minnesota suggested that although the exclusion of riparian cattle grazing is preferable, short-term duration grazing could also effectively decrease sediment pollution if well-managed (Magner *et al.* 2008). Other authors in Georgia, USA, have reported that the development of nonriparian shade areas and the establishment of water sources, such as water troughs, away from streams would minimise P loss from cattle grazed pastures by reducing the stress levels of cattle during these water shortages (Byers *et al.* 2005), although this may cause issues with landowners due to cost of increased labour and maintenance. Vast improvements have been reported as a result of such developments away from the vicinity of lakes and streams and stock excluded waterways. For example, Line *et al.* (2000) reported a 76% reduction in TP loss after a 137 week post-exclusion fencing-off of cattle treatment and the planting of trees along a stream in North Carolina, USA.

In the South Island of New Zealand it is common practice for animals to be strip-grazed on a forage crop such as swede (*Brassica rutabaga* L.) in winter. Under these conditions, grazing can cause treading damage that adversely affect soil properties and often leads to P and SS losses during a time of year when conditions are wet (McDowell *et al.* 2003a). A plot scale field trial at Hillend also indicated significant TP losses under cropland, Table 2.2 (McDowell *et al.* 2003a). However, when cattle treaded pasture and cropland were compared it was found that more P in the form of DRP was lost in grassland soil (0.22 mg/L after 30 imprints/ m^2) compared to that lost in cultivated soil (0.06 mg/L, and decreasing, after 30 imprints/ m^2) (McDowell *et al.* 2003b). The crushing and burial of grass after animal trampling was likely to have increased the P lost from plant cells. Other studies in New

Zealand reported a 94% increase of P loss in surface runoff from trodden grassland soil when compared to undamaged areas (Nguyen *et al.* 1998).

Phosphorus strongly sorbs to soil mineral surfaces and losses of particulate P (PP) can occur via erosive processes when the soil is not protected by grass cover (Quinton *et al.* 2001). Therefore, as the proportion of bare ground increases, as a result of the loss of grass cover due to animal trampling, erosion of PP forms increases, thereby reducing the proportion of TP loss in dissolved forms. This has been found to be the case in cultivated soil in New Zealand where PP forms account for most of TP losses (McDowell *et al.* 2003a). In conclusion, 'enhanced erosion of PP from cultivated soil means that a greater load of TP is lost from cultivated soils than grassland soils' (McDowell *et al.* 2003a).

In another study in New Zealand, large TP and DRP losses in surface runoff from a poorly structured undrained soil was associated with an increase in stocking density (Smith and Monaghan 2003a). It was concluded that the losses of P were the result of the return of nutrients via dung and the detrimental damage caused to the soil structure that is often observed during grazing under wet soil conditions. In South Carolina, runoff volumes have been reported to be greater for poorly drained soils than well drained soils (Butler *et al.* 2008b). Volumes were further exacerbated for a heavily cattle grazed treatment (heavy-use) when compared to a lighter cattle grazed treatment (light-use). Greater SS and TP losses occurred for heavy-use plots than light-use plots for both soils. Overall results revealed that heavy-use areas on poorly drained soils pose the greatest P and SS export risk (Butler *et al.* 2008b). In Ireland, it was reported that more P, mainly in the form of organic P, was lost when cattle had unrestricted access to plots compared to when cattle could graze the plots but could neither walk nor defecate on the plots (Kurz *et al.* 2006). Similarly, Heathwaite and Johnes (1996) described how the unreactive (particulate and organic) P form contributed to greatest surface P runoff in a plot scale study in the UK which significantly increased under heavy dairy and beef cattle grazing (> 15 stock per hectare; 249 mg P/ m^2 PP + DOP) compared with lightly grazed grassland (< 4 stock per hectare; 7 mg P/ m^2 PP + DOP).

McDowell (2006a) reported that DRP and TP lost from dung posed a threat to surface water quality if surface runoff occurred within a grazing period of 30 days. However, greatest risk occurred in the first few days after deposition when the cattle dung was wet, a time when the risk of P losses in surface runoff could be minimized by restricting grazing on wet soils. Phosphorus in dung has been found to be very mobile (McDowell and Sharpley 2002). It was found that all forms of P [dissolved, particulate, inorganic (reactive), or organic] lost from fresh dung in surface runoff was more than double the P lost from either the cultivated or pasture treatments (McDowell *et al.* 2003a). Dairy cattle dung was found to pose greater risk than either deer or sheep dung for DRP, dissolved unreactive P (DUP), total dissolved P (TDP), particulate P (PP), TP, SS, ammoniacal-N (NH^{+}_{4}-N), nitrate-N (NO^{-}_{3}), and *Escherichia coli*. However, DRP loss from sheep dung was found to be the most persistent (McDowell 2006a).

2.3.2 Impacts of sheep on losses of phosphorus

Few studies have focused specifically the impact sheep grazing have on P losses in surface runoff. When volumes of surface runoff and losses of TP, DRP, and sediment were compared, cattle grazing produced significantly greater losses of all pollutants under cropland when compared to sheep grazing under cropland (88 mm and 67 mm surface runoff, respectively) in North Otago, New Zealand (McDowell and Houlbrooke 2009). In the present review only one study was considered regarding the impact sheep and cattle grazing had on nutrient losses (Table 2.2). Cattle grazing resulted in twice (1.5 kg P /ha/year) the amount of nutrients lost than that of sheep (0.7 kg P /ha/year) (Lambert *et al.* 1985). Furthermore, there were no significant differences found in losses of P when rotational grazing of sheep was compared with the set stocking of sheep.

Li *et al.* (2008) reported that vegetative TP concentrations were not affected by either light, moderate or heavy sheep grazing when compared to ungrazed pasture. However, the heavily grazed treatment resulted in 7.6% and 16.4% reduction in soil TP concentrations and soil organic matter, respectively, when compared to ungrazed pasture. This was associated with a reduction in defoliation with grazing. A pasture trial in New South Wales, Australia, described how Colwell P increased from 44 mg/kg 20 m away from a sheep camping site to 125.9 mg/kg in the vicinity of the camp site (Niu *et al.* 2009). High TP concentrations in runoff have been linked with greater labile P contents in the soil, dung and herbage, all of which are associated with sheep camping (Melland *et al.* 2008). The complete decomposition of sheep dung can take over 75 days in summer and changes in concentrations of TP in sheep dung with time are minimal (Rowarth *et al.* 1985). Minimising sheep camping was recommended by Melland *et al.* (2008) to reduce labile P losses to freshwater systems in southern Australia.

In Northern England, two adjacent catchments grazed by sheep were compared which employed two different improved grassland treatments (7 and 47% of total area) (Withers *et al.* 2007b). Pasture improved by lime and P fertilisers increased the losses of DRP and PP, but not SS. The authors suggested that increased P losses were likely to be associated with soil P enrichment rather than those coupled with fertiliser application or increased sheep stocking densities (1-2.5 ewes/ha). Further, the loss of P in sheep excreta in streams or areas close to streams contributed to increased DRP and PP concentrations during stormflow and sometimes during baseflow events.

In a catchment study in New Zealand, grazed at approximately seven sheep per ha and 0.7 cattle per ha, 76% of the annual discharge of TP was PP. The correlation of PP and SS concentrations was highly significant and a major influence to losses was attributed to the impacts of grazing causing the exposure of bare ground from heavy grazing of forage crops (Bargh 1978). In a more recent study, 38 catchment studies were analysed to determine if there were differences in the loads of contaminants lost from varying land uses under different livestock (dairy, sheep, sheep and beef (mixed), and deer). Loads of P lost from deer and mixed stock were greatest and of equal importance followed by dairy and least from sheep (McDowell and Wilcock 2008).

2.3.3 Impacts of deer on losses of phosphorus

In addition to treading damage and deteriorated soil physical quality near fencelines, deer wallows have also been shown to be a major source for water contamination (McDowell 2007; McDowell 2009; McDowell and Wilcock 2008). Between 60-90% of annual P losses to streams have been associated with deer wallowing even though deer grazed for only half a year. Recommendations reported that the fencing off of old wallows and the formation of a 'safe' wallow, defined as a wallow unconnected to a stream, provided a decrease of up to 90% in the loss of P and SS contaminants (McDowell 2009).

Another study examined the presence or absence of shelter belts in deer farms to determine whether they decreased fence line pacing by reducing stress levels via creating cover and shelter (McDowell *et al.* 2006b). Results showed that shelter belts had no effect on the concentrations of P lost in surface runoff or any soil physical parameters. The authors concluded that future research would be required to collect comparable data between different soil types.

On balance, regardless of stock type and its impact on surface soil, Edmond (1966) concluded that "…accumulated evidence indicates that treading effects in pastures must be considered", whether be sheep, cows, or deer.

2.3.4 Soil physical quality and phosphorus loss

Data from studies around New Zealand were compared to examine how different stock types (sheep, beef cattle, dairy cattle and deer) and soil physical quality influence losses of total P (Figure 2.2a and 2.2b). Data in Figure 2.2 was generated from several studies that were reviewed in Table 2.2 and include Houlbrooke *et al.* (2009a), Nguyen *et al.* (1998), McDowell *et al.* (2006b) and (2003a), McDowell and Paton (2004) and Smith and Monaghan (2003b).

Significant relationships were observed for soil bulk density (P < 0.001) and Log_{10} K_{sat} (P < 0.05) for losses of TP in surface runoff. Figure 2.2a illustrates that an increase in soil bulk density increased TP losses in runoff. Conversely, a decrease in $\log_{10}$ K_{sat} increased TP losses in runoff, Figure 2.2b.

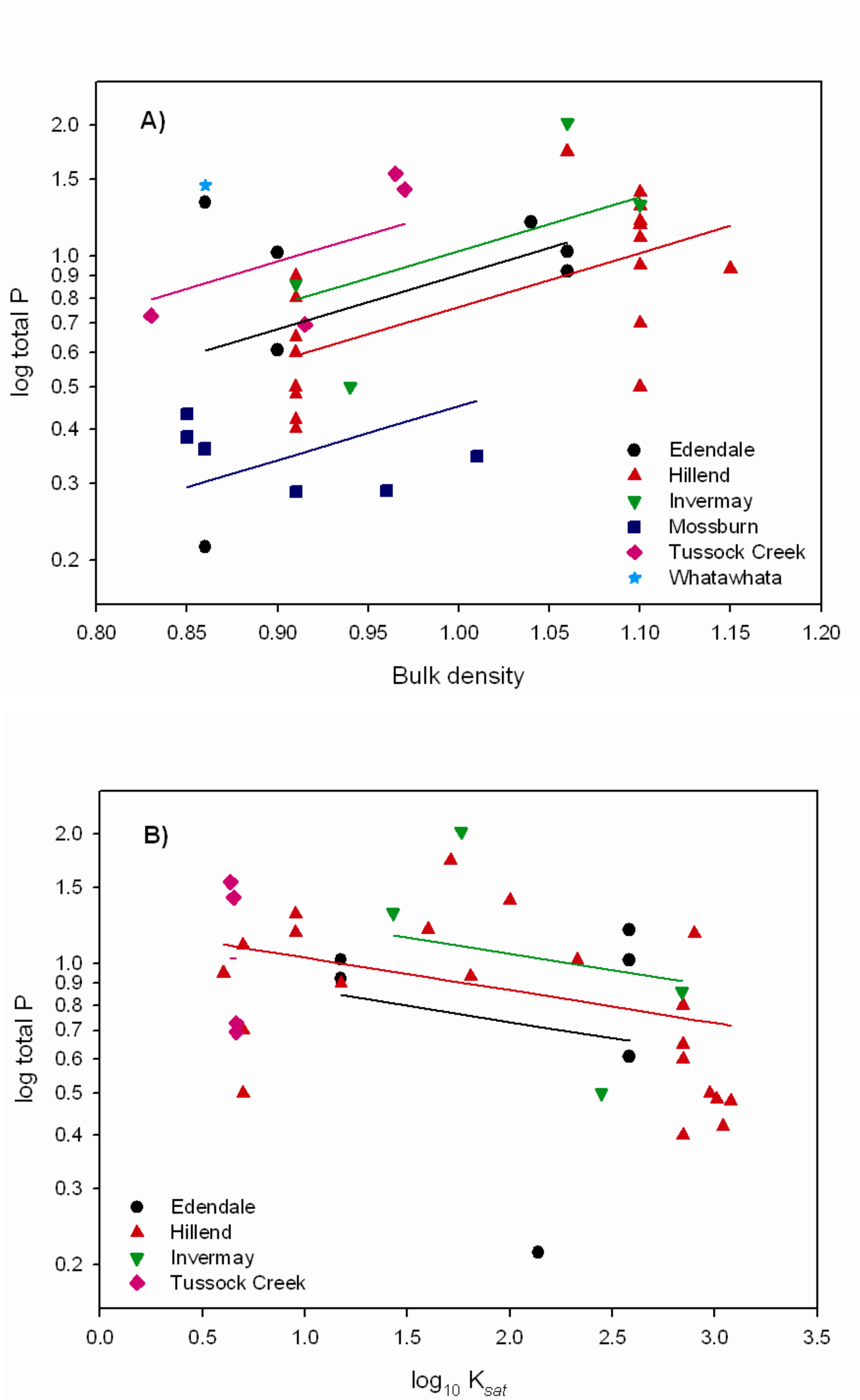

Figure 2.2. Significant relationships determined between (A) soil bulk density (P < 0.001) and log total P (mg/L) losses in surface runoff and between (B) $\log_{10}$ saturated hydraulic conductivity ($\log_{10} K_{sat}$) (P < 0.05) and log total P (mg/L) losses in surface runoff.

Soil bulk density was reported to have increased more so near fencelines compared to the rest of the paddock on deer farms, an area where accelerated P loss occurs in surface runoff (McDowell *et al.* 2006b). As soil bulk density increases under compaction (McDowell *et al.* 2003a; McDowell *et al.* 2003b) it decreases the infiltration capacity (K_{sat}) and pore volume of the soil leading to increased surface TP runoff and erosion. Significant relationships have also been observed for macroporosity and K_{sat} with time to ponding, volume of surface runoff produced, and grazing intensity (McDowell *et al.* 2003b). Damage to soil physical quality from treading will inevitably lead to the transfer of SS and P regardless of whether soils have been cultivated or in pasture (McDowell *et al.* 2003a; Nguyen *et al.* 1998).

2.4 Animal grazing and suspended sediment losses in surface runoff

Soil erodibility is a complex process and is influenced by many soil properties including its texture, structural stability, organic matter concentration, clay mineralogy, and chemical constituents. Many of these properties can be altered such as organic matter, particularly under livestock grazing (Lal and Elliot 1994). Soil texture is an important factor when determining erodibility. Sandy soils tend to have lower runoff rates and are easier detached, but less easily transported, than silt soils. The opposite is true for clay soils which are not easily detached but have lower infiltration rates which in turn may lead to larger runoff rates and consequently increased erosion (Lal and Elliot 1994). Many factors contribute to the loss of sediment, including soil compaction as measured by an increase in soil bulk density, soil strength (resistance to compaction) and cohesion, as well as decreased compressibility and infiltration (Climo and Richardson 1984). Other factors include the direct effect of increased physical and visual disturbance with contact between the soil and hoof as well as soil vegetative cover (Sanjari *et al.* 2009).

Evans *et al.* (2006) studied how sediment loads and sources degraded the habitat and spawning grounds of one of Ulster’s prime salmon rivers (the River Bush) in Northern Ireland. The authors concluded that the combination of flow damage and livestock pugging to riverbanks were one of the four major agents generating stream sediment load. The other agents included drainage maintenance work, ploughed arable land and clear felled forestry. Livestock pugging and peak flows contributed to approximately 2% of the annual suspended sediment (SS) and 60% of the annual bed load. Livestock pugging was a major activity that caused the most frequent occurrence of bare soil, although the largest tracts of bare ground were caused by tillage and forestry in the catchment. The authors recommended restricted access of livestock to control the loss of such sediment loads.

Many studies have investigated the rates of SS losses resulting from livestock grazing and, again, the impacts of different stock type on SS losses in surface runoff are reviewed separately below.

2.4.1 Impacts of cattle on suspended sediment losses

When comparing the quantities of SS lost at Hillend, Otago, Table 2.2, (McDowell *et al.* 2003b), it was found that more SS was lost under cattle grazed winter forage crop (0.063 g/L) than grazed grassland (0.013 g/L). An increase in sediment loss with increased treading pressure has been noted in many other studies e.g. (Nguyen *et al.* 1998). As more grass is crushed and buried with increased treading, the positive attributes of grass cover in the trapping efficiency of sediment are negated allowing rainfall to impact upon the soil surface (Nash and Halliwell 1999).

In a catchment-scale study in south-western England, erosion rates and yields of SS were assessed from improved or intensively managed grassland grazed by beef, dairy and sheep (Bilotta *et al.* 2010). It was reported that improved grassland can yield between 0.54-1.21 t ha/yr of SS losses. Concentrations of SS amounted to 1140 mg/L, averaging at 65 mg/L, several orders of magnitude greater than the recommended guidelines set by the European Freshwater Fisheries Directive (EU FFD). Guidelines specify that losses greater than 25 mg/L cause harmful impacts to salmonid and cyprinid fish populations. Similarly to TP losses, Line *et al.* (2000) reported an 82% reduction in SS loss after a 137 week post-exclusion fencing treatment of cattle and the planting of trees along a stream in North Carolina, USA. Likewise, Byers *et al.* (2005) suggested that the development of nonriparian shade areas and the establishment of water sources away from streams would significantly reduce SS loss from cattle grazed pastures in Georgia, USA.

Russell *et al.* (2001) concluded that there was a decrease in SS lost with increasing canopy height after cattle treading damage. Canopy heights, as low as 20 mm, could minimize surface SS loss. In contrast, Pande *et al.* (2002) reported that the presence of a taller canopy did not protect basal or tiller density from the adverse effects of cattle trampling and SS losses. Although animal treading can initially result in an increase in SS losses due to the disturbance of the soil, Russell *et al.* (2001) reported that further damage can decrease the amount of soil lost. These authors concluded from their results at Ballantrae, New Zealand, that although treading damage increases the number and area of clumps from which SS can erode, it can also decrease the length of slope and increase the volume of storage that can detain detached soil particles.

Mwendera *et al.* (1997) revealed from their results that as biomass utilisation and animal trampling increased, the rates of soil lost also increased in the Ethiopian highlands. Erosion rates were also shown to increase exponentially with grazing pressure and increased slope. They explained that heavily grazed pastoral lands above 5% slopes result in serious soil erosion over time if left unattended. This problem will also be shared on lower slopes with time. It was concluded that the problem of erosion with increased slope could be avoided if livestock were kept away from steep slopes during wet seasons, although extra feed to be cut and carried would be required which could conflict with labour allocation. Alternatively, these authors recommended that fertiliser application would increase vegetative growth of pastures on upper slopes reducing the exposure of bare ground and subsequent soil erosion.

In a New Zealand study, it was found that SS losses were greatest under cattle grazing (2740 kg/ha/year) when compared to sheep grazing on pasture (1220 kg/ha/year) (Lambert *et al.* 1985). The loss of SS under cattle grazing was visually evident during the trial period with increased losses occurring when soils were wet, runoff greatest, and pugging damage by cattle hooves most severe. Visual pastoral damage was so severe that the density of cattle grazing had to be decreased.

2.4.2 Impacts of sheep on suspended sediment losses

Although, sheep treading has a lesser impact on soil physical quality than that of heavier livestock such as cattle, they too have shown to have detrimental effects on soil health, pasture growth and soil erosion (Climo and Richardson 1984; Drewry *et al.* 1999; Edmond 1962; Greenwood and McNamara 1992). As noted previously in Table 2.1, soil type is a major determinant when assessing the grazing impact livestock have on soil physical quality and losses of P and SS. The Desert Steep in inner Mongolia was reported to be vulnerable to soil erosion which was majorly attributed to its sandy soil content (70%) as well as the generally low litter content and plant cover as a result of its extensive sheep grazing history (Li *et al.* 2008). Ensuring appropriate grazing pressure and animal distribution in combination with well-managed herding was recommended to reduce the risk of soil erosion.

When Bargh (1978) investigated the losses of SS from a small agricultural catchment in New Zealand, grazed on average by seven sheep per ha and 0.7 cattle per ha, the output amounted to 1.4×10^3 kg SS/ha/year. Losses were dependent on several factors which included antecedent soil moisture, rainfall quantity and intensity. Land management was another primary factor influencing SS losses. Heavy forage crop grazing caused 19 ha of bare ground to become noticeably visible.

With a general reduction of soil moisture during summer periods, root damage can prevail which is unlikely to recover afterwards. Clumps and tufts of shattered roots combined with the loss of soil were observed on pastures with heavily stocked sheep (Edmond 1962). The impact of sheep treading pushes plants down, particularly the highest growing plants, into the ground which encourage erosion by damaging vegetation and increasing the percentage of bare ground. Under wet conditions, the impact of compaction is less of a problem than that of pugging, when the soil pores become water-filled rather than air-filled (Drewry *et al.* 1999). Soil moisture modifies the effects of treading both directly, through visual soil disturbance and indirectly, by impacting soil physical quality and losses of P and SS (Figure 2.2a and 2.2b). The greatest effects occur when soil is wet, but damage is also important when soil is dry (Edmond 1962; Zegwaard *et al.* 2000).

McDowell and Houlbrooke (2009) reported that volumes of surface runoff and loss of SS were greater from sheep-grazed cropland (67 mm and 1.33 g/L, respectively) when compared to sheep-grazed pasture (33 mm and 0.62 g/L, respectively) in North Otago, New Zealand. Lastly, when rotational grazing and set stocking of sheep were compared in New Zealand there were no significant differences in SS losses (Lambert *et al.* 1985).

2.4.3 Impacts of deer on suspended sediment losses

Within deer farms in New Zealand, fence-line pacing is a major source of SS loss in surface runoff (McDowell and Paton 2004). McDowell *et al.* (2006b) hypothesised that shelter-belts may lessen fence-line pacing by decreasing the stress levels of deer. However, the presence or absence of shelter belts made no difference in decreasing the loss of SS in surface runoff. Studies at Mossburn and Invermay New Zealand, revealed that losses of SS were greatest around fencelines than from the rest of the paddock (0.225 g/L and 0.145 g/L, respectively) at Invermay and (0.117 g/L and 0.054 g/L, respectively) at Mossburn (McDowell and Paton 2004; McDowell *et al.* 2006b). Similar to the losses of P at both sites, over twice as much sediment was lost at Invermay than Mossburn. This can be explained by the differences in soil types, with Pallic soil being more prone to erosion than Brown soil, and the fact that Invermay had a greater stocking density. McDowell and Paton (2004) estimated that annually, fence line pacing contributed to about half of the visible erosion from sampled paddocks (0.5-1.0 t/ha), and that wallowing contributed to the remaining erosion loss.

An analysis of 38 New Zealand catchment scale studies reported that deer grazing had the greatest impact on loads of SS losses followed by sheep, mixed (sheep and beef) and dairy stock (McDowell and Wilcock 2008). Fine sediment cover was also reported to be highest in deer streams (88%) followed by dairy (47%), mixed grazed pasture (sheep and beef) (30%), and least from tussock (7%) in New Zealand (Matthaei *et al.* 2006). The addition of sediment caused a reduction of moss cover, invertebrate taxon and diversity richness when compared to controls.

Evans (1996) found that most damage caused by the overgrazing of reindeer *(Rangifer tarandus)* in Norway, occurred alongside an 8.0 km long fence, while trying to manage the reindeer herds in summer. Soil was being eroded at rates of 1-3 mm per year. Although not a large rate, these soils are very shallow and the rates were predicted to increase in the future once the infertile and erodible sands were exposed.

The fencing-off and vegetative planting in areas that are sources of contaminant loss such as wallows or eroded stream bank channels have been recommended to mitigate the excretal input of deer and allow the deposition and settling of SS and associated contaminants (McDowell 2008). McDowell (2009) further recommended the creation of a 'safe' wallow and the fencing off and planting of the old wallow. When put into practice the 'safe' wallow treatment had potential to show significant decreases in the loss of P species and SS.

2.5 Mitigation

This review has outlined the various ways livestock grazing can influence surface water quality via soil physical disturbance which encourages P and SS losses. Best management practices (BMPs) recommend ways to achieve sustainable land use and maintain surface water quality e.g. (Houlbrooke *et al.* 2009a; Monaghan *et al.* 2007a). Several BMPs have been described in the review and others include:

- limiting soil Olsen P to agronomic optimum levels,
- rotational and/or restricted grazing,
- creating larger areas to graze,
- grazing woodlots in winter,
- avoiding the winter grazing of poorly structured soils when wet,
- lowering stocking densities in vulnerable paddocks, and removing stock before the exposure of bare ground e.g. (Betteridge *et al.* 2003; Monaghan *et al.* 2007b).

Houlbrooke *et al.* (2009a) reported that cattle grazing strategies, such as (i) normal grazing management on undrained soil, (ii) normal grazing practice on drained soil, (iii) restricted autumn grazing, (iv) restricted grazing when soil conditions were wet, and (v) never pugged, the never grazed treatment (vi) had significantly greater macroporosity than all the other treatments in Southland, New Zealand. There were no significant differences in any soil physical quality (porosity, soil bulk density, K_{sat}) observed from the cattle grazed treatments. However, another study reported that rotational grazing was far more beneficial protecting against soil erosion when compared with continuous grazing in Queensland, Australia (Sanjari *et al.* 2009). Decreases in SS losses were attributed to an increase in ground cover with rotational grazing and a minimum of 70% surface cover was recommended to effectively protect against soil erosion.

Buffer zones or riparian areas can provide some efficiency in trapping sediment and associated P. The trapping efficiency of buffer zones is primarily associated with rainfall events and may be negated in storm events when channelized flow by-passes strips. If the major fraction of P export is particulate P buffer zones are effective. However, they play a limited role in trapping dissolved P forms which would bypass vegetative buffers (Heathwaite 1997). To maintain effective trapping efficiency of a buffer zone a ‘cut and carry system’ could be implemented whereby the P is removed from the buffer periodically and redistributed across paddocks (Monaghan *et al.* 2007a).

Regarding strategic deer grazing practices the implementation of ‘safe wallows’ have reduced P and SS losses (McDowell 2009). In order to decrease the detrimental impacts of fence-line pacing, land management practices should be directed towards decreasing the stress levels of hinds during calving, increasing cover and shelter, lowering stocking densities and the interaction of deer between adjacent paddocks (McDowell *et al.* 2008).

The effectiveness of BMPs can depend upon many factors such as soil type, previous land management practices, land topography and aspect, climate and stock type and therefore a combination of BMPs may need to be practiced to become effective in reducing the impacts livestock grazing have on soil physical quality and losses of P and SS from land to aquatic environments via surface runoff.

2.6 Conclusions

This review of literature has demonstrated that livestock grazing can have negative environmental impacts on soil physical and water quality. A summary of key findings and recognised research gaps from this review include:

- The very different roles stock type play on soil physical quality and subsequent losses of P and SS. However, research is limited when investigating stock type under the same field conditions (slope, aspect, soil type, soil moisture, food availability, stocking units etc.) and further study is warranted.
- Seasonal soil moisture dynamics have a major influence on soil physical quality and P and SS losses with extensive soil damage and contaminant losses associated with winter grazing. Yet, little research has reported on physical damage and surface runoff losses occurring in summer, a time of year when biological growth is most active. The impact various soil moisture regimes have on the different fractions of P and SS losses and soil quality requires investigation.
- The damage posed by hooves and subsequent P and SS losses is largely dependent upon soil type. Further research is warranted to establish vulnerable soils from resilient soils under the same field conditions.
- Land management practices such as mechanical soil aeration has been reported to improve soil physical quality and have various impacts on contaminant loss. Yet, studies have failed to investigate the combined effect of mechanical aeration on soil physical quality and consequent contaminant losses.
- Numerous studies have reported the detrimental impact dung-derived P has on surface runoff and the damage caused to soil structure after grazing. However, research has failed to isolate and compare the relative impact of either dung or treading.

It is the objective of this thesis to address these research gaps. Four research experiments were carried out to to assess and quantify the impacts livestock grazing have on soil physical quality and losses of P and SS in surface runoff from various treatments and land management practices (Figure 1.3).

Chapter 3- Combined effects of simulated cattle treading and soil moisture on phosphorus and suspended sediment losses in surface runoff from pasture

3.1 Introduction

Cattle grazing is a major contributor of suspended sediment (SS) and phosphorus (P) losses to surface waters e.g. (Derlet *et al.* 2010). Factors such as soil moisture, grazing duration, vegetative cover, and soil physical quality are also important factors influencing how treading affects P and SS losses (Climo and Richardson 1984; McDowell *et al.* 2008).

It is accepted that compaction occurs on medium to wet soils and pugging can result from grazing saturated soils (Bilotta *et al.* 2007a; Drewry 2006; Scholefield *et al.* 1985). However, it is unclear whether or not P and SS losses increase with increasing soil moisture on trodden soils, and if so, which P fractions are affected: dissolved- or particulate-P. Furthermore, soils with poor structural stability and drainage tend to be more affected by treading than well structured soils (Russell *et al.* 2001).

The objective of this study was to determine the impact of simulated cattle treading and moisture status on the loads and concentrations of dissolved and particulate P (PP) and SS in surface runoff generated from a rainfall simulator from four different pasture soils. This study tests the hypothesis than an increase in soil moisture increases losses of P and SS in surface runoff.

3.2 Materials and Methods

3.2.1 Experimental design

Four soil types were investigated, representing those commonly used or currently being developed for pastoral grazing by cattle in southern and eastern Otago and Southland, New Zealand. The soils were namely: a Recent Gley (Momona silt; Aquent) and Pallic soil (Warepa; Fragiochrept) of moderate to high structural vulnerability (SV); and a Brown (Cargill silt loam; Dystrochrept) and Melanic (Oamaru silt loam; Rendoll) soil of moderate to low SV according to Hewitt and Shepherd (1997) (soil name and USDA classification given in parentheses) (Table 3.1). The Brown and Melanic soils are silt loams and the Pallic and Recent Gley are silts. All soils had been in pasture for > 10 yrs and sampled at similar available water holding capacity (10-25%) in autumn, 2008, from paddocks chosen to have the same slope (~ 5%) and southerly aspect.

Twelve soil samples (0-7.5 cm depth) were taken at each site to determine background chemical properties (see below) and additional samples of the 0-5 cm depth taken for soil physical analyses (see below). Soils to be placed under a rainfall simulator to generate surface runoff were excavated using a purpose-built cutting blade modified from McDowell *et al.* (2007). Briefly, this involved hammering the 0.8 m long by 0.2 m wide blade to 0.15 m depth and carefully excavating undisturbed topsoil. Soils were then trimmed to 12.5 cm depth. Pasture (all > 90% ground cover) was trimmed to 5 cm height, and turves (the undisturbed blocks of excavated topsoil) placed into plastic lined wooden boxes where one of three treatments was imposed:

- 90% of available water holding capacity (AWHC);
- 50% of AWHC; or
- 10% of AWHC,

Available water holding capacity was calculated using the soil bulk density and volumetric soil moistures calculated from saturated and air-dry soil cores collected at the same time as soils for boxes. These moisture treatments are hereafter referred to as 10%, 50% and 90%. There were four replicates of each treatment for each soil type, except for the 90% treatment, which had 16 replicates.

After treatments had been established and watered appropriately to maintain desired moisture content for at least 7 days, each boxed soil was treaded upon by an artificial cow hoof five times (20 imprints/ m^2) to simulate treading during a 24 hr grazing event. It is recognised that the degree of overlap may not be equivalent to that seen in the field (McDowell *et al.* 2003b). The hoof was moved five times across the length of the 0.8 m soil for each imprint to prevent fixing it in one position. The artificial cow hoof (Figure 3.1) was modelled on a 2 yr-old Friesian cow and delivered 250 kPa of pressure over a 90 cm^2 area (Di *et al.* 2001). Soils were then left outdoors under shelter for 24 hr. The hoof itself is split into two halves which are joined together by a hinge in the middle to create independent movement of each half and vertical pivoting simulating the movement of a walking cow.

Figure 3.1. The mechanical cow hoof simulating treading on an excavated block of soil.

3.2.2 Surface runoff collection and analyses

Soils were placed under a rainfall simulator (tap water, P less than detection limit of 0.001 mg P/L) and rainfall applied to boxed-turves at a rate 30-35 mm/hr to create surface runoff (Figure 3.2). This rainfall intensity rate has a return frequency of approximately two-three times a year for a 10 min event (NIWA 2010b). Each box was inclined at a 5% slope as in the field. The rainfall simulator used one TeeJet 1/4HH-SS30WSQ nozzle (Spraying Systems Co., Wheaton, IL) approximately 250 cm above the soil surface to produce rainfall with size, velocity, and impact angles approximating natural rainfall (Shelton *et al.* 1985). The nozzle, plumbing, in-line filter, and pressure gauge were fitted onto a 305 cm high by 305 cm wide by 305 cm deep aluminium frame with tarpaulins on each side to provide a wind screen. After the initiation of surface runoff it was collected for 30 min. Runoff was collected and measured in a measuring cylinder and a 200 ml sub-sample returned to the laboratory for further analysis.

Figure 3.2. Rainfall simulator set up to create surface runoff from treaded upon soil blocks.

Within 24 hours, samples of surface runoff were filtered (0.45 µm) after collection and analysed for dissolved reactive P (DRP). Filtered and unfiltered samples were refrigerated at 5ºC for digestion within 7 days using persulphate (Eisenreich *et al.* 1975), which, after colorimetric analysis, determined total dissolved P (TDP) and total P (TP), respectively. Fractions determined as dissolved unreactive P (DUP) and particulate P (PP) were determined as the difference between TDP and DRP, and TP and TDP, respectively. Suspended sediment (SS) was analysed by weighing the oven-dried residue on a GF/A glass fibre filter paper (0.7 µm pore size; GF75 - MFS Advantec Inc., Pleasanton, Ca, USA) before and after filtration of a known volume of sample.

3.2.3 Soil analyses

Soils (0-7.5 cm depth) for chemical analyses were air-dried, crushed and passed through a 2 mm sieve before bicarbonate-extractable P [Olsen P; (Olsen *et al.* 1954)], water soluble P [WSP; (McDowell and Condron 2004)], and total P, after digestion in aqua regia [4:1 v/v concentrated $HCl:HNO_3$; (Crosland *et al.* 1995)], were determined. All P analyses were measured in duplicate and used the colorimetric method of Watanabe & Olsen (1965).

Soil physical measurements included macroporosity (volumetric percentage of pores > 30 µm), soil bulk density, and saturated hydraulic conductivity (K_{sat}) as outlined by Drewry and Paton (2005). Undisturbed soil cores were collected from the site of excavation using stainless steel rings (10 cm diameter and 5 cm deep) for macroporosity and soil bulk density analysis. Another set of stainless steel rings (10 cm diameter and 6.5 cm deep), used for saturated hydraulic conductivity (K_{sat}), were coated on the inside with petrolatum to avoid edge flow effects (Greenwood 1989). These longer rings were inserted into the soil to leave 1.5 cm above the surface, excavated and transported to the laboratory where plaster was applied to the bottoms (for measurements of K_{sat}) or tops (for macroporosity and soil bulk density) and "peeled" away to give an unsmeared surface (Greenwood 1989). Earthworms were removed with formaldehyde, before saturating the cores and equilibrating them at both -1 and -10 kPa on tension tables to determine macroporosity. Dry bulk densities and total porosity were calculated from oven dry weights.

The effect of soil moisture treatments on the potential for DRP release and the influence of microbial biomass (before and after chloroform fumigation) was further investigated by determining soil WSP for each soil type at a range of soil moistures (10, 20, 50, 75, and 100% of AWHC). Field moist samples (~ 500 g at 20 to 30% of AWHC at sampling) were sieved to 2 mm. Triplicate sub-samples (each 100 g air-dried equivalent) were then either fumigated with chloroform (Jenkinson and Powlson 1976) or not, and left at room temperature to dry to the equivalent moisture (determined using earlier estimates of AWHC, soil bulk density and daily assessment of weight). If the soil was not wet enough, sufficient sterile water was added and mixed in with the soil using a sterile spatula until the required moisture content was reached (by weight). Soils were left to equilibrate for one week, with daily correction for evaporation if needed, before extraction of WSP.

3.2.4 Statistical analyses

Mean concentrations and loads of P fractions and SS in surface runoff were tested for normality and transformed if necessary before being subjected to an ANOVA, fitting terms for soil type, treatment and the factorial interaction of soil type and treatment. The *F*-statistic or the least significant difference at $P<0.05$ (LSD_{05}) is presented to compare means between soils and treatments, and their interaction. All analyses (for all experimental chapters) were carried out using the statistical package GenStat 11 (2008) and SigmaPlot 10 (2006).

3.3 Results and Discussion

3.3.1 Surface runoff

Soil physical analyses showed that macroporosity was greatest for the Melanic soil followed by the Brown, Recent Gley and Pallic soils, while the opposite was similar for soil bulk density (Table 3.1). Previous work has shown an inverse relationship between macroporosity and surface runoff volumes, due to the decreased water holding capacity (fewer large pores) and time to ponding caused by pore blockage e.g. McDowell *et al.* (2003a). Greater compaction (lower macroporosity and higher soil bulk density) was also observed under irrigated practices when compared to dryland treatments for two out of three years on a Pallic silt loam (Houlbrooke *et al.* 2009b). In the present study, macroporosity in the Pallic soil was least (11%), and the volume of surface runoff was significantly (P<0.05) greater (2.17 L) than the other soils (Melanic, Brown and Recent Gley producing 1.67, 1.65, and 1.23 L, respectively; Table 3.2) for the 90% soil moisture treatment.

Table 3.1. Mean chemical and physical properties of the four soils (names in parentheses) investigated. The *F*-statistic is given for comparison of means between soil types when appropriate.

Parameter	Brown (Cargill silt loam)	Melanic (Oamaru silt loam)	Pallic (Warepa silt)	Recent Gley (Momona silt)	*F*-statistic
WSP (mg/L)[a]	0.051	0.056	0.105	0.178	<0.001
Olsen P (mg/kg)	19	24	21	41	<0.001
Total P (mg/kg)	747	601	209	680	<0.001
Organic C (g/kg)[b]	70	170	30	50	
P retention (%)[2]	59	26	17	37	
Bulk density (g/cm)	0.83	0.61	1.1	0.82	<0.001
Sand (g/kg)[b]	175	235	105	55	
Silt (g/kg)[b]	620	640	840	900	
Clay (g/kg)[b]	210	130	55	40	
K_{sat} (mm/hr)[c]	2.54	3.79	2.65	2.04	<0.001
Macroporosity (% v/v)	21	34	11	17	<0.001

[a] Water soluble phosphorus.

[b] Determined on bulked soil samples. Particle size distribution presented according to USDA classification.

[c] The *F*-statistic for saturated hydraulic conductivity (K_{sat}) was calculated using the $\log_{10}$ scale for normality.

Unsurprisingly, the 90% soil moisture treatment produced significantly more surface runoff than the 50% and 10% soil moisture treatments except for the Recent Gley (Table 3.2). The likely explanation for this is that most pores were already filled with water, thereby decreasing the time to surface runoff. For example, it took nearly twice as long to produce surface runoff in the Melanic soil at 10% soil moisture compared to the 90% soil moisture treatment (data not shown). This was not the case for the Recent Gley which produced a greater volume of runoff from the 10% than the 90% soil moisture treatment. Due to restricted drainage and infiltration rates much greater than rainfall intensity (Table 3.1), saturation-excess is the most likely cause of surface runoff. Data for the Recent Gley also suggests that a proportion of surface runoff may have been caused by infiltration excess conditions, possibly due to surface smearing. However, overall no difference was observed in surface runoff volumes for the interaction of soil type and moisture.

Table 3.2. Mean load (kg P/ha) of P fractions (dissolved reactive P, DRP; dissolved unreactive P, DUP; particulate P, PP; total P, TP), suspended sediment (SS, kg/ha), and volume (L) lost in surface runoff for each soil type and treatment during the rainfall simulation. The *F*-statistic (using log transformed data) is presented for treatment comparisons.

Soil Type	Moisture	DRP	DUP	PP	TP	SS	Volume
Brown	10%	0.021	0.003	0.001	0.025	0.08	1.05
	50%	0.004	0.001	0.006	0.011	0.17	1.59
	90%	0.001	0.002	0.011	0.014	0.52	1.65
Melanic	10%	0.013	0.002	0.002	0.017	0.06	1.09
	50%	0.003	0.001	0.006	0.010	0.10	1.14
	90%	0.002	0.002	0.014	0.018	0.78	1.67
Pallic	10%	0.024	0.001	0.001	0.026	0.07	1.09
	50%	0.009	0.001	0.005	0.015	0.39	1.14
	90%	0.013	0.005	0.031	0.049	2.32	2.17
Recent Gley	10%	0.090	0.002	0.005	0.097	0.08	1.43
	50%	0.016	0.003	0.004	0.023	0.16	1.12
	90%	0.038	0.003	0.013	0.054	0.92	1.23
F-statistic$_{\text{-soil}}$		<0.001	0.498	0.095	<0.001	0.006	0.046
F-statistic$_{\text{-moisture}}$		<0.001	0.508	<0.001	0.079	<0.001	0.021
F-statistic$_{\text{-soil} \times \text{moisture}}$		0.003	0.110	0.002	0.056	0.014	0.094

3.3.2 Phosphorus and suspended sediment losses

3.3.2.1 Soil type

Figure 3.3 and Table 3.2 present the mean concentrations and loads of P fractions and SS lost in surface runoff for each soil type and moisture regime. The greatest mean TP load across all three moisture regimen occurred from the Recent Gley soil (0.058 kg P/ha), followed by the Pallic, Brown, and Melanic soils (0.03, 0.017, and 0.015 kg P/ha, respectively). Although most surface runoff was collected from the Pallic soil at 90% soil moisture, the P lost appeared to be more influenced by the Olsen P concentration for the Recent Gley (41 mg/kg) compared to the other soils (19-24 mg/kg). This is consistent with the findings of Pote *et al.* (1996). However, it has also been established that the concentration of DRP in surface runoff is a function of the quotient of both P retention and Olsen P (McDowell and Condron 2004), which can be approximated by WSP. Among the soils tested here, the Recent Gley also exhibited the greatest WSP and DRP concentration (Table 3.1 and 3.2, Figure 3.3). Given that 40 to 81% of the TP load in surface runoff was as DRP, this indicates that soil P concentration (Table 3.1) outweighed the influence of greater surface runoff in the Pallic soil (Table 3.2). Although saturation-excess surface runoff has been found to be the more common pathway for P losses in Southland and Otago compared to infiltration-excess surface runoff (Smith and Monaghan 2003a), in the field, soils are deeper than 12.5 cm, and may be subjected to a wider range of hydrological processes (e.g., subsurface seepage) than those simulated here. The PP and DUP accounted for much less, on an individual basis, of the TP load in surface runoff (12-49% for PP and 5-12% for DUP; Table 3.2).

The Pallic soil yielded the greatest SS concentration and load in surface runoff (Table 3.2, Figure 3.3). McDowell (2006b) reported losses of 1499 kg SS/ha during the 2002 hydrologic year (March- February) from a Fragic Pallic soil within a grazed dairy catchment in South Otago. The large erosion losses in this study compared with the New Zealand average for dairy of 299 kg SS/ha/year (McDowell and Wilcock 2008), were ascribed to the poor aggregate stability of the Pallic soil (Rousseva 1998) as well as livestock grazing stream banks. Hewitt & Shepherd (1997) incorporated factors that influence aggregate stability within their measure of a soil's structural vulnerability (SV) to cope with stress. Among the soils tested in the present study, the Pallic soil exhibited the greatest structural vulnerability (SV of 0.69), primarily due to the low amounts of Al and Fe oxy-hydroxides and low organic carbon concentration (Table 3.1). Although classed as having 'moderate structural vulnerability' (SV of 0.5 to 0.6), the Recent Gley soil also exhibits poor drainage (Hewitt and Shepherd 1997) and experienced large SS losses in surface runoff at 90% soil moisture (Table 3.2). The greater organic C concentration and drainage (as indicated by macroporosity values) for the Melanic and Brown soils (Table 3.1) may have contributed to decreased SS losses in surface runoff for 90% soil moisture (Table 3.2, Figure 3.3).

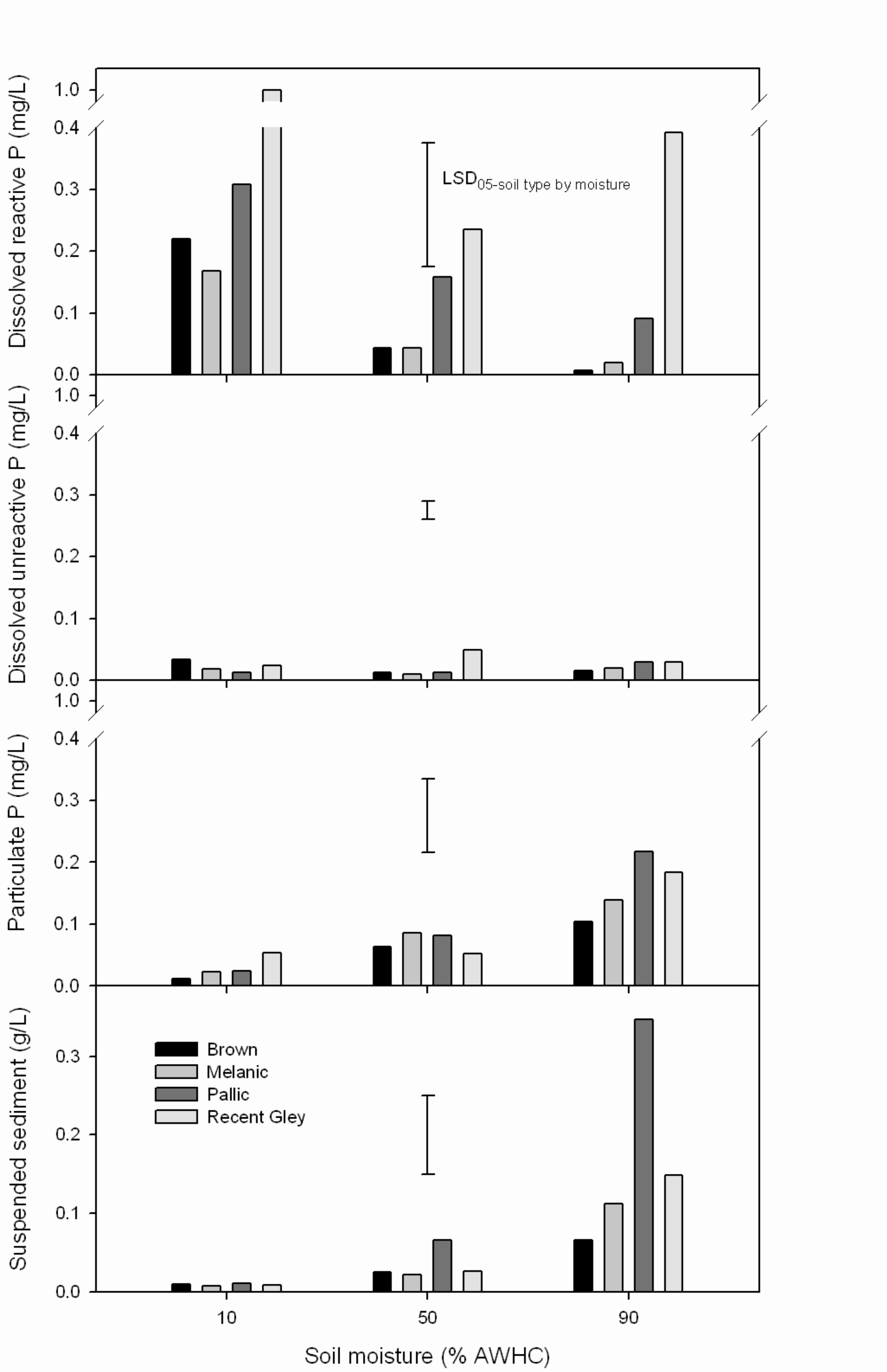

Figure 3.3. Mean concentrations of dissolved reactive P, DRP; dissolved unreactive P, DUP; particulate P, PP and suspended sediment lost in surface runoff for each soil and moisture content. The LSD_{05} is given for the interaction between soil type and moisture content.

3.3.2.2 Influence of soil moisture

Among soil moisture treatments, TP loads varied from an average of 0.010 to 0.097 kg P/ha (Table 3.2). Although differences in TP load across treatments were not significant, differences among DRP and PP fractions and SS losses were. On average, the greatest SS loads were lost from the 90% soil moisture treatment followed by the 50% and 10% treatments (1.135, 0.205, 0.073 kg/ha, respectively; Table 3.2). This is most likely due to increases in erosion power with increasing runoff volume e.g. (Quinton *et al.* 2001). Since PP losses usually arise via similar erosive processes (Ballantine *et al.* 2009), increased surface runoff also explains the greater PP load and concentration with 90% soil moisture content compared to runoff produced under 50% and 10% soil moisture treatments (Table 3.2, Figure 3.3).

In addition to increased surface runoff, the treading impact increased with soil moisture (Climo and Richardson 1984; Drewry 2006). Under wet conditions, the friction between soil particles is lessened and particles can move along points of weakness, decreasing soil strength and altering soil structure (Platto *et al.* 1978). Visual inspection of the hoof imprints indicated that while some compaction was probable on soils at 50% soil moisture content, raised edges around the imprint (Bilotta *et al.* 2007a), indicative of pugging, was probable on the soils at 90% soil moisture. A past study of a Pallic soil in South Otago showed that sediment and P loss in surface runoff was restricted by ponding and sedimentation in deep imprints (McDowell *et al.* 2003a; McDowell *et al.* 2003b). However, the sedimentation potential of the imprint will be dependent upon depth, whereby sedimentation is probably negated in shallow imprints by turbulent surface runoff, much like turbid waters of a shallow lake on a windy day. The imprints in the study of McDowell *et al.* (2003b) were up to 20 cm deep, while those in the present study were up to 12 cm deep, particularly for 90% moisture contents. Although imprint depth may have been restricted by the depth of the soil boxes (12.5 cm), the salient point is that PP and SS loss increased with soil moisture content to the point that PP losses accounted for up to >70% of TP losses in soils at 90% soil moisture. Possible reasons for the increase in TP and SS, in addition to surface runoff volumes, include the crushing and smearing of aggregates at the soil surface (Platto *et al.* 1978), an increase in soil bulk density beneath the imprint (O'Connor 1956), and an increase, via imprints, in the surface area of soil interacting with surface runoff.

In contrast to PP and SS, DRP concentrations and loads decreased with increasing soil moisture, where within the 10% and 50% and 90% soils, the load of DRP lost accounted for

88%, 53% and 40% of TP losses, respectively. Clearly, for DRP different processes were affected by soil moisture compared to SS and PP.

3.3.2.3 Water soluble P and soil moisture

Water soluble P extractable from soil (WSP-soil) can be used as a surrogate for DRP in surface runoff (McDowell and Condron 2004). Hence, WSP was used to further investigate the changes in DRP with soil type and moisture. The principal hypothesis was that desiccation and lysis of the soil microbial biomass at 10% soil moisture increased availability and losses of DRP in surface runoff (Sparling *et al.* 1987). If true, this may partly explain why concentrations of DRP in summer surface runoff events tend to be enriched for certain soil textures (Magid and Nielsen 1992; Magid *et al.* 1996). An alternative hypothesis is that DRP concentration becomes enriched and builds up simply because of reduced soil moisture in drier periods. Or in other words, DRP concentration in surface runoff under saturated excess conditions from wetter soils is lower due to dilution. Turner & Haygarth (2001) also reported that total amounts of WSP were minimal in moist soils (those drained from saturation for 48 h), but increased after drying by 185 to 1900%.

Fumigation with chloroform, followed by extraction of fumigated and unfumigated soils, has been used as a tool to give a measurement of microbial biomass P (Brookes *et al.* 1982). Although usually extracted with bicarbonate, water (WSP-microbial biomass) was used to extract the P fraction likely available to surface runoff. Precedents, where extraction with water has been used to establish WSP-microbial biomass, exist for soils (Morel *et al.* 1996; Oehl *et al.* 1998) and stream sediments (McDowell 2003). Data in Figure 3.4 indicates that WSP-soil and WSP-microbial biomass were much greater, at most soil moisture contents, in the Recent Gley soil than other soils, which mirrored DRP in surface runoff (Figure 3.4). Amongst soils, the relative proportion of WSP-microbial biomass compared to WSP-soil tended to be lower at 10% moisture compared to either 50% or 100% (Figure 3-4). This was especially so for the Pallic soil which had very little WSP-microbial biomass for any given soil moisture. Exact reasons for this difference compared with the other soils are unclear. The increase in the proportion of WSP-microbial biomass in the Melanic soil, up to 50% moisture, suggests that lysis played a limited role in supplying DRP for surface runoff compared to the other soils where WSP-soil increased while WSP-microbial biomass decreased. Although DRP enrichment at 10% soil moisture for the Melanic soil could be related to dilution/concentration, Haynes & Swift (1985) provided an alternative hypothesis. When looking at the effect of air-drying on soil chemical properties on four acid soils in New

Zealand, they showed that air-drying increased extraction of native soil P compared to moist soils. The increase in P extractability was thought to be associated with the decomposition of organic matter complexes upon drying. Although speculative, the enriched organic C concentration of the Melanic soil (170 g/kg, Table 3.1), an attribute for low SV (Hewitt and Shepherd 1997), makes this alternative hypothesis possible. Another reason could be that the microbial population was more resilient to desiccation, perhaps imparted by high organic matter concentrations (Jenkinson and Powlson 1976).

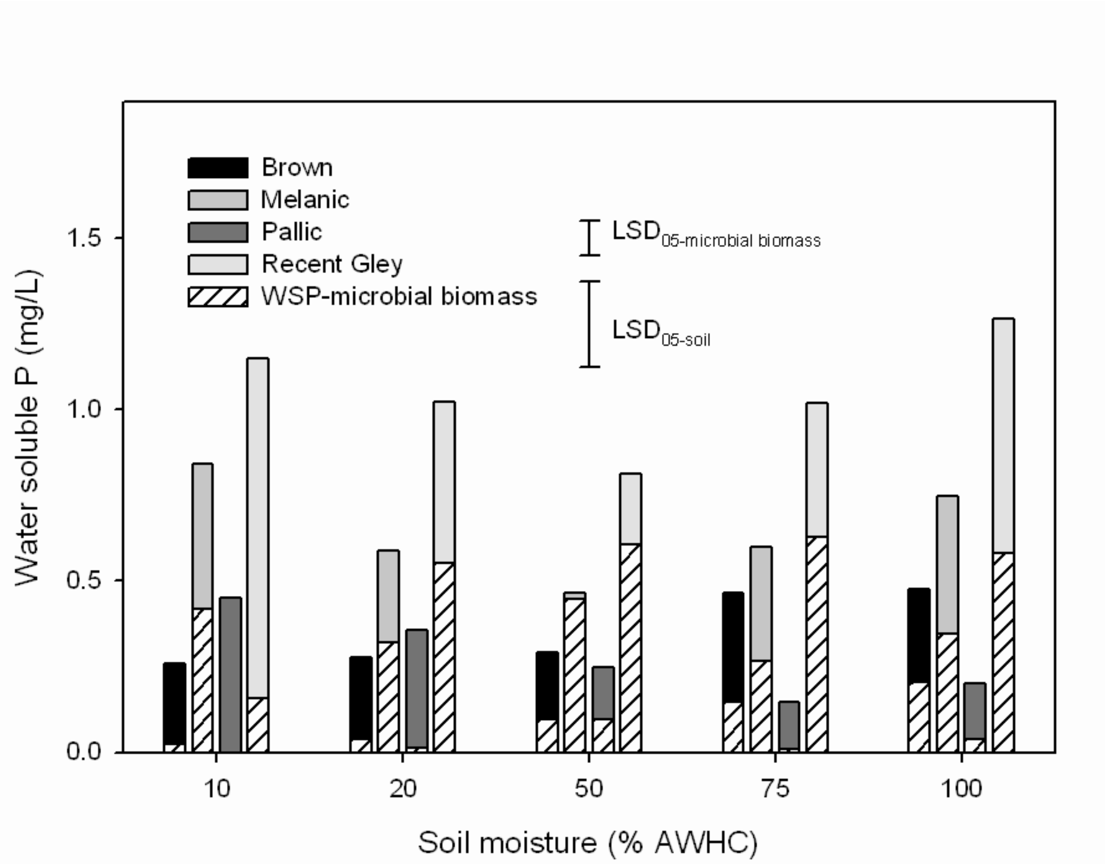

Figure 3.4. Mean concentration of water soluble P extractable from soil (WSP-soil) and from microbial biomass for each soil type and moisture content. The LSD_{05} is given for the interaction between soil type and moisture for WSP-soil and for WSP-microbial biomass.

In contrast to other soils, DRP loads in surface runoff from the Recent Gley soil at 90% soil moisture were more than double the DRP loads observed at 50%, despite a small but not significant difference in surface runoff volumes (Table 3.2). Magid *et al.* (1996) suggested that ferric minerals may become reduced via anaerobic conditions during the winter. Greenwood & McNamara (1992) noted that treading can cause a soil to become waterlogged and oxygen deficient. Data in Figure 3.4 illustrates that for the Recent Gley soil, WSP-soil was greatest at either wet or dry extremes, and for 100% soil moisture reduction via anaerobic conditions is very likely. In addition to the reduction of ferric-P, anaerobic conditions can contribute P via the death of plant roots (Gradwell 1968) and micro and macro fauna. At 10% soil moisture, it is likely that WSP-soil peaked due to the lysis of soil microbial biomass because WSP-microbial biomass was least at 10% soil moisture. Although these experimental conditions may have led to a reinoculation of microbes during the week where soil was left to equilibrate open to the air, it is highly unlikely that this would match the size and diversity of the original population. The fact that differences between WSP-soil and WSP-microbial biomass were large also indicates that reinoculation (which would minimise this difference) was not a significant factor.

These data suggest that the majority of DRP occurs from dry soils, while greater PP and SS losses occur from wetter soils, implying the periods of greater risk after a 24 hr grazing event (with associated treading) by cattle, are summer and autumn for DRP, and winter and spring for PP and SS. However, runoff in Otago and Southland is mostly generated by saturation-excess conditions (Srinivasan *et al.* 2007) implying that the risk of surface runoff associated with summer and autumn is less than in winter and spring. As shown with the Recent Gley soil, saturated soils or soils at field capacity for prolonged periods also have the potential to experience reducing conditions (Gradwell 1968), which can influence DRP losses. The current experiment did not allow for the fragipan within the Pallic subsoil; this severely limits drainage at 25 cm depth. This limitation, coupled with the low P retention, suggests that the Pallic soil has excessive PP and SS loss via saturation-excess surface runoff. In contrast, the Brown and Melanic soils had good structural stability and were unlikely to experience such reducing conditions. Best management practices (BMPs), such as restricted grazing, a decrease in stocking frequency or density should be considered, especially for poorly structured soils, during wet spring-winter conditions in Otago and Southland. Such BMPs could decrease the damage caused by the hooves to topsoil, the removal of pasture cover, and protect against erosion and losses of sediment P (Nash and Halliwell 1999).

3.4 Conclusions

The equivalent of 24 h treading by a mechanical hoof on soils of contrasting soil moistures caused differences in the concentrations and loads of P fractions and SS lost in surface runoff from four soil types. A Pallic and Recent Gley soil were shown to be more susceptible to SS and P losses than a Brown or Melanic soil. With increasing soil moisture, the combination of physical disturbance, and crushing and burial of pasture, depleting its protective cover against soil erosion, caused an increase in SS and PP losses. In contrast, increases in DRP losses from drier soils were attributed to the release of soil microbial biomass P, except for the Melanic soil where the effect was either absent or much decreased. The concentration of WSP in the Recent Gley soil was greater for 100% moisture than that at 50% moisture and was attributed to increased P release under reducing conditions.

These results indicate that the risk to surface water quality from runoff, when soil moisture is low, is high due to the coincidence of enriched bioavailable P (DRP) in the runoff, warm temperature, and high light levels that promote algal growth. However, the risk posed by losses of PP and SS with treading is also great, because surface runoff is more likely in wetter periods. This winter-derived loss is particularly important for the Pallic and Recent Gley soils studied, and suggests that care should be taken when grazing these soils under wet spring-winter conditions in Otago and Southland. Although collected under controlled conditions, these data will be useful in explaining the wide range of potential SS and P losses that may occur in a field due to hydrologic and soil variation.

Chapter 4- Do aggregation, treading and dung deposition affect phosphorus and suspended sediment losses in surface runoff?

4.1 Introduction

The loss of suspended sediment (SS) from land is an important cause of surface water quality deterioration via siltation, the loss of fish spawning grounds and as a carrier for contaminants, such as phosphorus (P) (Bilotta and Brazier 2008; Droppo and Ongley 1993; Quinton *et al.* 2001).

Bilotta *et al.* (2010) reported that although pastures appear to have good vegetative cover protecting them from erosion, fine material can still be eroded that may be enriched with P. The presence or absence of grazing animals will have a major influence on the amount of eroded particles and associated P lost from pastures (Heathwaite 1997). Increasing the frequency or intensity of grazing also causes more particulate P to be lost, although it is unknown whether this is due to increased soil disturbance or dung deposits (McDowell *et al.* 2003a; McDowell *et al.* 2003b; Nguyen *et al.* 1998). In addition, it has been found that recently manured, but ungrazed, fields had enhanced losses of particulate-P via aggregates sometimes referred to as "flocs" (McDowell and Sharpley 2002; McDowell and Sharpley 2003a). These flocs have been found in streams and rivers and contain a mix of particle sizes that are held together by organic materials and microbes that cause the resulting floc or aggregate to become more buoyant and transportable than the individual components (Droppo *et al.* 1998). The main objective of this study was to determine the concentration and load of SS and particulate-P and the role of aggregates in enhancing their transfer in surface runoff from soils under simulated cattle grazed pastures with and without dung deposition. It is hypothesised that the contribution of dung will cause greatest P and SS losses in simulated rainfall generated surface runoff. Further, four soil types with contrasting soil drainage and structural properties will be considered as it is known that the effects of treading on runoff processes and P losses is soil type dependent (Chapter 3).

4.2 Materials and Methods

4.2.1 Experimental design

Four soil types were investigated (Table 4.1), representing those commonly used or currently being developed for pastoral grazing by cattle in southern and eastern Otago and Southland, New Zealand, namely Brown, Melanic, Pallic and Recent Gley (see section 3.2.1). Sixteen soil samples (0-7.5 cm depth) of each soil type were taken to determine background soil chemical properties (see below and section 3.2.3), and twenty-four samples of each soil type were also taken from each site for soil physical analyses (see below and section 3.2.3).

All soils were collected in early summer at a soil moisture content of about 25% of available water holding capacity (AWHC). Soils were then excavated (80 cm long by 20 cm wide blade to 15 cm depth) using a purpose-built cutting blade (see section 3.2.1). Soils were trimmed to 7.5 cm depth, pasture trimmed to 5 cm height, and soils placed into 10 cm high metal boxes with the downslope end 7.5 cm deep. Boxed soils were placed under a sheet of plastic liner, dried or moistened to maintain 50% AWHC (see section 3.2.1) for one week. After a week, one of three treatments was employed, each with four replicates for each soil type:

1. an untreaded control (control);
2. a treaded treatment that did not receive cattle dung (treaded); and
3. a treaded treatment receiving an application of cattle dung (tread + dung).

For treading, each box was treaded upon by an artificial cow hoof five times (20 imprints/ m^2) to simulate treading during a 24 hr grazing event (see section 3.2.1). The dung treatment involved placing 0.5 kg of cattle dung (moisture content *c*. 78%) within a 15 cm diameter metal ring at the upslope end of the metal box. Dung (20 kg) was collected from fresh dung patches the day prior to the experiment on a feed-lot of the AgResearch-Invermay beef farm and thoroughly mixed before application. The application rate was equivalent to the weight of dung deposited over a 24 hr period, but due to an inability to keep its shape during application, the surface area was about twice what occurs in the field (Haynes and Williams 1993). Once treatments were imposed, metal rings were removed, and soils left outside under shelter until rainfall simulation the next day.

Soils were placed under a rainfall simulator, which applied water containing <0.001 mg P/L at a rate 30-35 mm/hr to create surface runoff (see section 3.2.2). After initiation, runoff was collected for 30 minutes.

4.2.2 Soil and runoff analyses

Soils for chemical analyses from each site were analysed for Olsen P, water soluble P (WSP) and total P via aqua regia digestion (see section 3.2.3). Total P via aqua regia and WSP were also determined on air-dried subsamples of dung that had been ground to < 1 mm (1978 mg/kg and 2.08 mg P/L, respectively).

Soil physical measurements included macroporosity, soil bulk density and saturated hydraulic conductivity (K_{sat}) (see section 3.2.3).

Within 24 hours, surface runoff samples were analysed for dissolved reactive P (DRP). Within seven days of refrigeration surface runoff samples were analysed for dissolved unreactive P (DUP), total dissolved P (TDP), particulate P (PP), total P (TP) and suspended sediment (SS) (see section 3.2.2).

4.3.3 Particle size analysis and aggregation

To determine the influence of aggregation on P losses, 500 mL sub-samples of surface runoff were mechanically dispersed using a high speed homogenizer (25,000 rpm, Kinematica®, Polytron-Aggregate®, Luzern, Switzerland) for 5 minutes. Dispersed and undispersed samples were put into 500 mL measuring cylinders, mixed and left to sit. After 19.1 minutes, which corresponds to the time taken for particles > 10 µm to travel 10 cm according to Stoke's law, the top 10 cm of sample was removed by vacuum. Fractions defined as < and > 10 µm from dispersed and undispersed samples were then analyzed for DRP, DUP, TDP, PP, TP, SS and particle size by a laser particle size analyser (Malvern Mastersizer, Malvern Instruments Ltd, Malvern, UK). The Malvern particle size analyser was used to help define particle size, but it neglects the variability in particle shape, an important variable in settling velocity (Droppo *et al.* 1998). Hence, settling time should be used as the true measure of aggregation.

The degree of aggregation was then expressed as the percentage change in P fraction or SS load between dispersed and undispersed sub samples and defined as light (those floating in the top 10 cm after 19.1 minutes) and heavy (those that sank after 19.1min) particles according to Stoke's law:

1. Degree of aggregation (light) = (dispersed < 10µm - undispersed < 10 µm / dispersed < 10 µm) × 100, and
2. Degree of aggregation (heavy) = (dispersed > 10 µm - undispersed > 10 µm / dispersed > 10 µm) × 100.

4.3.4 Statistical analyses

Mean loads of P fractions and SS in surface runoff, and soil chemical and physical properties were tested for normality and log transformed if necessary before an ANOVA was performed, fitting terms for soil type, treatment, aggregation (dispersed vs. undispersed samples) and their interactions. An additional ANOVA was performed using the percentage change in aggregation of light and heavy particles and fitting terms for the influence of soil type and treatment and their interaction. Where used, the *F*- statistic and standard error of difference (SED) is presented in tabular form while specific comparisons in graphs are made with the least significant difference at P<0.05 (LSD_{05}).

4.4 Results and Discussion

4.4.1 Soil type and treatment effects on phosphorus and suspended sediment loads in runoff

On average, the load of DRP, DUP and TP lost from the control treatment was significantly greater for the Recent Gley soil followed by Pallic, Brown, and Melanic soils (Table 4.2). These losses generally reflected soil chemical properties, principally Olsen P concentrations, which was greatest in the Recent Gley (41 mg/kg). However, despite having similar Olsen P concentrations, P loads from the Pallic soil were greater than from either the Brown or Melanic soils. This probably reflected the lower P retention of the Pallic soil (17%), which when used as the denominator in a quotient with Olsen P can be used as a predictor of DRP in surface runoff (McDowell and Condron 2004). As confirmation of loads, the WSP concentration in the Pallic soil was almost twice that in either the Brown or Melanic soil (Table 4.1). The mean loads of DUP and PP lost from the control soils were lower than DRP losses. For PP this may have reflected the good (generally > 80%) pasture cover decreasing the erosive impact of raindrop and filtering PP from surface runoff (Bilotta *et al.* 2007a). Volumes between soil types and treatments were not significant but they were for the interaction between soil type and treatment [P <0.01; standard error of difference (SED) 0.12]. Significant differences were mostly reflected between volumes produced for the Brown and Pallic tread treatment (2.4 and 1.4 L, respectively) but volume differences of only 0.3-0.4 L occurred between other soil types and treatments thereby losses of loads have been reported as opposed to concentrations (Table 4.2).

Table 4.1. Mean chemical and physical properties of the four soils (names in parentheses) investigated. The *F*-statistic and standard error of difference (SED) is given for comparison of means between soil types.

Parameter	Brown (Cargill silt loam)	Melanic (Oamaru silt loam)	Pallic (Warepa silt)	Recent Gley (Momona silt)	*F*-statistic	SED
WSP (mg/L)[1]	0.051	0.056	0.105	0.178	<0.001	0.016
Olsen P (mg/kg)	19	24	21	41	<0.001	5.23
Total P (mg/kg)	678	601	207	661	<0.001	56.1
Organic C (g/kg)[2]	70	170	30	50		
P retention (%)[2]	59	26	17	37		
Bulk density (g/cm^3)	0.77	0.59	1.05	0.77	<0.001	0.024
Sand (>63 µm; g/kg)[2]	175	235	105	55		
Silt (20-63 µm; g/kg)[2]	620	640	840	900		
Clay (< 2 µm; g/kg)[2]	210	130	55	40		
K_{sat} (mm/hr)[3]	2.54	3.79	2.65	2.04	<0.001	0.244
Macroporosity (% v/v)	24	40	13	17	<0.001	1.18

[1] Water soluble phosphorus.

[2] Determined on bulked soil samples.

[3] Saturated hydraulic conductivity (K_{sat}) was calculated using $\log_{10}$ transformed data.

Although PP lost in surface runoff was unaffected by soil type, SS losses were greater from the Pallic and Recent Gley soils compared to the Melanic and Brown soils (Table 4.2). The Pallic soil has a high potential for dispersion and slaking and is susceptible to damage under livestock treading (Hewitt and Shepherd 1997; Houlbrooke *et al.* 2009b). According to Hewitt and Shepherd (1997) the Recent Gley has a lower structural vulnerability score (0.54) than the Pallic soil (0.69), but texturally, was also a silt (90% < 20 µm), which is known to be susceptible to pugging (Hewitt 1998), which in turn encourages slaking and erosion of SS.

Table 4.2. Mean load (mg/m^2) of P fractions (dissolved reactive P, DRP; dissolved unreactive P, DURP; particulate P, PP; total P, TP) and suspended sediment (SS, g/m^2) lost in surface runoff for each soil type and treatment during the rainfall simulation. Numbers in parenthesis are the loads for dispersed samples. The *F*-statistic for the effect of soil type, treatment, aggregation and their interactions is also given.

Soil Type	Treatment	DRP	DUP	PP	TP[1]	SS
Brown	Control	0.56 (0.50)	0.13 (0.12)	0.35 (0.41)	1.04	0.04
	Tread + dung	0.69 (0.50)	0.91 (0.86)	0.90 (1.14)	2.50	0.23
	Tread	0.63 (0.60)	0.17 (0.18)	0.41 (0.43)	1.21	0.06
Melanic	Control	0.39 (0.38)	0.10 (0.12)	0.34 (0.31)	0.83	0.04
	Tread + dung	1.02 (0.98)	0.27 (0.27)	0.25 (0.29)	1.54	0.25
	Tread	0.45 (0.25)	0.16 (0.18)	0.54 (0.76)	1.15	0.06
Pallic	Control	1.60 (1.66)	0.07 (0.07)	0.37 (0.31)	2.04	0.11
	Tread + dung	4.29 (3.61)	0.57 (0.66)	1.15 (1.74)	6.01	0.37
	Tread	0.97 (0.68)	0.05 (0.07)	0.18 (0.44)	1.20	0.17
Recent Gley	Control	2.08 (2.18)	0.15 (0.10)	0.24 (0.20)	2.47	0.05
	Tread + dung	2.52 (1.14)	0.50 (0.52)	1.45 (2.81)	4.47	0.23
	Tread	2.34 (2.19)	0.11 (0.19)	0.39 (0.46)	2.84	0.07
F-statistic$_{\text{-soil}}$		<0.001	ns[1]	ns	<0.001	0.004
F-statistic$_{\text{-treatment}}$		0.007	<0.001	<0.001	<0.001	<0.001
F-statistic$_{\text{-aggregation}}$		ns	ns	0.001	-[2]	-
F-statistic$_{\text{-soil × treatment}}$		0.028	0.048	0.049	0.047	ns
F-statistic$_{\text{-treatment × aggregation}}$		ns	ns	ns	-	-
F-statistic$_{\text{-soil × aggregation}}$		ns	ns	ns	-	-
F-statistic$_{\text{-soil × treatment × aggregation}}$		ns	ns	0.022	-	-

[1] Not significant.

[2] Mean and statistical analysis of TP and SS in dispersed samples not applicable.

On average, losses of P and SS were greatest under the tread + dung treatment, followed by the tread and control treatments. Across all treatments, the tread + dung treatment accounted for about 50% of TP losses for the Brown soil followed by 45%, 65% and 45% of TP losses for the Melanic, Pallic, and Recent Gley soils, respectively. The ratios of WSP dung: WSP Brown, WSP Melanic, WSP Pallic and WSP Recent Gley soils were 40:1, 37:1, 20:1, and 12:1, respectively. The greatest fraction of P lost among treatments, but especially in the tread + dung treatment, was DRP (about half of TP) followed by PP and DUP (Table 4.2). These results are not unexpected since dung-derived P has been found to contribute 25-33% of P lost from farms, with DRP being the largest fraction (McDowell *et al.* 2006a). The tread + dung treatment also contributed to major SS losses and accounted for about 70% of total SS losses for the Brown soil followed by 70%, 60%, and 70% of total SS losses for the Melanic, Pallic, and Recent Gley soils, respectively.

The proportion of PP as TP and SS in surface runoff from the tread and tread + dung treatments was greater than that lost from the control treatment. Nash and Haliwell (1999) ascribed the enriched losses of PP and SS in surface runoff to the action of the cow hoof which causes the dislodgement of aggregates, the burial and crushing of grass (particularly when wet), and dung deposition. Smith and Monaghan (2003a) also concluded that P loss was the result of the surface runoff of dung-derived P and detrimental damage to soil structure observed after grazing, particularly in winter. However, neither study was able to isolate the relative impact of either dung or treading. These data would suggest that much of the PP lost from the tread + dung treatment was from dung rather than soil since the differences in PP load compared to the tread treatment were greater than the difference between the tread and control treatments (Table 4.2). Increased PP loss may also be ascribed to sediment sorting and the loss of more P-enriched fines, especially in the tread + dung treatment (Ghadiri and Rose 1991). However, although collected under controlled conditions, these data is only indicative of processes and should not be scaled up since the surface area of the dung applied was about twice that which occurs in the field.

4.4.2 Aggregation of phosphorus and suspended sediment in runoff

The settling and fractionation of undispersed runoff into light and heavy samples, separated at 10 μm according to Stoke's law, indicated similar loads of P fractions in both samples (Figure 4.1). However, considering that light particles (< 10μm according to Stoke's law) accounted for <30% of total runoff volume (i.e. the size of sample vacuumed off after 19.1 min), the P contained in the light sample is clearly more concentrated than P within the heavy sample. This is consistent with the enhanced transport of P in fine material, especially when recently manured (McDowell and Sharpley 2003a), or in this case under simulated grazing with dung applied. The greater P loss and enrichment of the light sample within the Pallic and Recent Gley soils is also consistent with soil dispersion and suspension of sediment in runoff waters (Withers *et al.* 2007a). Suspended sediment and PP correlated well in both light and heavy samples (r^2 = 0.84 and 0.86, respectively). The smaller load of P from the tread treatment of each soil compared to the tread + dung treatment was consistent with the enrichment of P from dung, more than treading, irrespective of settling time.

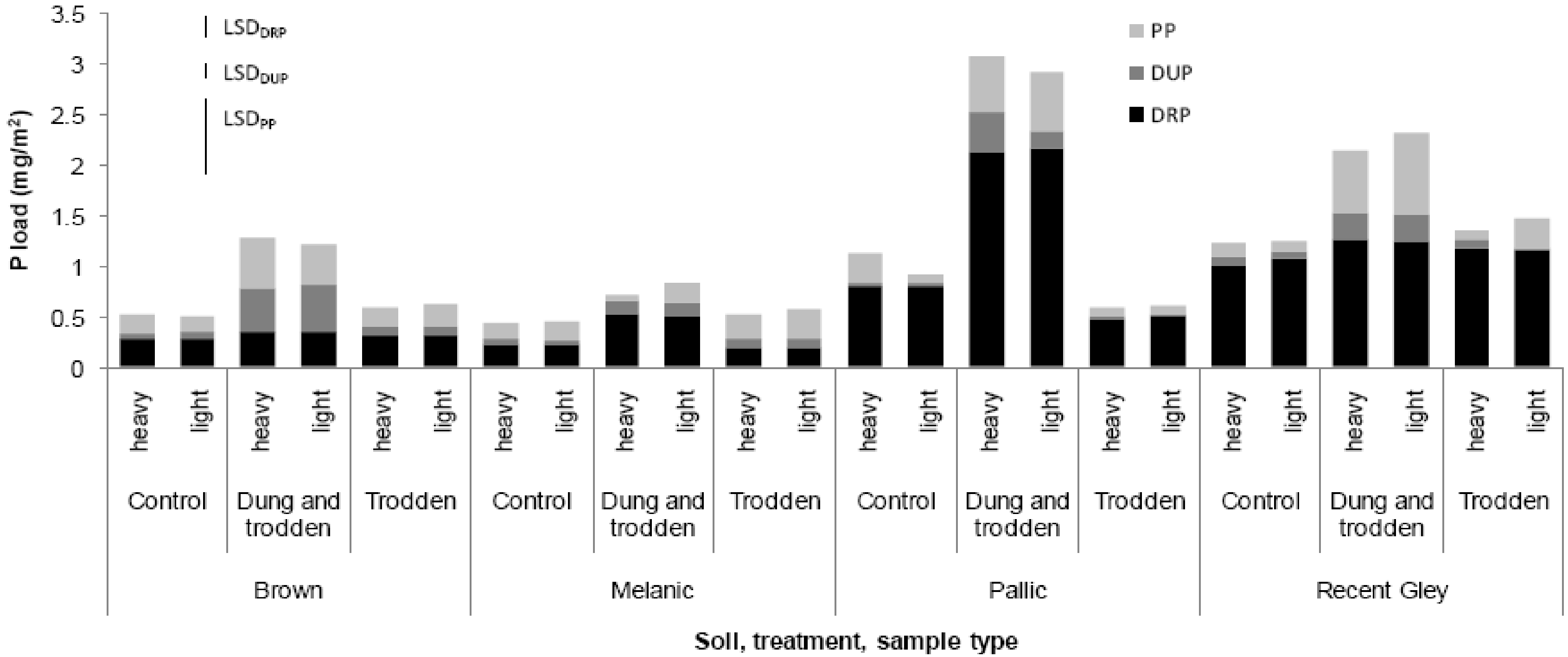

Figure 4.1. Mean load of P fractions in each soil, treatment and particle type in undispersed samples. The least significant difference (LSD_{05}) is presented for significance at the $P<0.05$ level for the interaction of soil, treatment and sample type (light vs. heavy) for each P fraction.

To determine if particles in the light and heavy samples were affected by aggregation, which may enhance P transport in surface runoff, both heavy and light samples were dispersed. Analysis via a Malvern laser particle sizer showed that the heavy samples tended to contain a greater proportion of large particles, but that the occurrence of particles <10 μm in the heavy (i.e. >10 μm) sample and vice-versa (Figure 4.2), could also indicate the influence of flocs or aggregates, which by variations in shape and porosity, alter their settling velocity (Droppo *et al.* 1998). Examination of loads in dispersed and undispersed samples of runoff indicated that there was a significant affect of aggregation on the load of PP lost (Table 4.2). The PP load was greater in dispersed than undispersed samples of the tread treatment, which was accentuated if dung was applied or if runoff originated from the Pallic or Recent Gley soils. Once dispersed, the constituent parts of the aggregate are exposed, creating a greater number of P-sorption sites. P sorption from the solution would lead to an increase in PP (Bilotta *et al.* 2007b; Haygarth *et al.* 2006).

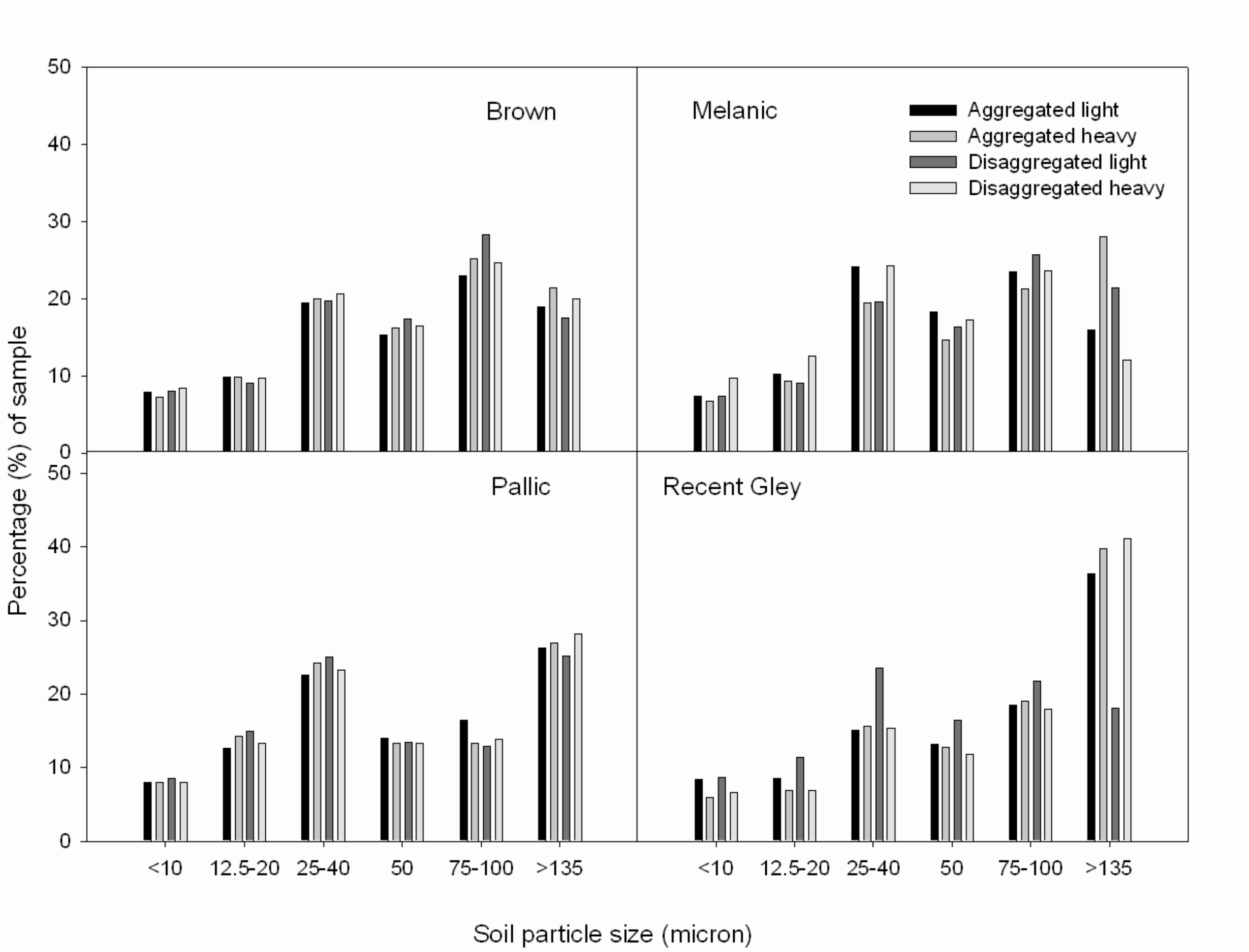

Figure 4.2. The percentage of suspended sediment occupied by each particle size class for dispersed and undispersed light and heavy samples for Brown, Melanic, Pallic, and Recent Gley soils.

Owing to the different initial soil P concentrations (Table 4.1), comparison of the effect of aggregation on PP load from different soils and treatments was assessed as the percentage change before and after dispersion (Figure 4.3). Figure 4.3a illustrates that while the Pallic and Brown soils appeared to be more affected by aggregation, especially within the light sample, only the Pallic soil was significantly different, and only for the light sample compared to the Melanic soil. Figure 4.3b shows that PP in both light and heavy samples for the tread and tread + dung treatments increased. In addition to the loss of aggregates from soil (e.g., tread treatment), it is important to note that aggregation in the tread + dung treatment may have been enhanced by the secretion of organic material from bacteria in dung which can bind particles together and thereby increase the size and strength of aggregates (Droppo and Ongley 1993; Kranck 1984). Although not presented, aggregates >135 µm were detected in many more samples of the dung + tread treatment (48) than the tread treatment (12) and were absent in the control treatment. Figure 4.3c further examines the change for small particles and shows that a major contributor to the aggregation effect was caused by losses from the Pallic soil. Although, on an absolute scale, the change in PP may not be great compared to that lost from the tread + dung treatment, these data emphasize the potential for light aggregates within the Pallic soil to contain and transport PP within surface runoff.

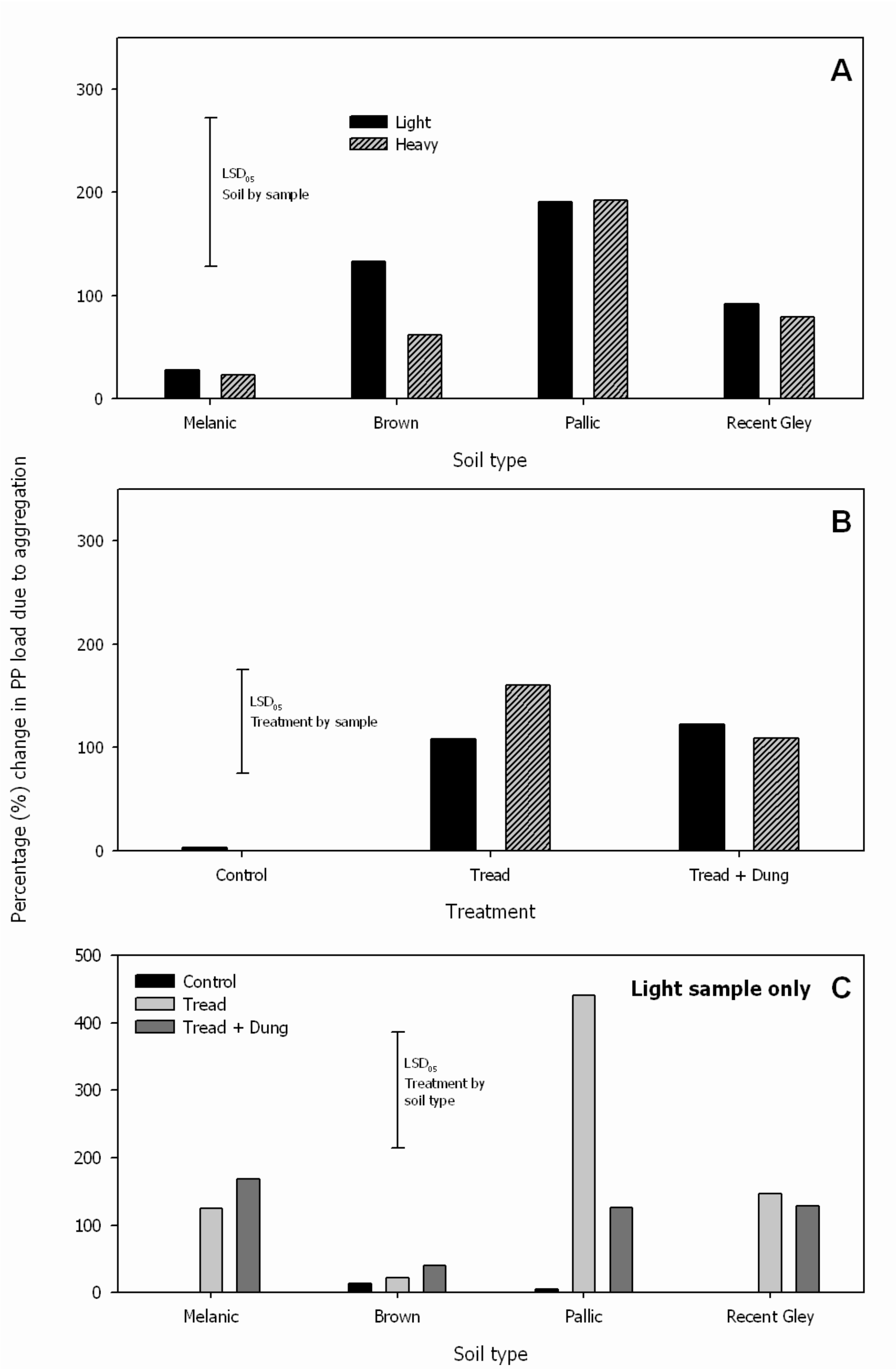

Figure 4.3. The percentage change in PP associated with "light" and "heavy" particles presented for (a) each soil type, (b) each treatment and for (C) light particles for each soil type and treatment. The least significant difference (LSD_{05}) is presented for significance at the P<0.05 level.

Considering that both treading and dung deposition occurs during grazing in winter when surface runoff is likely, or in wet spring conditions if stock grazing has been avoided during winter, these data suggest that the potential for P losses may be large. In addition, grazing under such adverse conditions leads to possible harm to soil quality and pasture production (Houlbrooke *et al.* 2009b). Much of this winter grazing occurs on Pallic soils. To prevent these adverse effects, strategies such as restricted grazing e.g. (Ward and Greenwood 2002) and the use of stand-off pads appear particularly pertinent for Pallic soils. Riparian vegetation may also act as buffer and provide some efficiency in trapping SS and PP (Monaghan *et al.* 2007a), the P fraction that was largely influenced by aggregation. However, additional work, especially field scale *in situ* plot studies, should be conducted to see which soils, in addition to those studied here, would produce more P losses in surface runoff as a result of aggregation and the interaction of grazing (treading and dung deposition). Given that greatest risk was identified from those soils that are classified as being poorly structured, it would make sense to assess those soils that met this criterion, namely: soils with low short-range-order Al- and Fe-oxy-hydroxides; have high clay concentration, and; have little organic carbon and poor drainage (Hewitt and Shepherd 1997).

4.5 Conclusions

Results reveal that cattle grazing and associated treading and dung deposition increased P and SS losses in surface runoff. These losses were greatest from soils with poor structural stability namely the Pallic and Recent Gley soils. Losses of DRP in the tread + dung treatment was a major contributor to these losses and to a lesser extent in the tread and control treatments. The addition of dung increased P losses much more than the action of treading. Fractionation of runoff samples into heavy and light fractions emphasized the loss of P in the lighter fraction, especially from the Pallic and Recent Gley soils. Additional analysis of samples before and after dispersion indicated that both the light and heavy samples contained aggregates. For P, the action of treading and the addition of dung enhanced the loss of these aggregates, especially in the lighter fraction from the poorly structured Pallic soil. The presence of aggregates in the light fraction suggests that they would travel much farther in surface runoff, and potentially be delivered to watercourses, than loss of PP in non-aggregated forms.

Chapter 5- Effects of cattle, sheep and deer grazing on soil physical quality and losses of phosphorus and suspended sediment losses in surface runoff

5.1 Introduction

Livestock grazing can have detrimental impacts on soil physical quality which can promote the loss of non-point source pollutants such as phosphorus (P) and suspended sediment (SS) in surface runoff (McDowell *et al.* 2003a; 2003b). Although cattle have been reported to have a greater negative influence on agronomic and soil physical quality due to treading than sheep (Houlbrooke *et al.* 2006), little work has contrasted potential environmental impacts by stock type (McDowell *et al.* 2003a; 2003b; Nguyen *et al.* 1998).

In a survey of catchment studies in New Zealand, McDowell and Wilcock (2008) reported that loads of P were greatest and of similar importance from catchments grazed by deer or mixed stock (sheep and beef) compared to those grazed by dairy cattle, whereas loads of sediment were significantly greater for deer followed by mixed stock and dairy cattle. However, the influence of treading and grazing among stock types on surface runoff losses could not be extracted from the data.

Recently, notable changes in land use in New Zealand include an increase in the conversion of sheep farms to dairy farms and a large increase in the number of farmed deer (MAF 2008). If the current rise in land-use intensity continues, new areas will be opened up for grazing that may not be suitable for such practices. This includes structurally poor soils that have slow recovery rates, which in turn may create problems of soil erosion and loss of contaminants in runoff (Drewry and Paton 2000; Steinfeld *et al.* 2006). The aim of this study was to examine the impact cattle, sheep and deer have on soil physical quality and losses of P and SS in surface runoff. This study will test the hypothesis that under the same field conditions (soil type, slope, climate etc) and with sufficient feed to prevent behavioural issues like fence-line pacing (McDowell *et al.* 2004b), cattle will have a greater deleterious effect on soil physical quality that, in turn, result in increased losses of P and SS in surface runoff compared to deer or sheep. A second experiment was conducted to account for the different time spent grazing depending on feed availability. This experiment investigated the effect simulated cattle grazing intensity (equivalent to 24-96 hr) has on surface runoff volume and P and SS losses. This study tests the hypothesis that an increase in treading intensity increases surface runoff volume and P and SS losses.

5.2 Materials and Methods

5.2.1 Site

The AgResearch Invermay farm near Mosgiel, Otago, New Zealand (NZ Map Grid E 2309050, N 5480010), is located on rolling to steep hill country at an altitude of 150-300 m. The sheep, beef cattle and deer farm has been operating since 1972. Average rainfall is about 735 mm per annum. The soil at the research site is an intergrade between a Porteous silt loam (Brown soil of low structural vulnerability) and a Warepa silt loam (mottled fragic Pallic soil of high structural vulnerability) (Hewitt 1998). The latter soil at the site is characterised by a fragipan in the Cx horizon at about 90-110 cm deep. Sulphur-super fertiliser is applied in October each year at about 24 kg P/ha.

5.2.2 Grazing experiment

Grazing experiment

The grazing experiment ran for 25 months, beginning in April, 2008, when three paddocks were chosen for the field trial. Two paddocks (3.6 and 1.3 ha) were allocated for sheep (Romney) and beef (Angus) cattle grazing and one paddock (1.3 ha) for deer (Red) grazing. Paddocks were adjacent to each other with similar slope (5-7%) and north easterly aspect. Within each paddock, a 20 x 20 m grid sampling was carried out and three replicate soil samples (7.5 cm depth within 50 x 50 cm area) taken at each point (total of 95 points) for Olsen P analysis (see below) and referenced with a global positioning system (GPS). Grazing areas (20 x 20 m) were chosen within each paddock with comparable Olsen P concentrations (10-25 mg/kg). Soil sampling for Olsen P and water soluble P analysis (see below) was repeated in 2009 and 2010 in grazed and ungrazed fenced areas. Twelve samples were also taken in July, 2008 before treatments were imposed for background soil physical data (see below), which included macroporosity and soil bulk density (6) and saturated hydraulic conductivity (6) in each 20 x 20 m area.

Within each sheep and beef cattle paddock, three 20 x 20 m areas were fenced off representing three treatments (Figures 5.1 and 5.2):

1) a control area with nil grazing (hereafter referred to as ‘mixed control’),

2) an area for sheep grazing, and

3) an area for beef cattle grazing.

Figure 5.1. Illustration of the experimental design for cattle and sheep grazing and control plots at Invermay farm. A pair of surface runoff plots (see below and Figure 5.2) are positioned in each fenced off area for cattle, sheep and the mixed control.

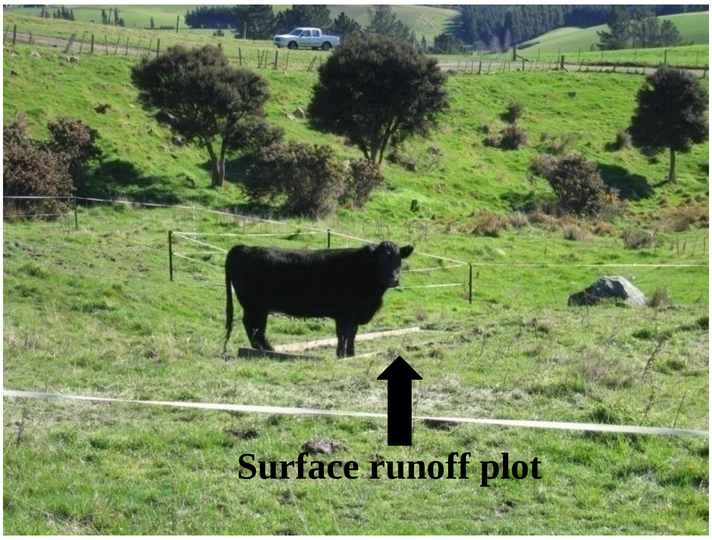

Figure 5.2. Illustration of the dimension of surface runoff plots in cattle-grazed areas at Invermay farm.

Within each grazing area, a pair of surface runoff plots was installed at least five metres apart. Each plot consisted of a 4 m long by 1 m wide section bordered with wooden boards 180 mm wide by 25 mm thick, buried 150 mm into the ground. A metal gutter was attached to the downslope end of the plot and connected to an underground hose. The hose led to a fenced off 50 L vessel which collected surface runoff on a sufficient rainfall event basis. Volumes were converted from litres to mm by dividing collected volumes by surface runoff plot areas.

Within the deer paddock, a 20 x 20 m area was fenced off and a pair of surface runoff plots representing a control/nil grazed area (deer control) was installed as above (Figure 5.3 and 5.4). Another two pairs of plots were installed 20 metres apart from each other and the control area. The deer-grazed surface runoff plots were not fenced off to avoid behavioural characteristics such as fence-line pacing (McDowell *et al.* 2004b).

This design was adopted taking account of logistical and behavioural constraints (e.g. the possibility that cattle would destroy fencing if fenced off individually) and with an emphasis on the interaction of stock type with processes within runoff plots over time.

Figure 5.3. Illustration of the experimental design for deer grazing and deer control plots at Invermay farm. Two pairs of surface runoff plots are unfenced for deer to graze in and around and one pair of surface runoff plots are fenced-off representing the deer control/nil graze.

Figure 5.4. Illustration of deer grazing in and around surface runoff plots at Invermay farm.

Grazing periods in spring and summer lasted between 24-96 hr and were spaced on average 8 weeks apart depending on available feed. Control zones of the three paddocks were mown to the same height as the grazed plots and the clippings removed. Grazings in autumn and winter were spaced up to 12 weeks apart.

The stocking density was kept the same among grazed treatments at 18 stock units per 20 x 20 m (i.e. 75 SU /ha) on an event basis, equivalent to 3 cows, 9 deer and 18 sheep. Supplemental feed, in the form of hay, was provided for stock in winter months if insufficient pasture was available.

The day after each grazing, soil samples were collected for physical analyses in each grazed and ungrazed area, but outside the surface runoff plots. Three stainless steel soil cores (10 cm diameter and 5 cm deep) were allocated to each plot for macroporosity and soil bulk density measurements, and three soil cores (10 cm diameter and 6.5 cm deep) for saturated hydraulic conductivity (see below). When surface runoff was produced it was collected within 24 h of an event.

Rainfall at the site was measured using a tipping bucket. Soil moisture was measured by an Aquaflex^{TM} soil moisture tape buried at 5-20 cm depth (to measure soil water from 0-20 cm depth) recorded downhill from the site. Field capacity (FC) has been defined by McLaren & Cameron (1996) as the soil moisture status ‘after rapid drainage has effectively ceased and soil water content has become relatively stable’. In the current study, FC was determined as the soil moisture content measured at a pressure of -10 kPa or *c.* 43 % v/v in the case for this soil type. Saturation was recorded as 58 % v/v. Surface runoff was defined as that produced under saturation- excess conditions when soil moisture reaches saturation (>58%, v/v) and likely to be produced under these conditions when soil moisture is greater than field capacity but less than saturation (i.e. between 43% and 58%, v/v). Surface runoff produced below FC (i.e. <43%, v/v) was defined as that produced under infiltration-excess conditions.

5.2.3 Soil and surface runoff analyses

Soils were analysed for Olsen P and water soluble P (WSP) concentrations (see section 3.2.3). Soil physical measurements included macroporosity, soil bulk density, and saturated hydraulic conductivity (K_{sat}) (see section 3.2.3).

Due to the high number of runoff samples to be analysed, particularly in and around winter, 200 ml sub-samples were frozen before analysis. Samples were analysed for dissolved reactive P (DRP), dissolved unreactive P (DUP), total dissolved P (TDP), particulate P (PP), total P (TP), and suspended sediment (SS) (see section 3.2.2).

5.2.4 Rainfall simulation

Given that grazing lasted for 24-96 hr, depending on feed availability per grazing event, a rainfall simulation study was conducted to determine if there was a significant effect of the number of treading imprints (used here as a surrogate for time spent grazing) on surface runoff volume and P and SS losses.

Rather than compromise the field plots, Warepa silt loam (mottled fragic Pallic soil) (Hewitt 1998) soil was collected from an adjacent sheep and beef cattle paddock. Intact soil turves were excavated using a purpose-built cutting blade (see section 3.2.1). Soils were trimmed to 7.5 cm depth, pasture trimmed to 5 cm height, and soils placed into metal boxes each under a sheet of plastic liner. Boxed soils were dried or moistened to maintain 50% of available water holding capacity (see section 3.2.1) for one week, after which one of five treatments was employed, each with four replicates:

1. Control with nil grazing;
2. 24 hr grazing (30 hoof imprints/m);
3. 48 hr grazing (60 hoof imprints/m);
4. 72 hr grazing (90 hoof imprints/m); and
5. 96 hr grazing (120 hoof imprints/m).

The number of hoof imprints (McDowell *et al.* 2003b) increased linearly and it is recognised that the degree of overlap may not be equivalent to that seen in the field. Hoof imprints were made using a mechanical hoof (see section 3.2.1).

Soils were then placed under a rainfall simulator (see section 3.2.2) and surface runoff was collected for 30 min, the volume determined and samples analysed for P fractions and SS (as above).

5.2.5 Statistical analyses

Phosphorus and suspended sediment surface runoff losses, Olsen P and WSP were tested for normality and transformed if necessary before being subjected to ANOVA. Terms were fitted for stock type, treatment and the factorial interaction of stock type and treatment, with runoff plot as the block structure, assuming no serial correlation across events. Values for K_{sat} were $\log_{10}$-tranformed before analysis. A gamma log generalised linear model was used to analyse factors affecting concentrations and loads of DRP, PP, TP and SS lost after adjusting for grazing treatment, stock, and time since grazing. Soil physical properties were analysed by REML, fitting overall and stock treatment splines, with stock type, linear and sinusoidal time components and their interactions as fixed effects.

Log runoff was estimated between known data points as a function of soil moisture and log rainfall using kriging, based on a linear fit to the variogram with nugget 1.3 and gradient 5.0. Seasonal loads and concentrations of dissolvable reactive and total P were analysed on the log scale by sinusoidal regression with a period of one year, with an additional term for rainfall.

For the rainfall simulation experiment, concentrations and loads of P fractions and SS in surface runoff were analysed by ANOVA, fitting treatment.

5.3 Results and Discussion

5.3.1 Hydrology and losses of phosphorus and suspended sediment

Surface runoff was produced in fifty events that mostly occurred during winter, with no runoff at all occurring between August 2009 and April 2010 (Figure 5.5). Confirmed saturation-excess (58% v/v), likely saturation-excess (43-58% v/v) and infiltration-excess surface runoff (<43% v/v) contributed on average across all plots 73%, 17% and 8% of total volume. The majority of surface runoff volume (*c.* 90%) was produced from four large winter storm events in and around August, 2008, May, 2009, July, 2009 and May 2010 when soil moisture was at or above FC.

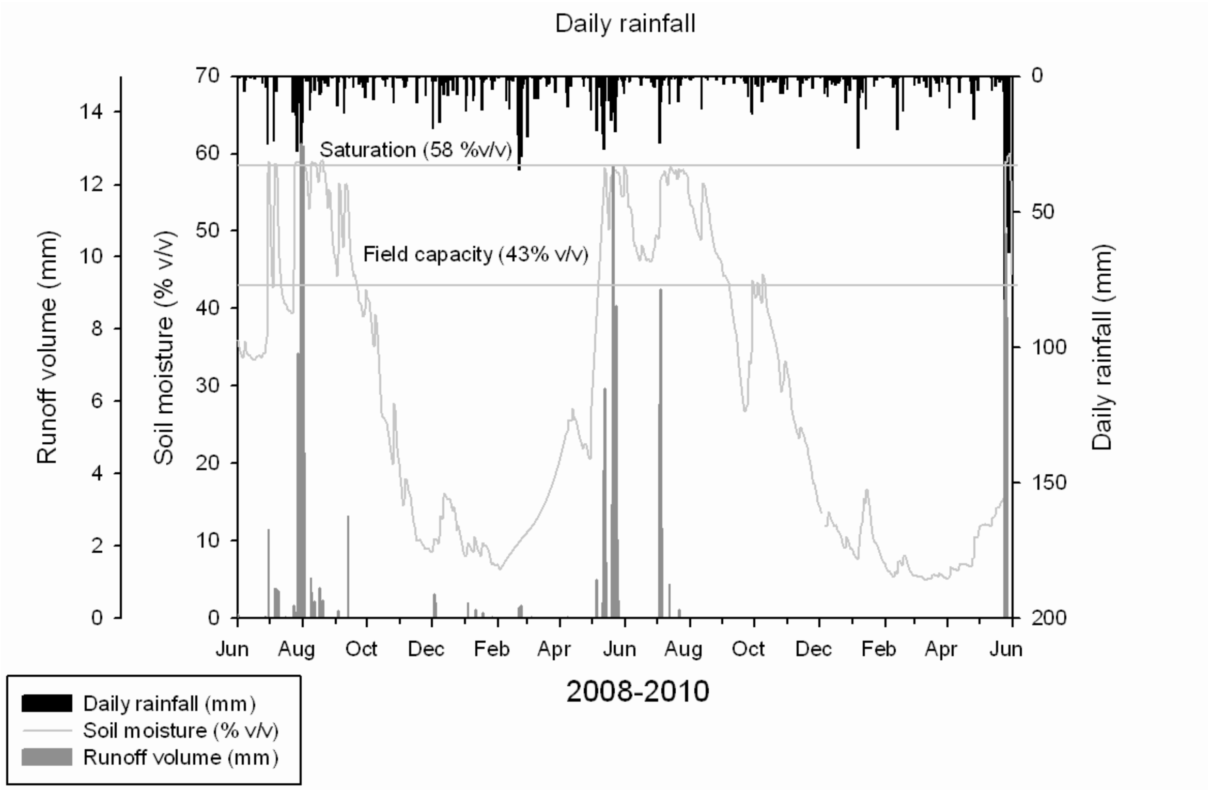

Figure 5.5. Summary of daily rainfall, soil moisture and the mean runoff volume of surface runoff produced for all plots from June, 2008 to June, 2010.

The first major storm event from 31 July to 02 August, 2008 accounted for over 70% SS and about 30% TP losses during the experiment (Figure 5.6). Among P fractions, the DRP load was greatest, closely followed by PP, and least for DUP (Figure 5.6). In the wider 9 ha catchment, McDowell and Srinivasan (2009) attributed three-quarters of P and SS loads in stream flow during 2008 to the three largest storm events and saturation-excess surface runoff which accounted for 50% of annual rainfall.

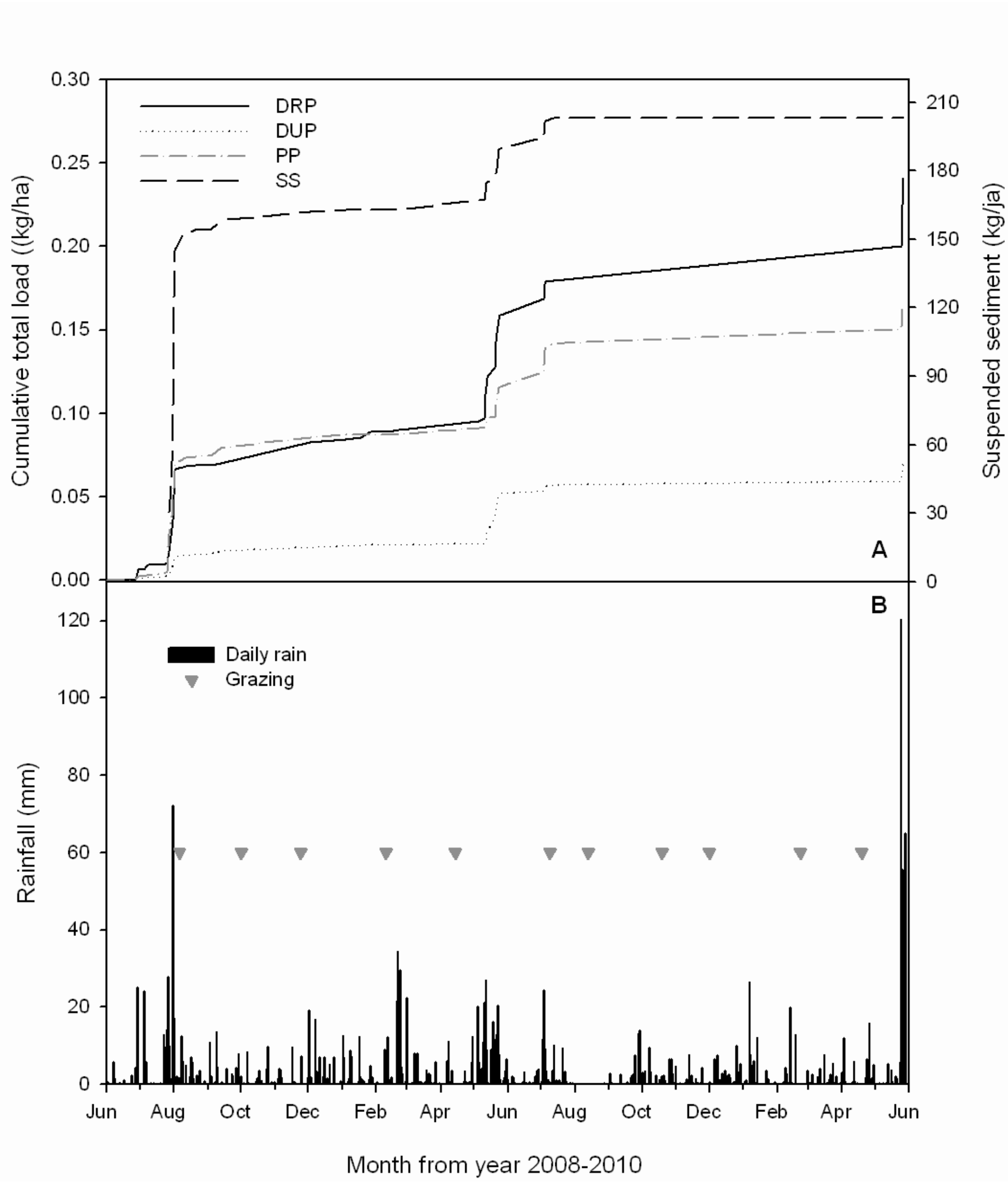

Figure 5.6. Cumulative (A) mean total load of dissolved reactive P (DRP), dissolved unreactive P (DUP), particulate P (PP) and suspended sediment (SS) from cattle, deer, sheep and control plots, and rainfall (B) from June, 2008 – June, 2010, with grey arrows indicating grazing periods.

A surface plot (Figure 5.7) shows runoff increasing with rainfall, while its relationship with soil moisture shows a distinct curvature, with somewhat greater runoff at 10-20% v/v than at 20-30% v/v soil moisture, and greater runoff when soil moisture was >30% v/v. Runoff that occurred when soil moisture was < 20% v/v during significant rainfall events in December 2008 and February 2009 was relatively low and attributed to infiltration-excess surface runoff.

Soils can experience periods of hydrophobicity under dry spells (i.e. < 20% v/v) and may repel any water it initially encounters enhancing infiltration-excess surface runoff (Doerr *et al.* 2000). On a wider catchment scale, McDowell and Srinivasan (2009) found that infiltration-excess surface runoff was an important mechanism of P and SS loss, but largely associated with impervious areas like tracks and lanes. These data add weight to these findings by showing that outside of impervious areas in the majority of the paddock, most surface runoff is derived from saturation-excess conditions with only small volumes generated from infiltration-excess conditions.

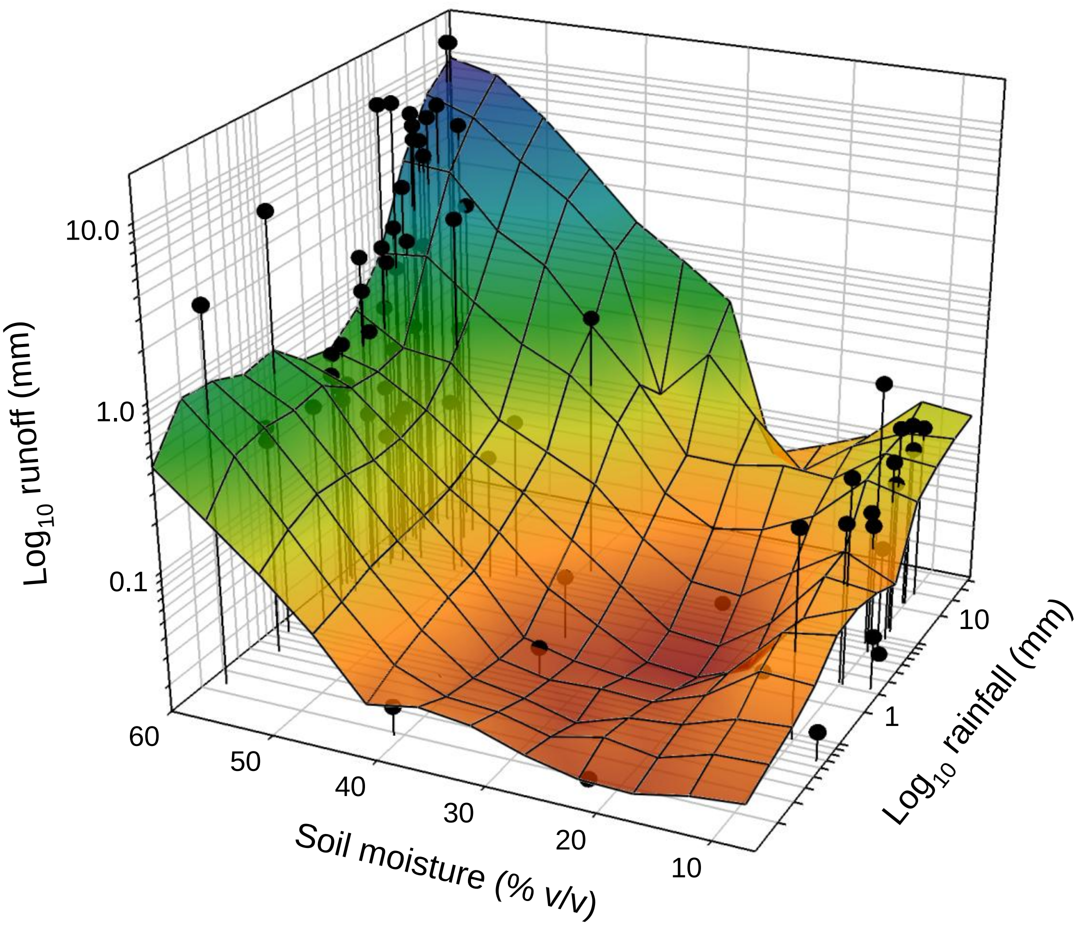

Figure 5.7. Interaction of rainfall and soil moisture on the mean depth of runoff produced from the plots.

A seasonal effect was observed where the greatest P loads occurred in winter and the greatest P concentrations in dryer summer months (Figure 5.8). The effect of rainfall was significant for loads, but not for concentrations. Such seasonality is not uncommon due to the temperate climate and the likelihood of runoff in winter months. However, during irregular rainfall events in summer months, which produced runoff, concentrations have been reported to be enriched as a result of low volumes and the greater incidence of available P like dung deposits (Girmay *et al.* 2009; Holz 2010; McDowell and Srinivasan 2009). Curran Cournane et al. [(2010b)- Chapter 3] attributed increased DRP losses at decreased soil moisture (10% available water holding capacity) to the death and decay of soil microbial biomass with desiccation or the lack of P dilution upon drying. Although more P may be lost during winter, losses in summer can also be detrimental to surface water bodies due to the greater potential for algal growth (i.e. increased light and temperature) (Carpenter *et al.* 1998).

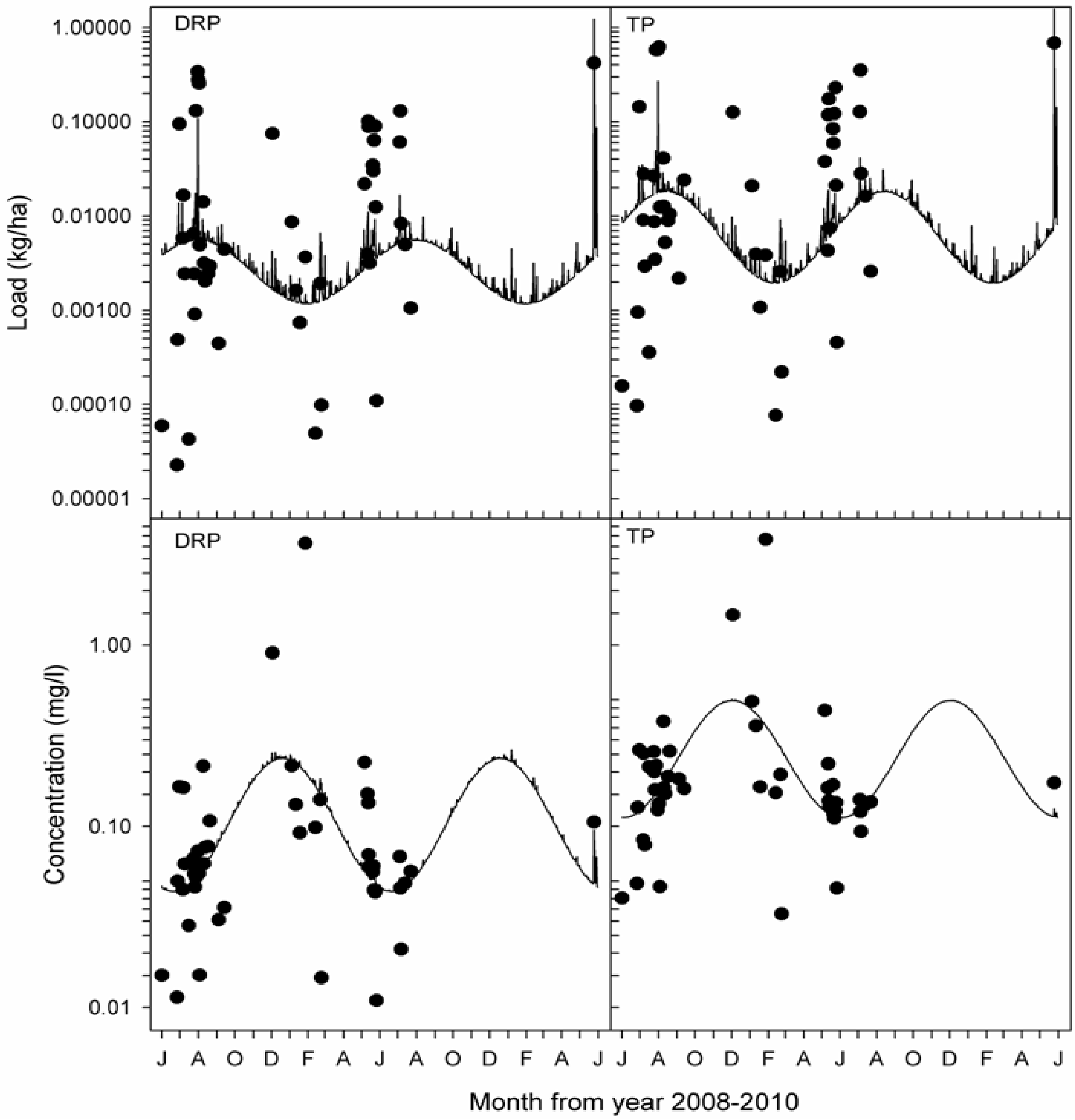

Figure 5.8. Seasonal distribution of mean loads and concentrations of dissolved reactive P (DRP) and total P (TP) lost from the mean of all plots from June, 2008 to June, 2010. Observed data are represented by points, while the fitted model, from terms for an annual sinusoid and rainfall, is represented by a line.

5.3.2 Stock type and losses of phosphorus

Paddocks were grazed 11 times between August, 2008 and April, 2010. No significant differences were found between mean concentrations and loads of P fractions and SS lost between stock types and the control plots. No behavioural characteristics for stock type were exhibited throughout the trial period such as camping for sheep and cattle (Lucci *et al.* 2010; Melland *et al.* 2008; Niu *et al.* 2009) or fence line pacing or wallowing for deer (McDowell *et al.* 2004b), all of which have previously been reported as major sources of P and SS losses.

However, seasonal variation and interaction with soil physical properties with stock type was evident (see below). Mean differences in losses of concentrations for TP varied between about 0.4-0.61 mg/L and 0.18-0.28 g/L for SS losses (Table 5.1). The mean concentration of Olsen P taken in autumn, 2008 was 20 mg/kg, while Olsen P and WSP concentrations measured from spring, 2009 were 26 mg/kg and 0.074 mg/L, and autumn, 2010 were 35 mg/kg and 0.099 mg/L, respectively. There was no significant differences for Olsen P and WSP for stock type and control treatments.

Table 5.1. Effect of stock type on mean concentrations of P fractions (dissolved reactive P, DRP; dissolved unreactive P, DUP; particulate P, PP; total P, TP) and suspended sediment (SS). The standard error of difference (SED) is given for comparison of cattle and mixed control. Significance is indicated at the 5% level (ns = not significant).

	DRP (mg/L)	DUP (mg/L)	PP (mg/L)	TP (mg/L)	SS (g/L)
Cattle	0.198	0.056	0.196	0.45	0.189
Deer	0.178	0.060	0.164	0.402	0.177
Sheep	0.185	0.050	0.203	0.438	0.283
Mixed control	0.199	0.046	0.174	0.419	0.202
Deer control	0.228	0.054	0.323	0.605	0.242
SED	0.062^{ns}	0.012^{ns}	0.085^{ns}	0.109^{ns}	0.057^{ns}

5.3.3 Soil physical quality

Soil physical quality showed a strong annual cycle (Figure 5.9), with soil physical quality generally better in summer than in late winter. The collection of soil physical samples one-day post-grazing will represent a worst-case scenario for grazed plots for this soil type. Cattle grazing had the greatest negative effect on soil physical quality, as indicated by the linear trend (P<0.01 for stock type by time interaction), while the mixed control treatment showed an improvement over time. Similarly, when 97 sheep and 87 dairy farms were surveyed in Southland and South Otago, New Zealand, soil macroporosity, bulk density, air permeability and hydraulic conductivity in dairy farms were classified as impaired compared to sheep farms (Drewry *et al.* 2000a). However, since deer could not be restricted to the 20 x 20 m grazing area, grazing near and within plots was not as consistent as cattle or sheep grazed plots. This will have affected soil physical data for the comparison of stock type, but is reflective of deer grazing behaviour. In other areas of the deer paddock, fence-line pacing was evident and soil physical quality likely impaired (McDowell *et al.* 2008). Therefore, although soil physical quality can be impaired in paddocks where deer pace fence-lines, these data suggest that in other areas soil physical measurements are similar to those under sheep grazed pasture (Figure 5.9).

During the trial, macroporosity and K_{sat} for the two controls were generally greater and soil bulk density lower than grazed treatments, but also varied over time - generally better in summer than in late winter (Figure 5.9). This temporal variation has been attributed to a number of processes including: shrinkage and swelling, dispersion and slaking due to rapid wetting (Ben-Hur *et al.* 2009), frost heave, plant root penetration, earthworm burrowing, and management practices (Drewry 2006). Although both grazed and ungrazed treatments exhibited a similar trend in K_{sat}, macroporosity or soil bulk density during the trial, it was evident that the difference between grazed and ungrazed treatments increased over time (Figure 5.9). This suggested that management of stock was exhibiting a major control in the recovery of K_{sat}, macroporosity or soil bulk density in ungrazed pasture.

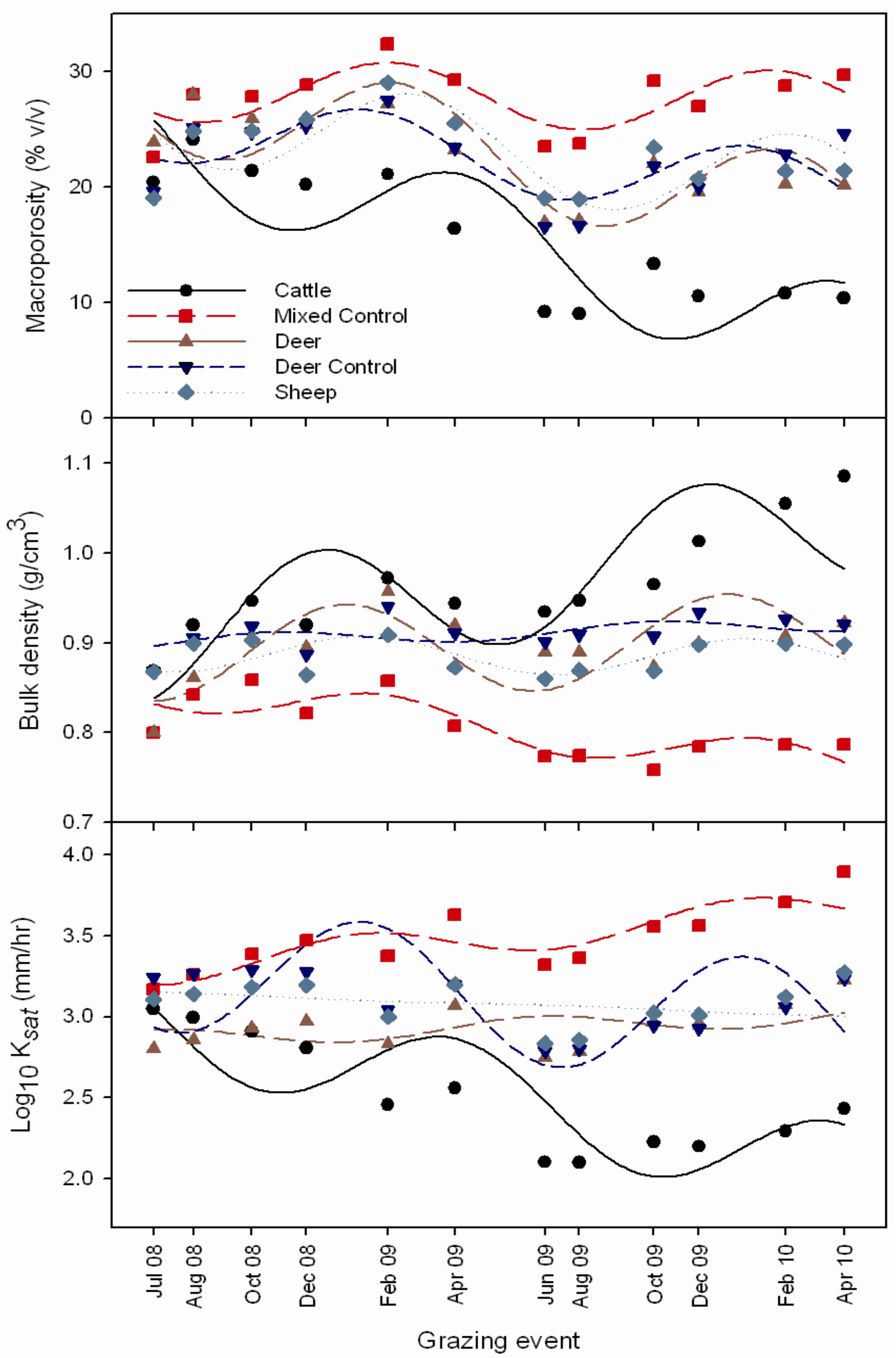

Figure 5.9. Effect of stock type on mean soil (0-5 cm depth) macroporosity, bulk density and $\log_{10}$ saturated hydraulic conductivity for all plots. Observed data are represented by points and the fitted model, for a sinusoidal time component, is represented by lines. In all cases the linear contrast for cattle was significantly different from zero (*P* < 0.01) indicating a deterioration in soil physical quality for cattle that was not observed for the other treatments.

5.3.4 Factors affecting losses of phosphorus and suspended sediment

A significant relationship was apparent between macroporosity or time since grazing and the concentrations or loads of DRP and TP lost (Figure 5.10 and 5.11). Similarly, McDowell *et al.* (2003b) reported a relationship between macroporosity and concentrations or loads of DRP, TP, and SS for grassland and cultivated soils, although this was based on rainfall simulation data. The relationship was explained by an increase in saturation-excess surface runoff and P losses due to a decrease in large soil pores, or soil water storage before runoff began. These findings confirmed that macroporosity is a valid measurement for environmental assessment (McDowell *et al.* 2003b) as well as an important tool for measuring potential agronomic impacts (Drewry *et al.* 2008; Houlbrooke *et al.* 2009b). Data presented in Figure 5.11 also highlighted decreased losses of concentrations and loads of DRP and TP with days since grazing; the same relationship was also true for DUP and SS (data not shown). The decrease could either be due to the recovery of soil physical properties like macroporosity (Figure 5.10 and 5.12), removal of stock (Drewry *et al.* 2006; Nash and Halliwell 1999), or to a decrease in the availability of P loss from dung [(Curran Cournane *et al.* 2010a)- Chapter 4].

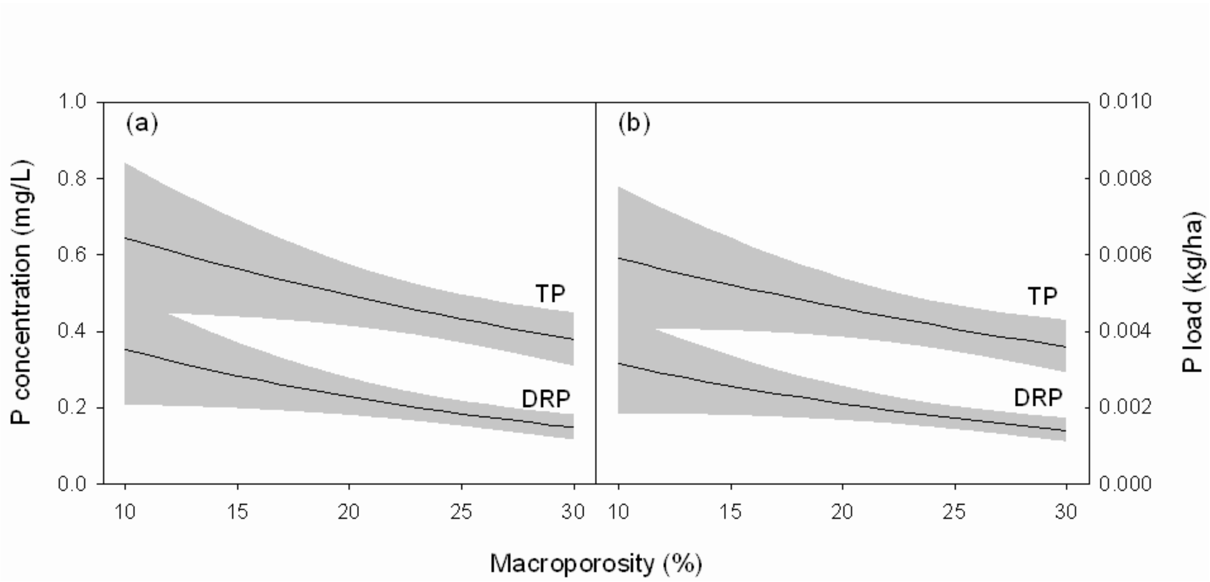

Figure 5.10. Regression of mean concentrations (a) and loads (b) of dissolved reactive phosphorus (DRP) and total phosphorus (TP) in surface runoff against % v/v soil macroporosity (0-5 cm depth) for all plots. Shading represents 95% confidence intervals.

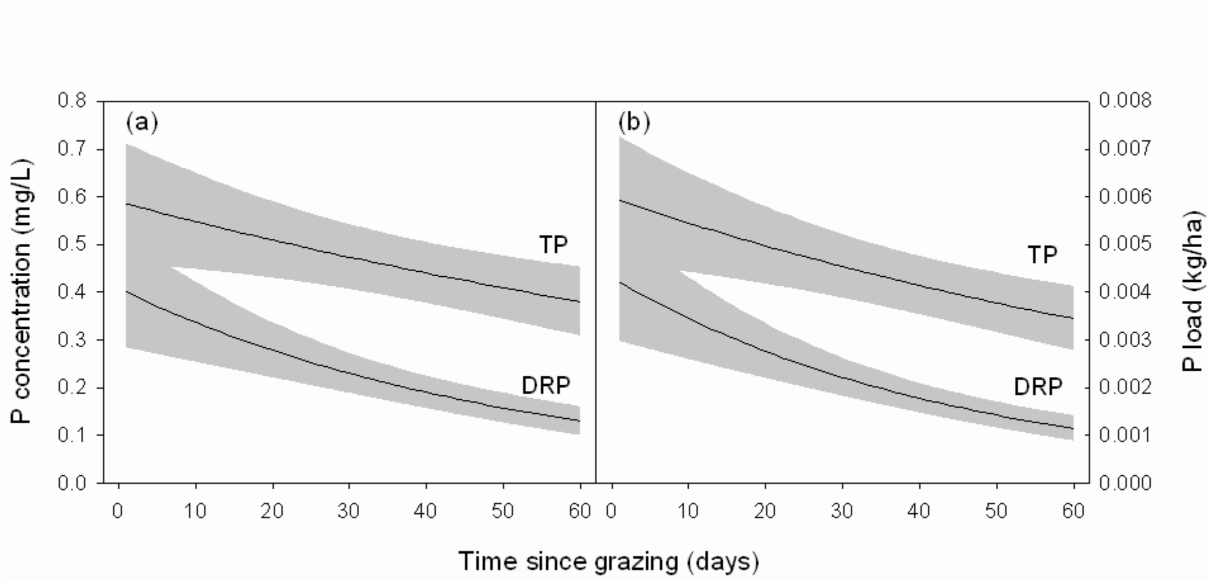

Figure 5.11. Regression of mean concentrations (a) and loads (b) of dissolved reactive phosphorus (DRP) and total phosphorus (TP) in surface runoff against time (days) since grazing for all plots. Shading represents 95% confidence intervals.

Similar to macroporosity, an inverse relationship was noted between the load of PP and SS and K_{sat} (Figure 5.12). McDowell *et al.* (2003b) also reported that an increase in treading decreased K_{sat}, producing greater volumes of surface runoff. These results are not unexpected since a decrease in soil infiltration rate would produce a greater volume of surface runoff. However, the majority of surface runoff produced in the current study occurred under saturation- not infiltration-excess conditions. No relationship was found between K_{sat} and the volume of runoff, suggesting that the relationship between K_{sat} and PP or SS losses may have been due to the concomitant decrease in macroporosity, especially in winter.

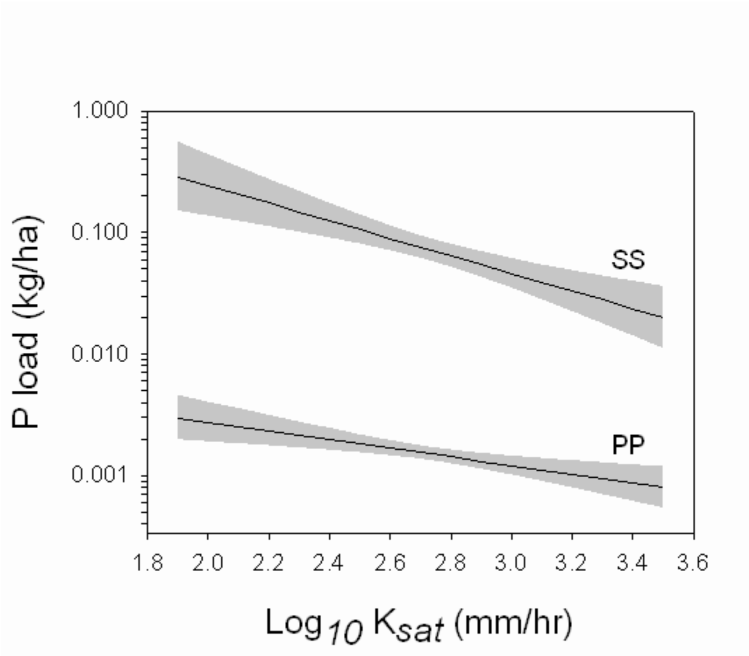

Figure 5.12. Regression relationship between mean $\log_{10}$ saturated hydraulic conductivity (Log_{10} K_{sat}; 0-5 cm depth) and the load of particulate phosphorus (PP) and suspended sediment (SS) lost during surface runoff for all plots. Shading represents 95% confidence intervals.

5.3.5 Simulated rainfall

Due to the grazing of stock between 24-96 h, depending on feed availability per grazing event, a rainfall simulation study was conducted to determine if there was a significant effect of increasing treading on surface runoff volume and P and SS losses. Results showed that, while there were no significant differences in treading intensity for losses of P species, the loads and concentrations of SS (P<0.001) and surface runoff volumes (P<0.05) increased with treading intensity (Table 5.2). Significant P losses have been attributed to the contribution of cattle dung as opposed to treading activity [Chapter 4- (Curran Cournane *et al.* 2010a)]. Since no dung was applied, these data concur with McDowell *et al.* (2003b), who concluded that the most likely effect of treading on P and SS losses is therefore associated with changes in soil physical properties like macroporosity, and in turn surface runoff processes.

Table 5.2. Effects of simulated cattle treading intensity on mean concentrations and loads of P fractions (dissolved reactive P, DRP; dissolved unreactive P, DUP; particulate P, PP) and suspended sediment (SS) and volume lost in surface runoff during the rainfall simulation. The standard error of difference (SEDs) and significance levels are given for the main effect of the grazing intensity treatment; ns= not significant, *, ** and *** represent significance at the *P*<0.05, *P*<0.01 and *P*<0.001 level, respectively.

Grazing intensity (hr)	DRP		DUP		PP		SS		Volume
	mg/L	kg/ha	mg/L	kg/ha	mg/L	kg/ha	g/L	kg/ha	L
0	0.21	0.015	0.03	0.002	0.04	0.003	0.012	0.106	1.14
24	0.20	0.016	0.03	0.003	0.04	0.003	0.020	0.192	1.31
48	0.25	0.017	0.02	0.002	0.03	0.002	0.019	0.172	1.13
72	0.21	0.015	0.04	0.003	0.04	0.003	0.045	0.380	1.10
96	0.21	0.019	0.02	0.002	0.06	0.005	0.055	0.322	1.44
SED	0.023^{ns}	0.0022^{ns}	0.011^{ns}	0.0008^{ns}	0.014^{ns}	0.0011^{ns}	0.0086^{***}	0.057^{***}	0.115^{*}

5.3.6 Management implications

For management, it appears that for this trial it was a case of not '*what'* stock type grazed, but '*when*' stock grazed paddocks relative to soil moisture that influenced P and SS lost in surface runoff. Although these data indicate that concentrations of P lost during spring/summer in infiltration-excess surface runoff are enriched, loads were small compared to loads lost via infiltration-excess runoff from impervious areas like tracks, lanes and stock camping areas (McDowell and Srinivasan, 2009). Compared to impervious areas, there is limited scope to manage for infiltration-excess surface runoff from pasture. In contrast, with most runoff occurring in winter, management to decrease P and SS losses could be driven by weather forecasting and soil saturation sensors [e.g. as run by some provincial government agencies in New Zealand; (ORC 2010)] in combination with strategies like restricted grazing (de Klein *et al.* 2006).

5.4 Conclusions

Soil physical data indicated differences between stock types and relationships between P and SS concentrations and loads with changes in macroporosity, K_{sat} and time since grazing. However, this did not translate into significant differences in P and SS losses in surface runoff between stock types. The greatest volume of surface runoff occurred under saturation-excess conditions in winter. The volume of infiltration-excess surface runoff during dryer periods was minimal, but concentrations of P and SS were enriched and can pose an environmental risk due to warmer temperatures and better light for algae to grow. While there is limited scope to manage for such infiltration-excess surface runoff losses from pasture, mitigation strategies such as restricted or nil grazing in winter when soil moisture has reached field capacity are recommended to minimise P and SS loss to surface water, regardless of stock type.

Chapter 6- Is mechanical soil aeration a strategy to alleviate soil compaction and decrease phosphorus and suspended sediment losses from irrigated and rain-fed cattle-grazed pastures?

6.1 Introduction

Losses of phosphorus (P) and suspended sediment (SS) can impair surface water quality (1998). Knowledge regarding the impact mechanical soil loosening equipment, commonly known as subsoilers or aerators, on contaminant losses is limited. When comparing non aerated soil to slit aerated soil (tines pushed into the soil to make elongated holes), work has reported that aeration effectively decreased DRP and TP losses for well-drained soils in West Virginia (Shah *et al.* 2004) and decreased runoff volume and DRP losses for well drained soils in Georgia, USA (Franklin *et al.* 2007). Decreases in volume and concentrations were attributed to increased infiltration of rainfall and binding of P with soil minerals. In contrast for poorly drained soils, soil aeration increased runoff volumes (4.8 mm/runoff event) and losses of DRP and TP (0.25 kg TP/ha per event runoff) when compared to a non-aerated soil treatment (Franklin *et al.* 2007). Similarly, whereas Butler *et al.* (2008a) reported decreased P losses with soil aeration (in particular for core aeration by cylindrical cores) compared to non-aerated soil in Georgia, Pote *et al.* (2003) reported that aerated soil plots with surface-applied poultry litter did not significantly decrease concentrations of soluble P in runoff compared to non-aerated plots in Arkansas, USA. In New Zealand, Houlbrooke *et al.* (2005) reported that the loss of farm dairy effluent (FDE) in surface runoff was less from aerated soil than from non-aerated plots due to improved surface infiltration rates in the former treatment.

From an agronomic standpoint to increase pasture growth, several authors have advocated the use of aerators to improve macroporosity and infiltration rates on soils that are susceptible to treading with improvements lasting for ten months (Burgess *et al.* 2000) up to 2.5 years (Drewry *et al.* 2000b). The benefits of aeration can be soil type dependent with improvements for some soils quickly reverting back to its original state shortly after subsoiling (Houlbrooke 1996), particularly dispersive soils (Adcock *et al.* 2007).

Although informative, these studies have not sampled or comprehensively examined the role soil physical properties play in soil hydrology and runoff events. This study will consider how soil physical quality under both aerated and non aerated soil treatments might influence P and SS losses in surface runoff. Since both surface infiltration rate and soil compaction influence surface runoff losses of P and SS [(Curran Cournane *et* *al.* 2011)- Chapter 5], this study tests the hypothesis that mechanical soil aeration could decrease volumes of surface runoff and losses of P and SS for a soil both poorly draining and susceptible to treading damage.

6.2 Materials and Methods

6.2.1 Site details and treatments

The trial was located on a silt loam [a Mottled Fragic Pallic soil (Hewitt 1998) or an Aeric Fragiaquept, USDA taxonomy classification] located near Windsor, North Otago, New Zealand (2334000E, 5573900N). The soil is poorly structured and vulnerable to cattle treading (Hewitt 1998). The area has a historic mean annual rainfall of 550 mm/yr, but annual rainfall for the trial period (01 May, 2009-31 May, 2010) was 696 mm, with one event (26-28 May, 2010) delivering more than 200 mm of rainfall. Such an event has, on average, a return period of 40 years (NIWA 2010a). The site has a mean July temperature of 5°C and mean January temperature of 15°C.

The site chosen had an approximate slope of 15° and had previously been used to investigate soil physical quality under irrigated and dryland winter forage crops that were grazed by cattle (Houlbrooke *et al.* 2009b). Two previously irrigated plots (~10 m wide and 25 m long) were chosen for the trial. The plots were aligned adjacent to one another on a similar slope and aspect. In December, 2008, soil was mechanically loosened (aerated) to a depth of 20 cm using a 'Clough panaerator' with winged tips. The 'Clough panaerator' is pulled through the soil by a tractor and the winged tips shatter the soil to the depth of operation. Soil in one half of each plot (assigned randomly as upslope or downslope) was aerated with the remaining half of the plot left undisturbed (control). A ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) pasture mix was sown at a rate of 20 kg/ha in January 2009 to the plots and wider site. Pasture was established by direct drilling. Urea (80 kg/ha) was applied in March and December, and 45 kg P/ha was applied as superphosphate in November 2009. Following soil aeration in December, 2008, pasture was lightly grazed twice by sheep (20-April and 18-May 2009) at a stocking density of *c.* 1700 sheep/ha, to control pasture growth. However, cattle grazing commenced on the 23-July at a stocking density of *c.* 350 cattle/ha when 2500-3000 kg DM/ha pasture mass was reached (determined using a plate meter). A total of nine grazings by cattle occurred during the study period.

Soil moisture content was determined by an Aquaflex^{TM} soil moisture tape buried at 5-20 cm and located adjacent to the plots. Field capacity (FC) (see section 5.2.2) was determined as the soil moisture content measured at a pressure of -10 kPa or *c.* 39 % v/v in the case of the Mottled Fragic Pallic soil. Saturation was recorded as 55 % v/v. Surface runoff was defined as that produced under saturation-excess conditions when soil moisture reaches saturation (> 55 % v/v) and likely to be produced under these conditions when soil moisture is greater than field capacity but less than saturation (i.e. between 39 % and 55 % v/v). Surface runoff produced below FC (i.e. < 39% v/v) was defined as that produced under infiltration-excess conditions.

All pasture was irrigated using fixed low application rate (< 2 mm/hr) sprinklers when soil moisture contents fell to 50% of available water holding capacity. From May, 2009 to May, 2010, 12 irrigation events occurred, delivering 432 mm of water during the growing season, nominally defined between September and April each year. On average, each irrigation event lasted 20 hr (range; 14-32 hr) delivering, on average, 33 mm water (range; 22-83 mm). Rainfall at the site from the beginning of May 2009 to the end of May 2010 was 696 mm, with approximately 420 mm falling outside the growing season.

Five surface runoff plots (2 m x 1 m, see section 5.2.2) were installed across the two aerated plots, three in one and two in the other, and another five runoff plots were installed within the two non-aerated (control) plots, in a same manner (Figure 6.1). The surface runoff plots had the same aspect and were staggered in elevation over two meters to avoid one plot influencing the hydrology of another. The surface runoff plot was connected to a 20 L collection vessel via an underground hose. When the 20 L vessels were full and soil moisture content saturated (i.e. greater or equal to 55% v/v) rainfall volume and surface runoff plot dimensions were used to infer excess surface runoff volumes greater than 20L. This occurred during the events 27-29 September, 2009 and 27 May, 2010. Volumes were converted from litres to mm by dividing collected volumes by surface runoff plot areas.

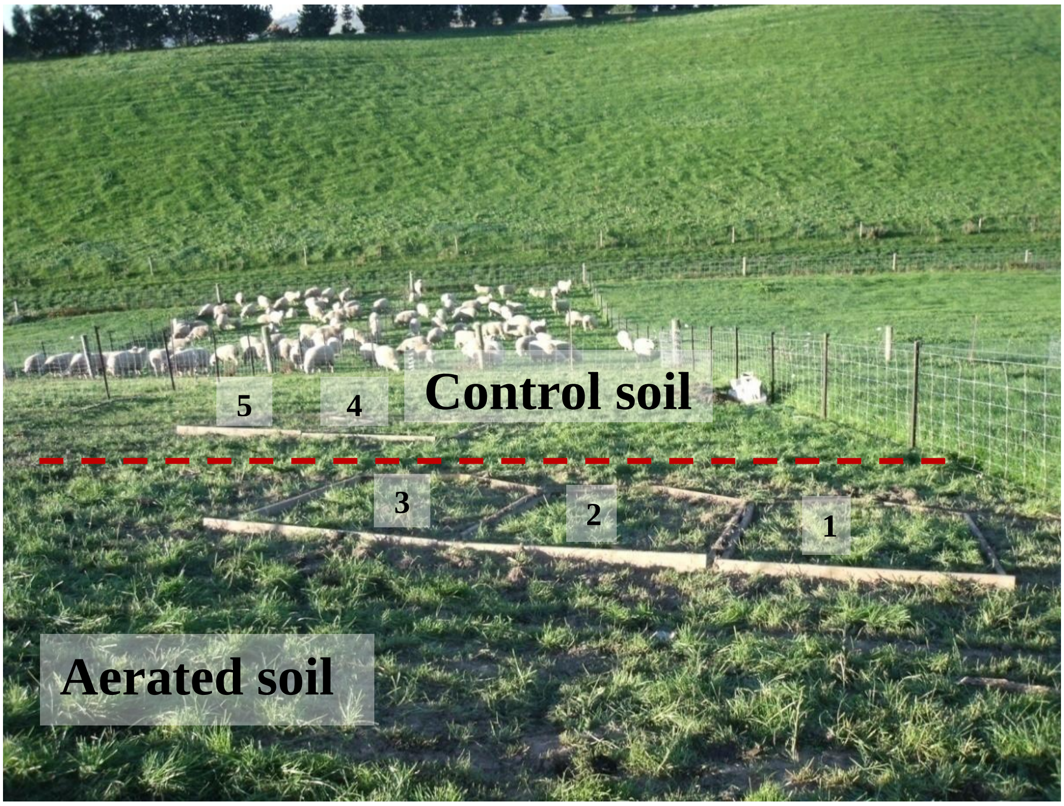

Figure 6.1. Illustration of trial set-up for aerated and control surface runoff plot treatments in one cattle-grazed plot and vice-versa in an adjacent plot at Windsor.

6.2.2 Soil and surface runoff analyses

Seven days after soil aeration in December 2008, samples for macroporososity and soil bulk density were taken from the aerated and control plots. Six months thereafter these measurements were repeated and included determination of saturated hydraulic conductivity (K_{sat}, mm/hr), taken within a metre outside each surface runoff plot (three replicates of each) for further 'background' information. These samples (see section 3.2.3) were taken after each grazing event (two in total- 23 July and 02 September 2009) before irrigation events commenced on the 09 of September 2009. Thereafter, a day before each irrigation event, soil physical samples also included determination of unsaturated hydraulic conductivity [(K_{unsat}, mm/hr), i.e. soil pores >300 µm excluded from the flow process (Greenwood and McNamara 1992)].

Soil chemical status was also determined. Thirty topsoil (7.5 cm depth) samples for Olsen P and water soluble P (WSP) were analysed (see section 3.2.3) at the beginning of the trial and sampled again at the end of the trial.

Due to the high number of runoff samples to be analysed, 200 ml sub-samples were frozen before analysis. Sub-samples were analysed for dissolved reactive P (DRP), dissolved unreactive P (DUP), total dissolved P (TDP), particulate P (PP), total P (TP), and suspended sediment (SS) (see section 3.2.2).

6.2.3 Statistical analyses

Soil physical properties, P fractions and SS, Olsen P and WSP were tested for normality and transformed if necessary before being subjected to ANOVA, with runoff plot within subplot as the block structure, and aeration treatment as the treatment structure as well as event and the factorial interaction of treatment and event, assuming no serial correlation. A gamma log generalised linear model was used to analyse factors affecting concentrations and loads of DRP, PP, TP and SS lost in relation to time (days) since grazing up to 35 days, which excludes only the fertiliser event and the last storm event.

6.3 Results and Discussion

6.3.1 Hydrology and soil physical conditions

Surface runoff was produced from 32 events measured between May 2009 and May 2010. Twenty were rain-fed producing events (mean = 15.2 mm) with the last event, 27-28 May, 2010, contributing to 73% of the rain-fed runoff (Figure 6.2). Irrigation contributed to 12 of these surface runoff events (mean = 4.6 mm) with each irrigation event inducing runoff of various volumes across surface runoff plots.

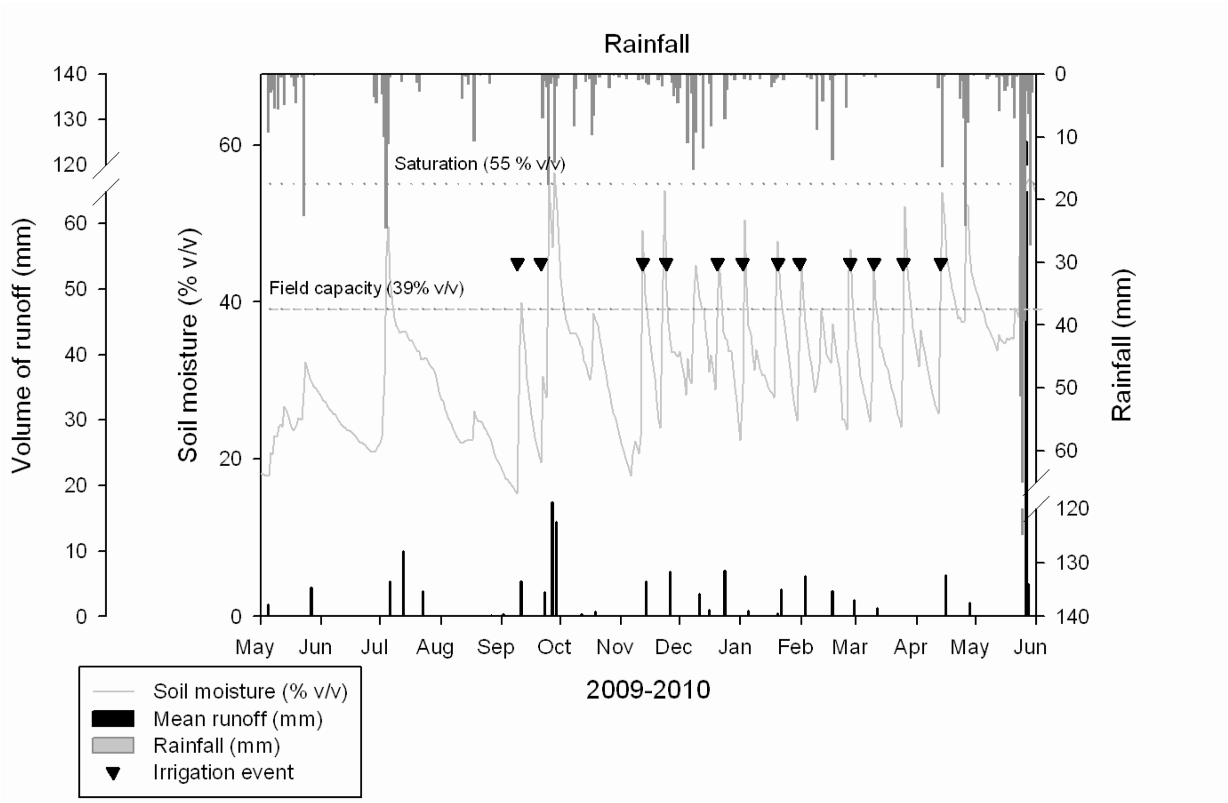

Figure 6.2. Summary of daily rainfall, soil moisture contents and the mean volume of surface runoff for all plots produced per event, from May, 2009 to May, 2010.

Before the last storm event, surface runoff generated when soil moisture was likely saturation-excess (i.e. between 39-55 % v/v) and infiltration-excess (i.e. < 39 % v/v) were almost equal (26% and 22% of mean total surface runoff volume, respectively). The last storm event caused the majority of saturation-excess runoff (i.e. > 55 % v/v) which represented 52% of the total surface runoff measured (Figure 6.1). Saturation-excess surface runoff has been found to be an important pathway for P losses in Southland and Otago (Smith and Monaghan 2003a), particularly for Pallic soils which are commonly known to have poor drainage properties (Hewitt 1998). Excluding the last event, these findings indicated that infiltration-excess surface runoff was also an important runoff process. While the mean K_{unsat} value for the control treatment was significantly greater (1.44 mm/hr) than observed for the aerated treatment (0.77 mm/hr; SED 0.27; $P < 0.05$), mean flow rates never exceeded 4 mm/hr (Figure 6.3). The increase in K_{unsat} for the control treatment largely reflected one soil sampling event, 381 days after mechanical aeration, and the data from one sampling should not be interpreted to infer that mechanical aeration effectively decreased K_{unsat}. K_{sat} rates for both aerated and control treatments decreased significantly ($P < 0.05$) from background rates after initial cattle grazing and tended to stay low over time (Figure 6.3). Rates were occasionally below 10 mm/hr, which would suggest that infiltration-excess surface runoff did occur. A similar pattern emerged for macroporosity and soil bulk density whereby soil physical quality deteriorated following the onset of cattle grazing (Figure 6.2).

On 18 December 2008, seven days after soil aeration was carried out, mean soil macroporosity was significantly greater ($P < 0.001$) in the aerated treatment than control (28 vs. 11% v/v, respectively). Soil bulk densities were accordingly less in the aerated treatment than control (1.05 vs. 1.39 g cm^{-3}, respectively). However, no significant differences were noted between treatments for bulk density and macroporosity six months after soil aeration which was largely attributed to the re-settling of the poorly structured soil (Figure 6.3). The pasture had been lightly grazed twice by sheep within these six months. These two events may have also influenced the re-settling of the soil to some degree, but not to the same extent as that caused by cattle grazing. Seven months after soil aeration was carried out, a significant decrease in macroporosity and increase in bulk density were observed for both treatments, which coincided with the onset of cattle grazing. It has previously been reported that improvements brought about by aeration can be lost within nine months, or sooner if the soil has poor structure and/or drainage (Burgess *et al.* 2000). Burgess *et al.*,(2000) attributed the reversion of soil physical properties of aerated soil to non-aerated soil as a result of soil re-packing as well as re-compaction from cattle grazing. Short-lived improvements may also be further explained, for example, by soil dispersion (Adcock *et al.* 2007). For dispersive soils, aeration alone has been reported to unlikely sustain long-term improvements in soil structure unless the addition of gypsum or a build-up of organic matter occurs to stabilise the structure of dispersive soils (Adcock *et al.* 2007).

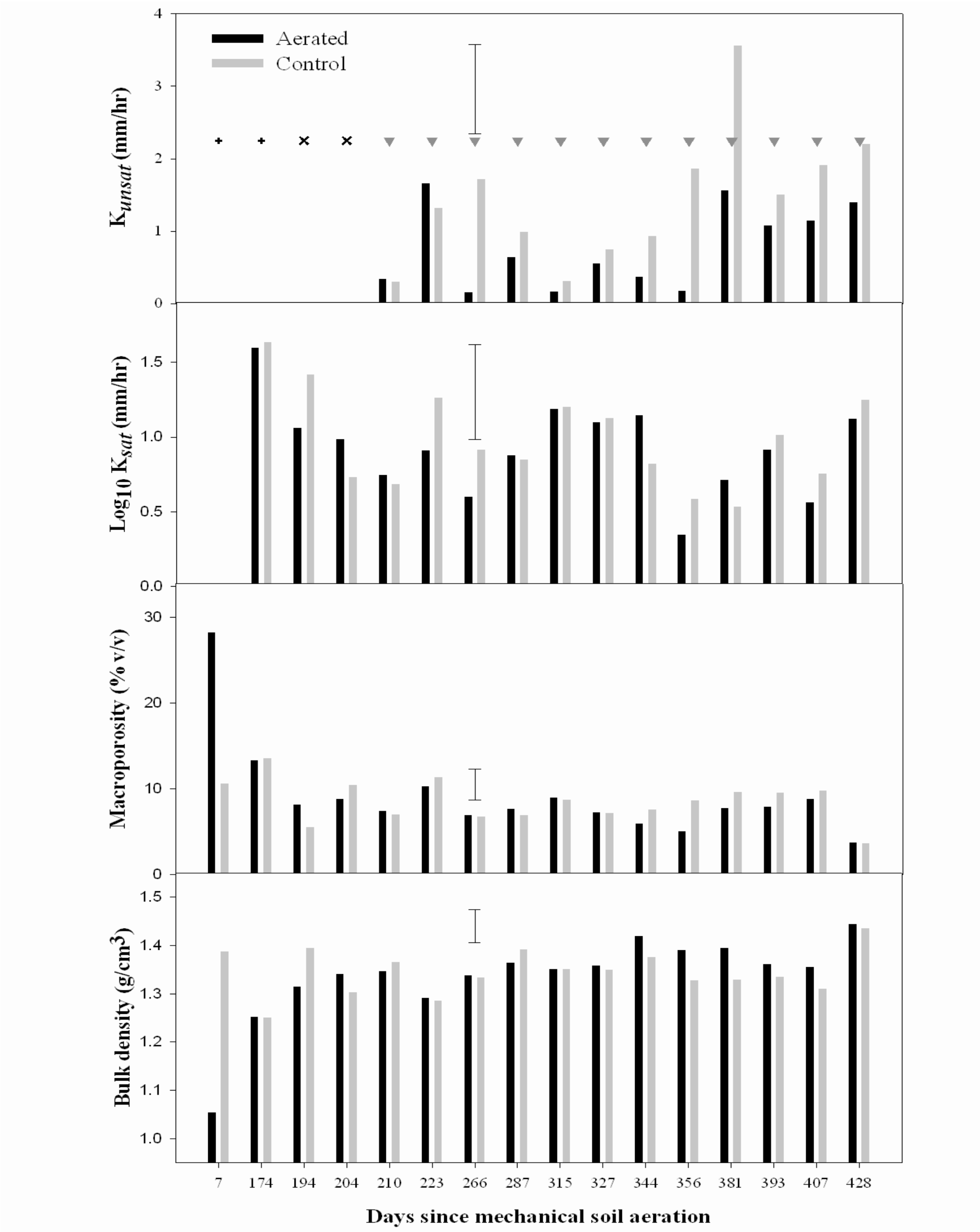

Figure 6.3. Mean soil unsaturated hydraulic conductivity (K_{unsat}), saturated hydraulic conductivity ($\log_{10}K_{sat}$,), bulk density and macroporosity in aerated and control treatments. The least significant difference (LSD_{05}) is given for the interaction of aerated and control treatments and days since mechanical soil aeration. Note that +, × and ▼ indicate background events in 2008 and 2009, grazing and irrigation events, respectively.

In contrast to Houlbrooke *et al.* (2005), who reported less surface runoff of farm dairy effluent applied to aerated soils compared to non-aerated soils, no significant difference was found for the mean volume of surface runoff generated between treatments (12 and 10 mm, respectively). Considering that the majority of surface runoff was produced under saturation-excess conditions, improving infiltration would unlikely decrease volume losses compared to a site that was dominated by infiltration-excess conditions. Furthermore, in the study of Houlbrooke *et al.* (2005), only two months had elapsed before surface runoff occurred, whereas in this study, surface runoff was generated six months after aeration occurred, at which time soil physical properties were not significantly different between treatments. The long period following the aeration treatment was required to optimise plant establishment and maintain, improved soil physical condition. These findings would suggest these soil types can re-compact within six months of mechanical soil aeration.

6.3.2 *Losses of phosphorus and suspended sediment*

For flow-weighted mean concentrations and annual loads, most P was lost as DRP, closely followed by particulate P (PP) and lastly dissolved unreactive P (DUP) (Table 1 and Figure 6.4). The enrichment of DRP was attributed to the last rainfall event which may have caused a prolonged period of saturation causing anoxia and the dissolution of DRP [Chapter 3-(Curran Cournane *et al.* 2010b; Franklin *et al.* 2007)], Figure 6.4. The high proportion of dissolved P loss is consistent with similar studies on irrigated pastures in southern Australia with most P being mobilised by dissolution rather than detachment processes (Nash *et al.* 2002).

Table 6.1. Mean annual loads and flow weighted mean concentrations of P fractions (dissolved reactive P, DRP; dissolved unreactive P, DRP; particulate P, PP; total P, TP) and suspended sediment (SS) for aerated and control treatments, including the last storm event in May, 2010 (A) and excluding the last storm event (B). Standard error of difference (SED) and *F*-statistic is given for the comparison between treatments.

A) Including the last storm event

Treatment	DRP	DUP	PP	TP	SS	DRP	DUP	PP	TP	SS
	----------------------- mg/l ------------------------				g/l	--------------------------------kg/ha -------------------------------				
Aerated	1.03	0.19	0.72	1.95	0.42	2.24	0.402	1.53	4.17	831
Control	0.54	0.12	0.68	1.33	0.29	1.20	0.26	1.51	2.97	636
SED	0.117	0.069	0.217	0.386	0.189	0.59	0.13	0.68	1.30	361
F-statistic	0.05	0.38	0.86	0.25	0.57	0.22	0.38	0.98	0.46	0.64

B) Excluding the last storm event

Treatment	DRP	DUP	PP	TP	SS	DRP	DUP	PP	TP	SS
	----------------------- mg/l ------------------------				g/l	--------------------------------kg/ha -------------------------------				
Aerated	0.94	0.16	1.18	2.28	0.79	0.92	0.12	0.97	2.01	556
Control	0.61	0.10	1.09	1.79	0.48	0.73	0.12	1.16	2.01	495
SED	0.082	0.068	0.360	0.376	0.407	0.669	0.08	0.087	1.484	349.7
F-statistic	0.054	0.46	0.84	0.33	0.54	0.80	0.94	0.84	0.99	0.88

Flow-weighted mean concentrations and annual loads of P fractions and SS showed no significant differences between treatments commensurate with soil physical quality six months after mechanical soil aeration. However, the AVOVA for flow-weighted mean DRP concentrations indicated a difference (P = 0.05; Table 1), with losses almost double for aerated soil compared to non-aerated soil. Mean Olsen P and WSP, which have elsewhere shown to be related to DRP in surface runoff (McDowell and Condron 2004), were similar between treatments, but generally greater in the second sampling (Table 6.2). Franklin *et al.* (2007) concluded that mechanical soil aeration of poorly draining soils, prone to episodes of wetness as a result of soil compaction, stimulates reducing and oxidising conditions and DRP loss in runoff. For this study, a more likely explanation is that soil disturbance via aeration exposed a greater specific surface area to runoff and thus P desorption, subject to kinetics (McDowell and Sharpley 2003b; Sharpley *et al.* 1981), which may have been enhanced by a prolonged period of saturation during the last storm event. Another possible explanation is that soil disturbance as a result of soil aeration may have liberated P from the microbial biomass which has been reported to be sensitive to disturbance (Sparling *et al.* 1987), although this would have likely been short-lived.

Table 6.1. Effect of treatment and year on mean Olsen P and WSP concentrations. Standard error of difference (SED) is given for the comparison of concentrations by treatment, year and the interaction of treatment and year. ** = *P*<0.01 and ns = not significant.

	Olsen P (mg/kg)		Water soluble P (mg/L)	
Treatment	2009	2010	2009	2010
Aerated	32	30	0.084	0.104
Control	29	37	0.086	0.125
	$SED._{05\ treatment}$ 2.76 ^{ns}		$SED._{05\ treatment}$ 0.0084 ^{ns}	
	$SED._{05\ year}$ 3.45 ^{ns}		$SED._{05\ year}$ 0.0111 **	
	$SED._{05\ interaction}$ 4.42 ^{ns}		$SED._{05\ interaction}$ 0.0139 ^{ns}	

There were a total of 9 grazing events during the trial (Figure 6.4). Concentrations of DRP in surface runoff were greatest following the fourth irrigation event on 13 November, 2009. Superphosphate fertiliser was applied (45.5 kg P/ha) on 12 November, 2009 and, excluding the large storm event in May, 2010, surface runoff generated the following day accounted for 27% of the total DRP load measured during the trial (Figure 6.4).

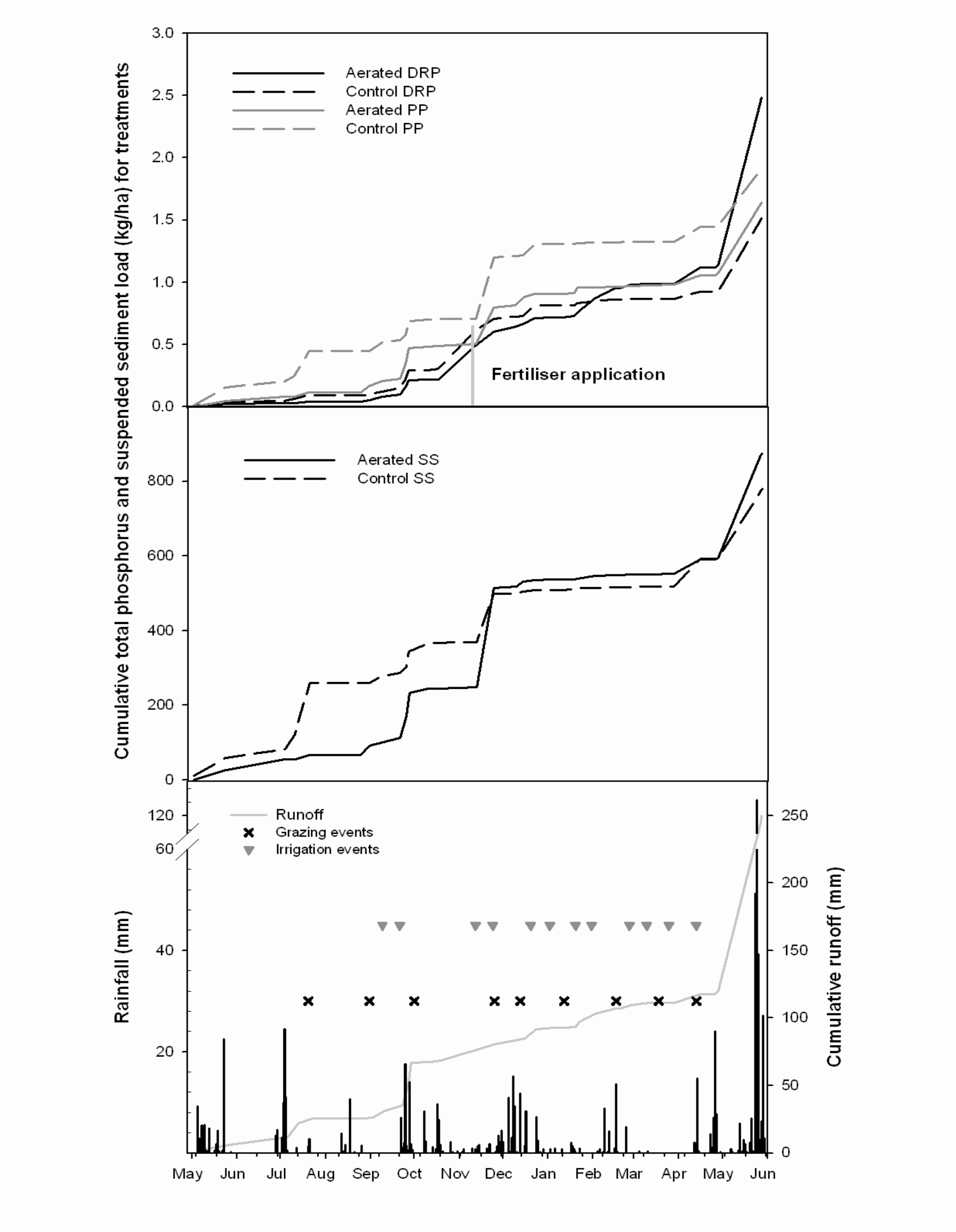

Figure 6.4. Cumulative total load of dissolved reactive P (DRP), dissolved unreactive P (DUP), particulate P (PP) and suspended sediment (SS) (top), based on each surface runoff producing event, and cumulative total runoff and rainfall from May, 2009-May, 2010 (below). Note that × and ▼ indicate grazing and irrigation events, respectively.

Again, excluding the last storm and fertiliser event in May 2010 and November 2009, respectively, there was a significant relationship between loads and concentrations of P fractions and SS with time (days) since grazing, based on all grazing events (Figure 6.5). The decreasing trend is likely due to the recovery of the soil from treading, a decrease in P availability from dung [(Curran Cournane *et al.* 2010a)- Chapter 4] and/or the uptake of soluble P from fertiliser with time (McDowell and Houlbrooke 2009). Excessive losses from grazing and fertiliser can be controlled by land managers by increasing the time interval between fertiliser application or grazing and irrigation or predicted rainfall (Nash *et al.* 2005) or by irrigating to a soil moisture content just below (*c.* <5%) FC (McDowell and Houlbrooke 2008).

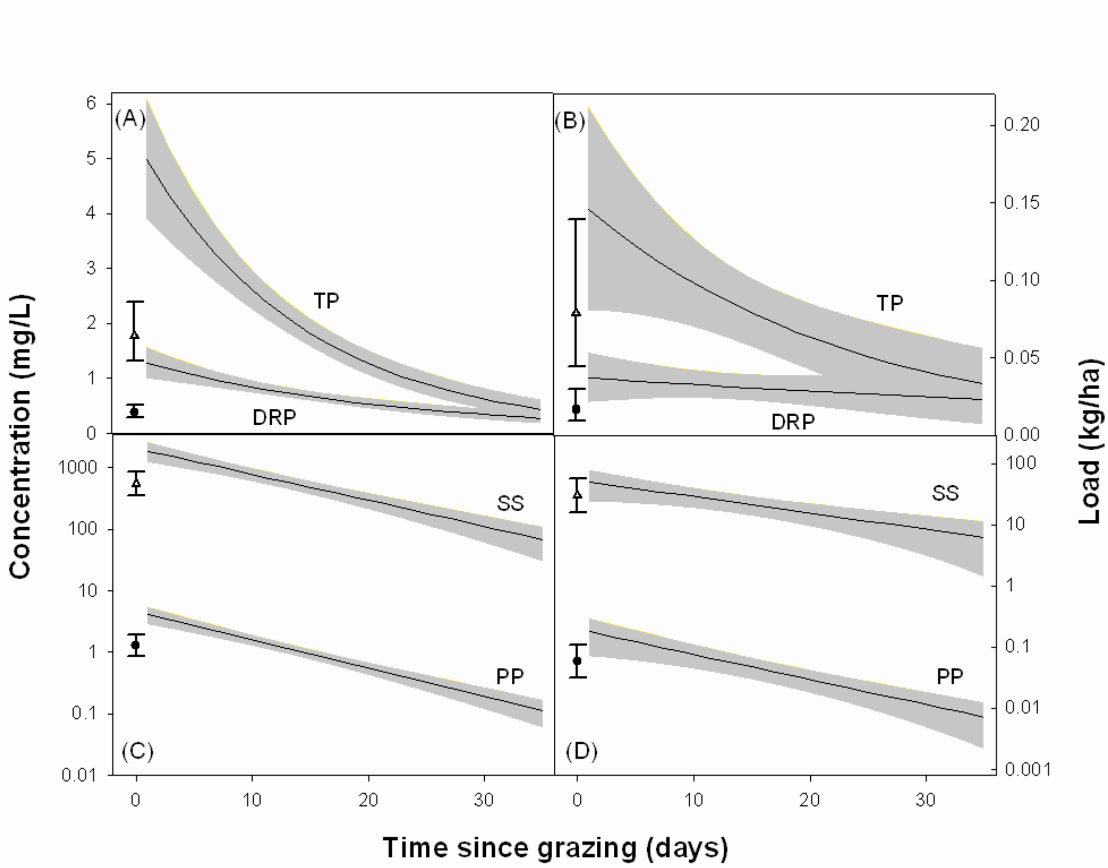

Figure 6.5. Regression relationships between concentrations and loads of phosphorus fractions (dissolved reactive P, DRP; particulate P, PP; total P, TP) and suspended sediment (SS) with time since grazing. The trend lines were derived from event for each of the ten runoff plots. Shading represents the 95 % confidence intervals. Background (pre-cattle grazing) means (and 95% confidence intervals) are given at time 0.

6.4 Conclusions

Significant differences in macroporosity and bulk density were found between aerated and control treatments. However, six months without cattle grazing and from seven months onwards with grazing, there were no significant differences in the above parameters and saturated hydraulic conductivity between aerated and control treatments. This lack of treatment difference was largely attributed to the re-settling of the poorly structured and dispersive soil studied. Although, flow weighted mean concentrations and annual loads of DRP on the mechanically aerated soil were approximately double those from the control treatment, no significant differences were observed between treatments for surface runoff volumes and losses of P and SS, which may reflect the similar soil physical conditions exhibited throughout most of the trial. As observed elsewhere, time (days) since grazing and fertiliser application were found to influence P and SS losses. We conclude that the proposed mitigation practice, soil aeration, thought to alleviate soil physical quality, did not decrease P and SS losses during the period of the trial. Any changes in soil physical properties like macroporosity that may have decreased losses were short-lived and therefore unlikely to influence surface runoff and P and SS losses in the long term for this poorly structured soil. These findings may apply at other sites where surface runoff is generated under saturated-excess conditions, but not for soils where infiltration-excess surface runoff is the dominant runoff process. However, further research, particularly in relation to hydrology and different soil types, is required to confirm this before aeration can be recommended to landowners as a means to mitigate P and SS losses.

Chapter 7- Conclusions and Recommendations

This research programme demonstrated that soil type, soil moisture, and land management practices have a major control and influence on P and SS losses in surface runoff from grazed pastures (Figure 7.1).

7.1 Influence of soil type

Differences in soil type (i.e. texture and structure) had a major influence on P and SS losses and soil physical quality (Chapter 3 and 4). For example, results described in Chapters 3 and 4 showed that losses of P and SS were greatest from the Pallic and Recent Gley soils compared to the Brown and Melanic soils. Both of the Pallic and Recent Gley soils exhibit poor structural stability and enhanced slaking and dispersion attributes compared with the well-drained and better structured Brown and Melanic soils (Hewitt 1998). Soil physical quality exhibited a similar pattern with soil compaction (measured by macroporosity, soil bulk density and K_{sat}) greater for the poorly structured Pallic and Recent Gley soils. The Pallic soil also generated greater concentrations and loads of P than the more resilient Brown and Melanic soils although all three soils had comparable Olsen P concentrations. This was also reflected in concentrations of WSP where concentrations were almost double for the Pallic soil than either the Brown and Melanic soils. These findings again confirm the susceptibility of the Pallic soil to losses which may be further attributed to having the least P retention value of all the four soil types investigated.

Results from the field trial carried out at Invermay showed that soil physical quality exhibited a direct relationship to loads and concentrations of P. For example, P loss in runoff decreased with increasing soil macroporosity. Hence, management of different soil types needs to be considered when grazing livestock to control P and SS losses in surface runoff and maintain soil physical quality. Strategies such as restricting grazing or stocking density on soils of poor structural stability are recommended. These findings also highlight the use of soil physical quality, in particular macroporosity, as a valid measurement with regard to P and SS loss.

7.2 Influence of soil moisture

Soil moisture regimes played a major role in P and SS losses (Chapter 3 and 5) and runoff processes (Chapter 5 and 6). Soils maintained at high soil moisture (90% available water holding capacity- AWHC) led to an increase in PP and SS in runoff, whereas soils at low soil moisture (< 10% AWHC) increased losses of DRP in runoff. The loss of PP and SS was attributed to deeper hoof penetration and soil disturbance with increased soil moisture, while enriched DRP losses was attributed to the death and decay of soil microbial biomass or the lack of P dilution upon drying.

Seasonal soil moisture dynamics influenced P losses in the field. Plots at Invermay (Chapter 5) showed that the greatest DRP and TP loads occurred in winter, while the greatest DRP and TP concentrations occurred in summer. The enrichment of P concentrations in summer can pose a significant algal response due to increased light and warmth (Carpenter *et al.* 1998). The generation of summer losses in surface runoff was attributed to infiltration-excess conditions, possibly due to hydrophobic conditions. In contrast, runoff in winter was attributed to saturation-excess conditions when soil moisture was above or equal to field capacity. While there is limited scope to manage for infiltration-excess surface runoff losses from pasture outside impervious zones, these findings reinforce the use of mitigation strategies such as restricted or nil grazing when soil moisture has reached field capacity.

No such seasonal soil moisture effect was observed for P losses at North Otago, (Chapter 6), as irrigation continuously hydrated soil throughout the growing/spring-summer season. However, infiltration- and saturation-excess conditions were almost of equal importance in North Otago (22 % and 26 % of total runoff volume, respectively), until the last storm event (generating 52% of total volume) which has a return period of about 1 in 40 years. The volume losses generated by infiltration-excess surface runoff were attributed to poor soil physical quality, which can to some extent be managed by re-scheduling irrigation after grazing. However saturation-excess runoff during the growing season from over-irrigation would need to be minimised to prevent losses associated with grazing (e.g. dung [see below], treading and pasture removal). This could be achieved by irrigating to 5% < field capacity, not field capacity.

7.3 Interaction of livestock grazing and P and SS losses in runoff

As well as soil type and soil moisture, livestock were shown to have a significant influence on P and SS losses in surface runoff (Chapter 4). Dung was shown to be a major factor influencing P and SS losses in surface runoff, while P and SS losses of soils that were only treaded upon were comparable to nil treading (control) losses when soil moisture was at 50% AWHC. However, as shown in Chapter 3 and 5, losses change with soil moisture.

Significant relationships were found between loads and concentrations of P and SS with changes in macroporosity, K_{sat} and time since grazing at Invermay (Chapter 5). Similar relationships were found between runoff losses with time since grazing at North Otago (Chapter 6). Cattle proved to have greater negative impacts on soil physical quality than sheep or deer, but no significant difference was found in P and SS losses in surface runoff between stock types at Invermay. The relationship between soil physical quality and the occurrence of runoff suggested that at Invermay it was not a case of *'what'* stock type grazed but *'when'* stock were grazed, and that winter grazing, when soil moisture has reached field capacity should be avoided, particularly for vulnerable soil types, to minimise P and SS losses, regardless of stock type.

7.4 Potential to decrease P and SS losses in runoff

Mechanical soil aeration failed to mitigate P and SS losses in surface runoff at North Otago (Chapter 6). Any soil physical improvements as measured by indicators like macroporosity were short lived. Concentrations and loads of DRP were approximately double from aerated than non-aerated soil which was attributed to soil disturbance allowing for more dissolution of dissolved P, which may have been further enhanced by a prolonged period of saturation during the last storm event. No significant differences for soil physical properties were evident for aerated and non-aerated soil by the time the first runoff event occurred six months after aeration, and were therefore unlikely to influence P and SS losses. Given that runoff was generated by saturation-, in addition to, infiltration-excess conditions, improving infiltration would be unlikely to decrease P and SS losses compared to a site that was dominated by infiltration-excess conditions.

Time since grazing and fertiliser application were both found to influence P and SS losses in runoff. This can be controlled by increasing the time interval between fertiliser application or grazing and irrigation or predicted rainfall or preventing runoff by irrigating to < 5% of field capacity not field capacity.

Overall, impacts of livestock grazing can have detrimental impacts on soil physical quality and losses of P and SS in surface runoff, but if well managed, the extent of contaminant losses can be decreased. My work reinforces best management practices (BMPs) to control P and SS losses in surface runoff such as:

- Restricting grazing or stocking density on vulnerable soil types (e.g. Pallic or Recent Gley soils);
- Avoid grazing stock when soil moisture is at or above field capacity, particularly for vulnerable soil types, by using a stand-off pad or area of the farm that does not produce runoff;
- Increasing the time interval between fertiliser application or grazing and irrigation or predicted rainfall, and irrigating to $< 5\%$ of field capacity, not field capacity; and
- Employing the use of soil physical properties, such as macroporosity, not only to assess soil quality, but also the potential for P and SS losses in surface runoff.

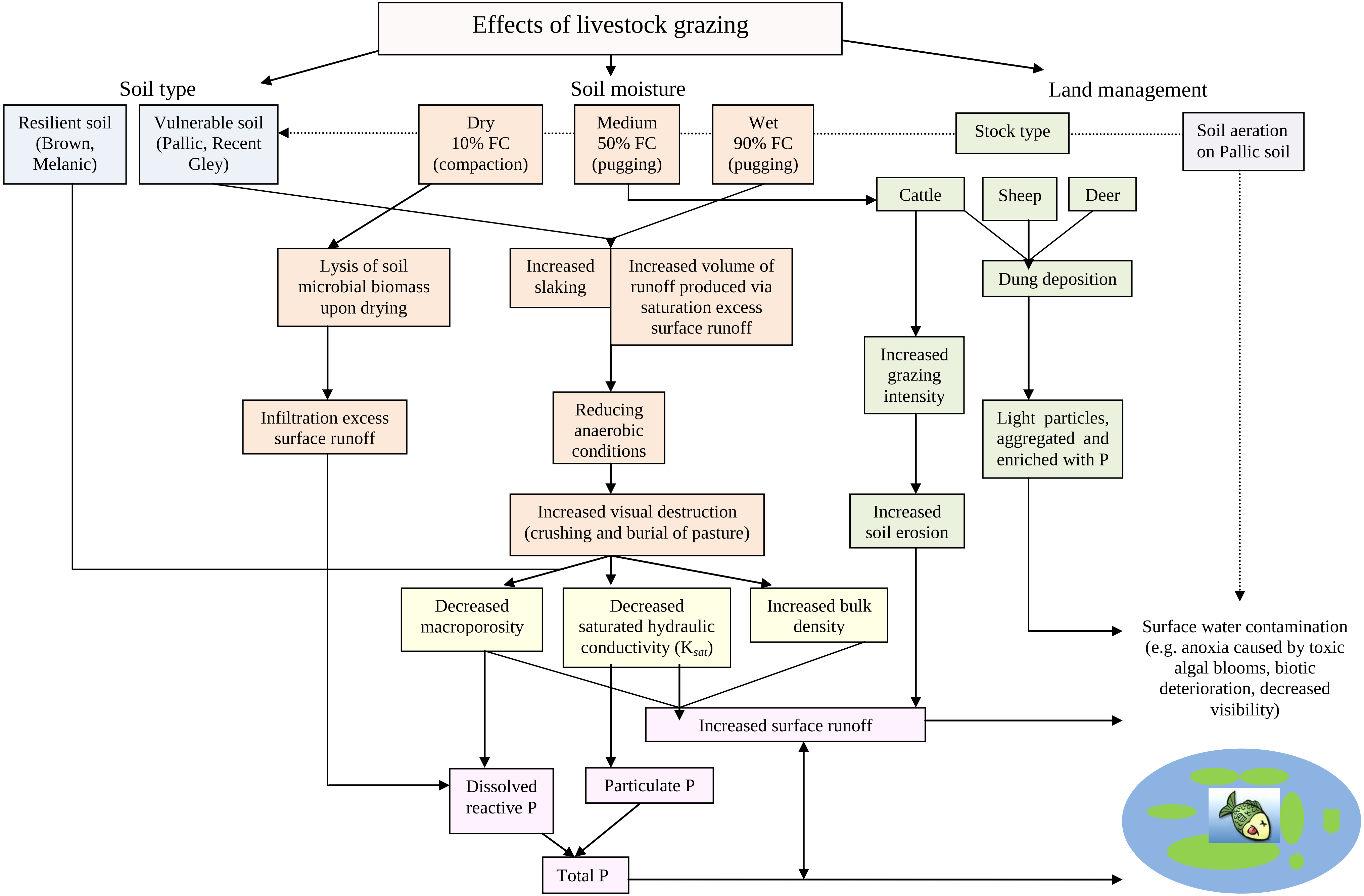

Figure 7.1. Conceptual diagram illustrating the key findings livestock grazing have on soil types, soil moisture and different land uses and fractions of phosphorus and suspended sediment lost in surface runoff.

7.5 Summary and recommendations for future work

Pastoral agriculture is the largest contributor to New Zealand’s export income. However, intensive livestock grazing is a major factor in accelerated transfer of P and SS from soil to waterways, and therefore investigation of P and SS loss in surface runoff is very important.

Herewith a summary of the key findings to emerge from this research project and recommendations for future studies in this area:

- The differences in surface runoff losses across contrasting soil types. These findings were consistent for soils from both simulated controlled experiments and field trials, particularly for Pallic soils. These soils are vulnerable to compaction and surface runoff losses when disturbed and landowners should be very cautious when grazing Pallic soils. Soil aeration, thought to alleviate soil compaction, failed to mitigate surface runoff losses and soil compaction for this soil type. The impact of other management practices such as restricted and/or nil grazing when certain soil moisture contents have been reached on P and SS losses need to be thoroughly evaluated for this soil type, which in turn will help landowners manage these poorly structured soils that occupy 12% of New Zealand’s landscape. It may be a case that cropping these soils is accompanied by a cut and carry system but this would have to be investigated. Future research should also explore the vulnerability of a wide range of soil types to livestock grazing especially given the ongoing expansion of high intensity dairying in New Zealand. This research will enable the development and evaluation of appropriate management practices to minimise P and SS loss in surface runoff.
- Soil moisture also played a key role influencing surface runoff losses and soil physical quality for soil types. As well as influencing the fraction of P loss (high soil moisture = increase in PP and SS loss; low soil moisture = increase in DRP loss), soil moisture also determined the generation of either saturation- and infiltration-excess surface runoff. The majority of surface runoff produced in both field trials was generated under saturation-excess conditions and this may have further implications on the generation of subsurface runoff for soils across Otago and Southland. Although P loss in subsurface pathways is thought to be minimal, it is not the case for all catchments, particularly for soils with large macropores and infiltration rates such as Melanic soils and other soils with high sand contents. The climate is changing and the frequency and severity of large storm events is on the rise. For poorly draining soils the likelihood of

increased storm events will cause exacerbated P and SS losses in surface runoff and further research is warranted to deal with the possibility of large exports.

- Losses of P from the contribution of dung were found to be greater than losses from either treading or nil treading amongst soil types at 50% soil moisture for a simulated controlled experiment. Similarly, in both field trials losses of P and SS decreased with time since grazing which was likely attributed to a decrease in the availability of P loss from dung as well as the recovery of soil physical properties with the removal of stock. For intensive dairy farms in the vicinity of freshwater systems, buffer strips (*c*. 10 m) can be a successful measure for trapping P and SS transported from land to stream. A large proportion of dung loss was particulate material (PP and SS) and buffer zones could therefore provide some efficiency in the trapping of such losses. However, the effectiveness of buffer strips needs to be more extensively investigated, especially in relation to the prevalence of light and heavy aggregates in different soil types and rainfall patterns. It may be a case that light aggregates, which are enriched in P compared to heavy aggregates, are not stopped by buffer strips.
- Key processes from both field scale *in-situ* plot studies included relationships between time since grazing and soil physical properties with P and SS losses in surface runoff. These relationships highlighted the importance of macroporosity as an environmental indicator for P and SS loss. Future research should also investigate losses at the catchment scale where other factors like impermeable surfaces (including land and tracks) may respond differently and contribute a large amount of P and SS despite only covering a small part of the catchment. It may be a case that these impermeable surfaces may benefit from strategies such as soil aeration, but again, this can vary between soil types and would have to be investigated. Catchment scale studies also have the potential to buffer or avoid influences associated with extremes in for example climate (e.g. wetting and drying cycles) which influence the robustness of surface runoff plots. Such long-term studies will aid in the identification of possible trends and processes that will broaden our knowledge and understanding of P and SS issues that deteriorate water quality.
- Lastly, it is pivotal to educate and communicate findings to fellow researchers, land owners, stakeholders and general public. By informing landowners of how their soil types, slopes, soil moistures, and stock types influence P and SS loss, they then know where and when on the farm the risk of such losses are greatest. Better decisions on

fertiliser, effluent, irrigation and grazing management can then be made with weather forecasting to assess the likely risk of contaminate loss. This will aid in the voluntary and regulatory protection of aquatic ecosystems and reduce or reverse the environmental damage posed by the livestock grazing industry as well as maintaining productivity.

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