Impacts of forage management on phosphorus cycling and sediment and phosphorus transport in surface runoff

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ABSTRACT

Sediment and phosphorus (P) in runoff from pastures are potential non-point source pollutants in surface waters. Sediment and P loads in runoff from pastures are partially influenced by surface cover, sward height, treading damage, surface slope, soil moisture, and soil P. Forage productivity and nutrient content are affected by seasonal conditions and management practices. Appropriate forage management practices will reduce P and sediment losses from pastures and provide adequate nutrition for grazing cattle, limiting the need for imported feed supplements. The objectives of the current study were to quantify amounts of sediment and total and soluble P in runoff produced during simulated rainfall from pastures, to determine which soil and forage characteristics are related to sediment and P loads in runoff, to determine the effectiveness of buffers in the control of sediment and P displaced from pastures, and to determine the amounts of aboveground plant biomass production and P uptake by forage under different forage management systems. During a 3-year period, a study was conducted at the Iowa State University Rhodes Research and Demonstration Farm to determine the impacts of forage management on P cycling and sediment and P loads in surface runoff from pastures. Pastures had slopes of 0 to 15° and the primary forage species in the pastures was smooth bromegrass (*Bromus inermis* Leyss). Three pastures of approximately 2.75 ha were subdivided into five 0.4-ha paddocks, with a 10-m wide vegetative buffer downslope. Sandbags were placed around the perimeter of the pastures and between each paddock to prevent contamination from surface runoff during natural rainfall events from outside the experimental area and between adjacent paddocks. Forage management treatments were randomly assigned to one of 5 paddocks in each pasture. Treatments included an ungrazed control (U), summer hay harvest with fall stockpiled grazing to a residual sward height of 5 cm (HS), continuous stocking to a residual sward height of 5 cm (SC), rotational stocking to a residual sward height of 5 cm (5R), and rotational stocking to a residual sward height of 10 cm (10R). Paddocks were initially stocked with 3 mature Angus cows in May of all three years. Rainfall simulators measured 1 by 0.5 m. Rainfall simulations were conducted four times per year at two slope ranges within the paddocks and at 6 locations within the vegetative buffers, at a rainfall rate of 7.1 cm·hr⁻¹ for 1.5 hours to determine amounts of surface runoff and sediment and P loss in surface runoff. During rainfall simulations, the amounts of rainfall and runoff were measured at 10-minute intervals and runoff samples were composited over the simulation period. Forage and soil measurements taken during rainfall simulations.
included surface roughness, ground cover, penetration resistance, antecedent soil moisture, Bray-1 P, sward height, and forage mass. Forage management did not affect sediment load (7.3 ± 5.0 kg·ha⁻¹·hr⁻¹). Total P load was greatest (P < 0.05) from 5C treatment (0.071 ± 0.011 kg·ha⁻¹·hr⁻¹), did not differ between U, HS, and 10R treatments (0.019 ± 0.011 kg·ha⁻¹·hr⁻¹) and was intermediate in 5R (0.053 ± 0.011 kg·ha⁻¹·hr⁻¹). Soluble P load was greatest (P < 0.05) from 5R and 5C (0.037 ± 0.004 kg·ha⁻¹·hr⁻¹) and did not differ between the U, HS, and 10R (0.011 ± 0.004 kg·ha⁻¹·hr⁻¹) forage management treatments. No single factor was a good predictor of sediment or P load in surface runoff. Sediment load (R² = 0.17) and total P load (R² = 0.13) were most closely related to the proportion of vegetative surface cover, of the soil and forage characteristics measured. Forage samples were clipped monthly from April through November of each year and forage mass and P content were determined. Forage management practice and sampling month had a significant interactive affect on forage P content. In general, forage P was greatest (P < 0.05) during the spring (0.21 ± 0.016%), decreased with forage maturity, and was lowest (P < 0.05) in the fall (0.13 ± 0.016%). Annual forage productivity was greater (P < 0.05) in the 4 harvested treatments (6744 ± 890 kg·ha⁻¹) than in the U treatment (1872 ± 890 kg·ha⁻¹). Annual P uptake by forage followed the same trend as forage production, being greater (P < 0.05) in harvested treatments (13.9 ± 2.1 kg·ha⁻¹) than in the U treatment (3.7 ± 2.1 kg·ha⁻¹). Surface runoff from pastures managed to maintain adequate residual forage cover will not contribute greater sediment or P to surface waters than does an ungrazed grassland. Forage production and P cycling were increased by forage harvest by either grazing or hay harvest.
CHAPTER 1. GENERAL INTRODUCTION

THESIS ORGANIZATION

This thesis is organized as an introduction to the research and related literature review followed by a brief description of the hypothesis for developing this research and its objectives. Manuscripts for submission to Rangeland Ecology and Management follow the literature review and introduction of research. Following the manuscripts are a general conclusion section, appendices of additional information, and acknowledgements.

INTRODUCTION

The proportion of phosphorus (P) contributed to fresh waters from agricultural land, as a non-point source pollutant, is increasing as contributions from point sources (industry) are being reduced by easier identification and control mechanisms (Sharpley et al., 1994). Phosphorus is of concern because in freshwater lakes and streams, it is often the nutrient limiting the growth of aquatic plants and bacteria (USDA, 1999b). This over-enrichment of receiving waters with mineral nutrients resulting in the excessive production of plants and bacteria is referred to as eutrophication (Correll, 1998). Phosphorus concentration in water greater than 0.1 ppm can have negative impacts on surface waters (Sharpley et al., 2000) restricting their use for aesthetics, fisheries, recreation, industry, and drinking, and thus has serious local and regional economic impacts (Sharpley et al., 1994).

A large body of evidence suggests that P from agricultural sources represents a significant input to fresh water and the increase in P concentrations in agricultural drainage water over time reflects the accumulation of P in soils (Sharpley et al., 2000). Much of the increased amounts of P in the soil are associated with confinement livestock feeding operations and the growing disintegration between crop and livestock production (Slaton et al., 2004; Cahoon and Ensign, 2004). A number of reports in the U.S. (CAST, 2002b) and New Zealand (Gillingham and Thorrold, 2000) have also implicated grazing of livestock in the increase in P loss from pastures to surface waters. These losses are related to soil disturbance, plant damage, and the deposition of dung to the soil surface.

Phosphorus loss to surface waters can be controlled with proper management practices. The use of management tools, such as P indices, allows field staff, watershed planners, and land users to assess the various landforms and management practices that contribute to the potential risk of P movement to waters (Lemunyon
and Gilbert, 1993). Management of grazing lands to enhance the vegetative cover provided by forage crops not only provides feed for grazing livestock, but also acts to hold soil in place, filter water, and recycle nutrients (CAST, 2002b; Hubbard et al., 2004), resulting in improved water quality.
CHAPTER 2. REVIEW OF LITERATURE

WATER QUALITY

Impact of Livestock on Water Quality

Livestock production has been associated with negative impacts on surface water quality. Water quality concerns related to livestock production include elevated N and P, excess sediment, the presence of pathogenic organisms, and high biological oxygen demand (Hubbard et al., 2004). Nitrogen excreted in the feces and urine of cattle can potentially volatilize as ammonia, causing odor problems and lead to contamination of surface and ground waters as it returns to the land or water with precipitation (CAST, 2002a). Alternatively, livestock manure applied to the land in quantities greater than the ability of plants to take up N can result in leaching of NO₃-N into surface and ground waters (Smith and Frost, 2000; CAST, 2002a) resulting in a reduction in water quality.

The primary forms of P delivered from an agricultural field to surface or groundwater are P adsorbed to eroding sediment, soluble P in runoff water, soluble P in leaching water, and P losses related to the type of waste P applied (Havlin, 2004). Correll et al. (1999) reported annual losses of P from a watershed dominated by pasture to range from 0.071 to 0.44 kg P·ha⁻¹. These losses were similar to the annual losses of P from forest land (0.013 to 1.18 kg P·ha⁻¹) and lower than annual P losses from a watershed dominated by row crop production (0.14 to 13.3 kg P·ha⁻¹). The largest concerns for managing P from an environmental perspective are the concentration of P in feedlots and the spreading of the P in manure on croplands in amounts exceeding crop requirements (CAST, 2002a). Hubbard et al. (2004) reported that watersheds with concentrated livestock populations have been shown to discharge 5 to 10 times more nutrients than watersheds in cropland or forest. These excess nutrients, P in particular, may lead to eutrophication of surface waters (USDA, 1999).

Sediment from agricultural lands acts as a vector in the transport of nutrients to surface waters (Daniel et al., 1994). In addition to their role in the transport of nutrients, sediment particles suspended in the water can block the transmission of light, increase water temperature, and interfere with the respiration of aquatic animals. Sediment that has settled out of the water can clog navigation channels and cover food sources of aquatic animals. All these factors act to reduce the usability of water bodies for economic, aesthetic, recreational, and...
wildlife uses. Sediment loss from agricultural land also has negative impacts of the productivity of that land (Uri, 2000).

Contamination of surface waters by pathogenic organisms, such as *Listeria* spp., *Escherichia coli*, and *Salmonella* spp., in the manure of grazing animals can negatively impact both human and animal health (Pell, 1997). Mean fecal coliform counts were shown to be greater in streams located in continuously stocked pastures than in streams located in rotationally stocked pastures (Sovell *et al.*, 2000). When manure is spread on grass pastures, the concentrations of most pathogens begins to decline within 2 to 3 days, however some pathogens may still be detectable at elevated levels even after 128 days (Hutchison *et al.*, 2005).

The decreasing integration between row-crop and animal production presents a potential problem for agronomically and environmentally sound nutrient management (Slaton *et al.*, 2004). In North Carolina, the relationship between land use practices and soil P was compared over the 20-year period from 1980 to 2000. The study found that the amount of animal production, particularly swine, increased significantly during the study period, resulting in an excess amount of manure for land application. This excess manure did not displace the use of commercial P fertilizer, which continued to rise during this period (Cahoon and Ensign, 2004). This over-application has resulted in 69% of soil samples tested in North Carolina in 2001 being high or very high in available P (Fixen, 2002). Mean soil test P levels in agricultural watersheds have been shown to be highly correlated with total P concentrations of surface waters, $r^2$ ranging from 0.88 to 0.96 depending on soil test method used, in those watersheds (Klatt *et al.*, 2003).

Once P reaches a lake, it is retained and recycled within the aquatic system (Correll, 1998). The easiest way to control excretion of P from livestock is to control intake of P (Tamminga, 1996). Based on these two statements, it is apparent that management practices on the farm can have long-term impacts on surface water quality and P losses from animal production should be controlled so that animals excrete less P into the environment and P-rich sediment particles are prevented from reaching surface waters by proper management of pastures.

It has been claimed that livestock utilize P inefficiently; this claim was supported by the fact that 60 to 80% of P consumed by livestock is excreted in the manure (Knowlton *et al.*, 2004). Phosphorus excretion can be reduced by balancing rations to meet requirements at different parts of the production cycle and by
improving animal performance so that nutrient excretion associated with maintenance can be reduced per unit of animal product produced (Knowlton et al., 2004). Over-feeding of P is a common practice in the United States beef and dairy industries (Knowlton et al., 2004). Dairy producers in the U.S. typically formulate rations to exceed P requirements by 20 to 25%. This over-feeding not only has an environmental cost, but also an economic cost of $10 to $15 per cow per year in the purchase of supplemental P (CAST, 2002a).

Several strategies exist to reduce P excretion from livestock feeding operations. Nutritional and management strategies related to the environmental impacts of livestock and the reduction of these impacts have been reviewed by Tamminga (1996), Van Horn et al. (1996), Jongbloed and Lenis (1998), CAST (2002a), CAST (2002b), and Knowlton et al. (2004). Based on simulations using the Dairy Forage System Model, two dairy farming systems were evaluated and it was determined that P balance in the simulated farms could be reached by feeding P to meet NRC recommendations, using a cropping strategy in which the farmland base provided all forage needs of the farm, feeding animals a high forage diet, and producing replacement heifers on the farm (Rotz et al., 2002). Purchased feeds are the primary source of nutrients imported onto a farm (Rotz et al., 2002). Decreasing the amount of purchased feeds will decrease the excess nutrient balance on the farm, decreasing the potential of leaching and runoff of P to surface waters (Wang et al., 1999).

Decreasing dietary P from 0.49 to 0.4% resulted in a 23% reduction in fecal P excretion without impacting milk production or animal condition (Wu et al., 2000). Erickson et al. (1999) fed beef steers diets containing 0.14% P, a level less than the current NRC requirements, without a negative impact on animal performance. Between 1973 and 1996 in the Netherlands, P excretion from pigs was reduced by nearly 58% by using improved dietary management strategies without impairing animal health while improving the feed conversion ratio (Jongbloed and Lenis, 1998).

**Eutrophication of Surface Waters**

Reviews of the role of P in the eutrophication of surface waters have been presented by Correll (1998) and Correll (1999). The trophic state of a body of water refers to the overall level of nutrients and related algae and plant growth within the system and the relationship of this primary productivity to aquatic animal growth. As nutrient levels in the water increase, the trophic state of the water body increases; in order from oligotrophic, mesotrophic, eutrophic, or hypereutrophic (USDA, 1999). Accelerated eutrophication of surface waters is
associated with inputs from surface runoff as opposed to subsurface flow (Sharpley et al., 1993). In fresh water environments P is generally considered to be the primary nutrient limiting eutrophication, while N is the nutrient generally considered to limit growth in marine environments (Correll, 1998; USDA, 1999). Dodds et al. (2002) challenged this belief based on an analysis of the nutrient content and tropic state of streams reported in the literature from the US, Canada, Europe, New Zealand, and Australia. They determined that both N and P might play a role in limiting algae growth in streams. Nitrogen, however, only becomes important in the control of eutrophication when the level of P is high (Downing and McCauley, 1992). Little is known about the role of P in relation to hypoxia in the coastal waters of the Gulf of Mexico (CAST, 1999).

In an oligotrophic system with a low level of primary productivity, most P will be stored in the bottom sediment and, as such, is not available for plant or algae growth. In a eutrophic or hypereutrophic system with an excessive level of primary production, much of the P in the sediment will be released to diffuse into the water column as orthophosphate (PO$_4^{3-}$). Orthophosphate is available for supporting plant and algae growth and is the only form of P that can be used by autotrophic organisms (Correll, 1998). If an elevated trophic state is caused by human interaction, such as an agricultural practice, it is referred to as cultural eutrophication (USDA, 1999). Eutrophication can lead to hypoxia or anoxia in poorly mixed bottom waters and at night during calm warm conditions (Correll, 1998), leading to the death of aquatic animals.

No set value of P concentration in surface waters exist that will lead to eutrophication in all situations. The P concentration leading to eutrophication will vary with the amounts of other nutrients and environmental conditions present (Correll, 1998). Eutrophic conditions in lakes are common when total phosphorus levels range from 30 to 100 ppb. At P concentrations in lakes greater than 100 ppb, a hypereutrophic condition exists. Streams may be considered eutrophic at total P concentrations exceeding 20 ppb (USDA, 1999).

The principal impacts of eutrophication relate to increased aquatic plant growth, oxygen depletion from the decay of plant material, pH variability, and plant species quality and food chain effects. These changes in water quality brought on by eutrophication restrict the use of surface waters for aesthetics, fisheries, recreation, industry, and drinking, and thus have serious local and regional economic impacts (Sharpley et al., 1994).
Grazing and Water Quality

Grazing lands are the single largest land type in the United States (CAST, 2002b). In 2002, the Council for Agricultural Science and Technology published a report detailing the environmental impacts of livestock grazing in the United States including the impacts of grazing on soil erosion, nutrient distribution, wetland communities, and hydrological cycles (CAST, 2002b). Most environmental concerns associated with the grazing of livestock occur at high animal densities resulting in overgrazing and trodden forages. When animal density is low and a good forage stand is maintained, there are few environmental concerns associated with grazing, but problems may still develop if animals have direct access to water bodies (Hubbard et al., 2004). A computer simulation indicated that highly intensive stocking of cattle on continuously grazed pastures in northeast Texas may be partly responsible for excessive N and P loads to lake water and that appropriate stocking density management could reduce nutrient losses from dairy pastures (Osei et al., 2003).

Sustainable management of grazing lands requires managing vegetative cover, not only to provide feed for grazing livestock but also to hold soil in place, filter water, and recycle nutrients (CAST, 2002b). Management of an ecosystem for a single factor, for example vegetative height, cannot address complex issues such as sediment particle detachment, movement, and filtration (Pearce et al., 1998b). Optimum pasture cover and height and minimizing grazing pressure, especially adjacent to drainage lines, are known to reduce the movement of nutrients into streams (Nelson et al., 1996). Several environmental benefits are associated with forage and grazing systems. These benefits include the action of forage roots to penetrate the soil, which helps to fragment compacted layers to enhance water and air infiltration into the soil and improve production of future crops. Pasture improves soil tilth by providing site habitat for earthworms, insects, and microorganisms. Forage crops decrease erosion compared to annual crops. Perennial grasses form dense root systems that serve as filters to remove contaminants before they reach groundwater. Organic compounds in animal feces build soil organic matter and increase water-holding capacity of the soil (Hubbard et al., 2004).

Grazing of sheep has been shown to increase the likelihood of high concentrations of nutrients in surface runoff compared to leaving an ungrazed area (McColl and Gibson, 1979). The concentration of particulate matter in runoff has been shown to be greater from plots that have received manure application than from plots not having received manure. This increase in particulate matter was likely related to manure having
a lower density than sediment, making it more easily transported (McDowell and Sharpley, 2002). The high nutrient content in manure and its ease of transport can contribute a large amount of nutrients to surface runoff. Schepers et al. (1982) showed that the amount of nutrients lost from grazing lands increased as the stocking rate increased. In a companion study, Schepers and Francis (1982) reported that an ungrazed area adjacent to the cattle pasture produced runoff of lower water quality than did the grazed area. This response was attributed to wildlife activity and the leaching of nutrients from the forage in the ungrazed area.

As stated previously, P in runoff is either bound to sediment or dissolved in solution. Phosphorus, able to pass through a 0.45-micron filter, is considered to be soluble P. Soluble P is immediately available for algal growth while sediment-bound P is not immediately available, but can become available with time (Sharpley et al., 1992; Sharpel et al., 1993). Forms of soluble P released from plants include dust on the surface of the plant, phosphate ions on the plant surface exuded by the plant, and slow release of phosphate ions from cellular components of the plant (Dougherty et al., 2004). Correll et al. (1999) reported that the majority of P from grazing lands was in forms highly available for algal growth, either as soluble or organic P, in contrast to P from land in row crop production that is usually bound to sediment particles. Uusitalo et al. (2000) reported that of the total P load in runoff from wheat and barley fields, an average of 74% (range 39 to 92%) was bound to sediment particles. Soluble P is also able to adsorb to sediment particles in water to become particulate P. Losses of dissolved P in runoff from pasture can exceed losses of P from tilled fields because the low amount of sediment in the runoff from pastures is not able to adsorb P leached from plant material and animal waste. However, in a cultivated field, soluble P in runoff is more likely to become adsorbed by soil particles and settle out of runoff with sediment. As P moves to a lake by stream flow, there is a general reduction in P load because of dilution and sediment deposition. However, the P that remains is more available to support algal growth (Daniel et al., 1994; Sharpel et al., 1994) than P that is deposited with sediment. Some studies have reported that 46% of the total P in surface runoff from pastures in the soluble form (McDowell and Wilcock, 2004), while others have reported values as high as 80 to 100% of the total P may be in the soluble form (Mathews et al., 1994).

Even considering the benefits associated with grazing, a number of studies in New Zealand have shown that grazing leads to an increase in P loss from pastures related to soil disturbance, plant damage, and the
deposition of dung to the soil surface (Gillingham and Thorrold, 2000). Nutrient and sediment concentrations measured in lake water from a watershed in which 71% of the land was in pasture, primarily for dairy production, were 0.27, 0.67, and 112 mg·L⁻¹ for soluble P, total P, and sediment, respectively (Osei et al., 2003). Annual P losses in surface runoff from pastures grazed by sheep and cattle in New Zealand range from 0.11 to 1.67 kg P·ha⁻¹·yr⁻¹ with a national average of 1.3 kg P·ha⁻¹·yr⁻¹ (Gillingham and Thorrold, 2000). Grazing management practices such as rotational grazing, the use of portable water and shade structures, and fencing to limit animal access to water bodies have been used to alleviate nutrient, sediment, and pathogen contamination of surface waters (Hubbard et al., 2004). Seasonal variation in P movement also exists and may be related to the leaching of P from dead plant material and other residues by rainfall (Sharpley et al., 1993).

The amount of P lost in surface runoff from pastures is relatively small compared to the total P cycling through the grazing system. Lambert et al. (1985) found annual P losses in runoff from a sheep pasture receiving annual P fertilization to be about 1.1 kg·ha⁻¹. They determined that this amount of P represented about 3.1% of P cycling annually through plant pool and 2.9% of P inputs to the system.

PHOSPHORUS IN CATTLE NUTRITION

Functions of Phosphorus in the Body

The function, requirements, regulation, and excretion of P in relation to cattle production has been reviewed by a number of authors (Littledike and Goff, 1987; Kerridge et al., 1990; Wadsworth et al., 1990; Ternouth, 1990; Winks, 1990; Karn, 2001; NRC, 2001). Cattle weighing 300 kg will contain approximately 2 kg P in their body (Ternouth, 1990). The largest percentage of this P is contained in the bones; approximately 80 to 85% of P in the body of cattle is found as hydroxyapatite in bone tissue. In addition to bone mineralization and growth, P is required for energy metabolism and transfer (ATP, ADP, AMP, and creatine phosphate), the transfer of genetic information (DNA, RNA), cell membrane structure in phospholipids, as a buffer system in the blood and rumen to regulate pH, and milk secretion (Karn, 2001; Knowlton et al., 2004).

Phosphorus Requirements in Beef Cattle

Phosphorus requirements for beef cattle are determined using a factorial method by summing the individual P requirements for maintenance, growth, pregnancy, and lactation and correcting for the percentage of dietary P absorbed by the animal. Maintenance requirements for all classes of beef cattle are set at 16 mg P·kg⁻¹ body
weight with an additional 3.9 g P per 100 g protein gain required for growth, 7.6 g P·kg\(^{-1}\) fetal weight required in the last 3 months of gestation and 0.95 g P·kg\(^{-1}\) milk production required during lactation. A true absorption of 68% is assumed to calculate absorbed P from dietary P for all feeds (NRC, 1996). Feeding P in excess of requirements is common in both beef and dairy cattle production; because of the fear that P deficiency will lead to reduced reproductive performance (CAST, 2002a).

**Regulation of Phosphorus in the Body**

Phosphorus homeostasis in the body is regulated by the interaction of parathyroid hormone, calcitonin, 1, 25 dihydroxy-cholecalciferol (a.k.a. vitamin D). Parathyroid hormone (PTH) increases P and calcium (Ca) absorption from the bone, decreases renal excretion of Ca, increases excretion of P, and stimulates the activation of 1,25 di-hydroxy-cholecalciferol (vitamin D), resulting in an increase in plasma Ca and P and has no direct effect on salivary P. Calcitonin (CT) is secreted in response to high levels of blood Ca, and facilitates the transfer of Ca and P to bone from plasma. The overall effect of CT is to depress absorption of P and Ca from the gastrointestinal tract and stimulate secretion of P while reducing Ca secretion in the saliva. 1, 25 Di-hydroxy-cholecalciferol (1,25 di-hydroxy vitamin D) is secreted by the kidney in response to PTH secretion or low concentrations of P in the plasma. 1, 25 Di-hydroxy-cholecalciferol increases absorption of both Ca and P across the gastrointestinal wall and, with PTH encourages the resorption of Ca and P from bone (Littledike and Goff, 1987; Ternouth, 1990).

**Indicators of Phosphorus Status in the Body and Signs of Deficiency and Toxicity**

The NRC (1996) for beef cattle makes no mention of P toxicity in cattle. The NRC (2001) for dairy cattle states that the long-term consumption of a high P diet may interfere with calcium (Ca) metabolism and reduce the apparent absorption of magnesium (Mg). A wide range of Ca:P ratios have been shown not to have a negative impact on beef cattle performance (Scott and McLean, 1981; NRC, 1996). A maximum tolerable limit of 1.0% P in the diet, assuming Ca is adequate, is presented for dairy cattle (NRC, 2001).

Phosphorus deficiency, arophorosis, has been characterized by the animals desire to eat bone, sticks, or rock. During P deficiency, feed intake and growth may be reduced, fertility my be impaired as exhibited by abnormal estrous cycles, milk production may be decreased, stiffness develops in the front limbs causing a plodding style of movement, a rough coat that may turn to a red-brown color in black cattle, and bones may
often become weak and may be easily broken (Blair-West et al., 1992; Karn, 2001). Prolonged deficiency resulted in an increase in plasma Ca, vitamin D and PTH (Blair-West et al., 1992). In steers fed a P deficient diet, feed intake was reduced by 14% after 7 weeks compared to animals receiving a diet in which P was adequate (Bortolussi et al., 1996). The reduction in growth is not necessarily related to the reduction in feed intake, but likely related to the role of P in energy metabolism (Blair-West et al., 1992). Phosphorus (or Ca or vitamin D) deficiency in young growing animals may result in rickets because of failure of the bone to mineralize at the growth plate and in osteoid matrix of the bone. In older animals, P (or Ca or vitamin D) deficiency may result in osteomalacia brought on by the failure of the osteoid matrix to mineralize during bone remodeling (Littledike and Goff, 1987). Phosphorus deficiencies in cattle fed typical feedlot diets are not common because of the large amount of P found in most cereal grains (Erickson et al., 2002). A P deficiency in grazing cattle is generally associated with P deficient soils (Karn, 2001).

Valk and Sebek (1999) reported that Holstein cattle consuming a diet containing 2.4 g P·kg⁻¹ DM, 67% of the current requirement in the Netherlands, experienced reductions in feed intake and milk yield, and exhibited signs of P deficiency severe enough to require them to be removed from a research trial during the second lactation. Dairy cattle producing 11,050 kg milk during a 308-day lactation and fed a ration containing 0.31% P exhibited a greater occurrence of foot rot than did cattle producing similar levels of milk and fed a ration containing 0.40 or 0.49% P (Wu et al., 2000). No differences were reported in animal performance or health status in beef cattle fed rations containing 0.14 to 0.34% P (Erickson et al., 1999) or 0.16 to 0.40% P (Erickson et al., 2002). The low P diets in both of these studies were lower than the P requirements established in the beef cattle NRC (NRC, 1996), implying that the P requirements for beef cattle may be lower than the current requirements.

In Angus heifers fed either a P deficient (0.12% P) or P adequate (0.20% P) diet from breeding until 3 weeks post-partum, neither group exhibited physical signs of lameness or stiffness, but the P deficient group had reduced body weight gains compared to the group receiving adequate P. At the completion of the study bone ash was lower in the P deficient group (67.2 %) than in the P adequate group (68.0 %) while Ca, P, and Mg, as percentages of bone ash, were not different between treatment groups. Cortical bone index, medial-
lateral wall thickness, breaking load, and breaking strength of the 3rd metacarpal were all greater in the adequate P group than in the P deficient group (Williams et al., 1991).

When cattle were fed a diet adequate in N, a P deficiency reduced liveweight gain of the animals. If cattle were fed a ration deficient in both N and P, no additive effect of the P deficiency over steers just deficient in N was observed. In the animals receiving adequate N, but inadequate P, the signs of P deficiency including reduced feed intake and reduced plasma inorganic P were more severe than in animals deficient in both nutrients (Bortolussi et al., 1996).

Bone, blood, rumen fluid, forage, feces, saliva, hair, and urinary P have all been suggested as indicators of P status (Kam, 2001). Fecal P concentration was observed to increase linearly as dietary P concentration increased from 0.31 to 0.47% of DM intake in Holstein cows during lactation (Wu et al., 2001). Salivary P concentration varied between 4.3 and 8.6 mmol·L⁻¹ during lactation and 8.2 and 12.1 mmol·L⁻¹ during the dry period in dairy cows fed P at 67 or 100% of Dutch P recommendations (Valk et al., 2002).

When dietary P was adequate, blood plasma and serum P were maintained between 1.5 and 2.5 mmol·L⁻¹ (Valk et al., 2000). Inorganic P concentrations in the plasma of beef cows have been shown to be much lower during lactation than during pregnancy for cows on diets providing similar amounts of P (Ternouth and Coates, 1997).

Urinary P concentrations ranged from 12.9 to 23.4 mg·L⁻¹ during lactation and 12.1 to 23.6 mg·L⁻¹ during the dry period in dairy cows fed P at levels from 67 to 100% of Dutch P recommendations (Valk et al., 2002). In dairy cattle fed diets containing 0.67% P, urinary excretion of P was 6.04 g·d⁻¹ during the first week of lactation while cattle fed a diet containing 0.34% P excreted 0.27 g·P·d⁻¹ (Knowlton and Herbein, 2002). In Holstein cows fed either 0.31, 0.39, or 0.47% P, on a DM basis, no difference was found in the shear stress required to break bones although the concentration of bone ash was lower in the 0.31% P group than in cattle fed higher concentrations of P (Wu et al., 2001). The concentration of P in rib bone is considered to be the best indicator of P status, but is difficult to determine under production conditions (Kam, 2001).

**Phosphorus Availability, Absorption, and Excretion in Cattle**

Much of the P in plants is bound in the organic form, phytate, and is unavailable to monogastric animals in this form. Because of the poor availability of phytate in monogastric animals, the enzyme, phytase, is commonly added to swine diets to increase the availability of organic P in plant material or low phytic acid corn varieties.
are fed that allow the P levels of diets to be decreased (Knowlton et al., 2004). In contrast to monogastric animals, ruminants are able to utilize the P in phytate, because the microbial organisms in the rumen can hydrolyze phytate. Morse et al. (1992) utilized an in situ digestibility technique to determine the availability of phytate P in several concentrate feeds commonly fed to dairy cattle, including cotton seed meal, dried distillers grains, hominy, rice bran, SBM, and wheat middlings. In all feeds except cottonseed meal, 99% of phytate P was available to cattle within 12 hours and 99% of cottonseed meal phytate P was available in 24 hours. Based on these findings, it was concluded that essentially all phytate P in concentrates should be considered to be available for absorption and can be used to meet dietary requirements of dairy cattle.

The high extent of P released from forages observed by Emanuele and Staples (1990) indicated that P probably was not associated tightly with the plant cell wall, but was in a soluble form in the cell. Smil (2000) stated that P is not associated with the cellulose, hemicellulose, or lignin fractions of plants. The availability of this forage P to ruminants was evaluated for alfalfa (Medicago sativa L.), rhizoma peanut (Arachis glabrata Benth.), dwarf elephantgrass (Pennisetum purpureum Schum.), bahiagrass (Paspalum notatum Flugge), bermudagrass (Cynodon dactylon (L) Pers.), and limpograss (Hemarthria altissima (Poir.) Stapf and Hubbard), using a mobile bag technique. The rumen was the major site of P release from these forage species. Phosphorus releases from forages in the rumen ranged from 71 to 94% of the total P in the forage depending on forage species (Emanuele et al., 1991) with 46% to 84% of the total P immediately available and the maximum extent of P release occurring for most forage species within 2 to 6 hours of entering the rumen (Emanuele and Staples, 1990). The average P released from the legumes evaluated was 95% of the total P while grasses had an average P release of 82% throughout the entire digestive system. It was also observed that the forage species with the lowest P content, bermudagrass, bahiagrass, and limpograss, also had the lowest P release coefficient (Emanuele et al., 1991). Ledoux and Martz (1991) reported that in an in situ digestibility trial, greater than 60% of the P in alfalfa hay (Medicago sativa L.), fescue hay (Festuca arundinacea Schrep.), bromegrass hay (Bromus inermis Leyss), and corn silage (Zea mays) was solubilized by washing in distilled water. Maximum P solubility of the four forage species was reached within 24 hours of rumen incubation, averaging 71.4% across forage species. The authors stated that P solubility might have been underestimated as a result of contamination by rumen microbes and rumen fluid (Ledoux and Martz, 1991). Martz et al. (1990) reported true P absorptions
of 64.4 and 74.6% from alfalfa and alfalfa-corn silage diets in lactating dairy cattle. Martz et al. (1999) reported the true absorption of P from corn silage diets providing 20.1 and 11.7 g P·d⁻¹ to be 83.3% and 96.0% in nonlactating, pregnant dairy cows. Based on the results of these studies, the assumption that the true absorption of P is the same across all feeds is not accurate and the assumed value of 68% is probably too low for most common feeds. Bormann (2004) demonstrated that apparent digestibility of P decreases with forage maturity when smooth bromegrass hay of different maturities was fed to beef steers.

Phosphorus is primarily absorbed from the duodenum of the small intestine (Valk et al., 2000) and can be effectively recycled to the rumen in ruminants. In a study by Khorasani et al. (1997), dietary P intake ranged from 82 to 105 g·d⁻¹ for dairy cattle receiving a diet with a 50:50 forage:concentrate ratio, while P flow to the small intestine ranged from 136 to 151 g·d⁻¹ in these animals. The P reaching the small intestine in excess of intake is a result of recycling through the salivary glands; which are the main route for excretion of excess absorbed P by ruminants (Valk et al., 2002). The important role of saliva in maintaining P homoeostasis in ruminants was demonstrated by increased urinary P and decreased fecal excretion of P when the parotid salivary ducts were ligated in beef cattle (Scott and McLean, 1981). Ruminants fed diets high in forage exhibit increased salivary P excretion compared to animals fed a high concentrate diet (Valk et al., 2000).

Phosphorus is normally excreted from the ruminant animal in the feces, however at high dietary P intakes, urine can become an important route for excretion of P (Knowlton and Herbein, 2002). Urinary P excretion in grazing steers averaged less then 1 g·d⁻¹ in steers grazing pasture with a P content of 0.32%, while these steers excreted 10 to 23 g P·d⁻¹ in the feces (Betteridge and Andrews, 1986). Dairy cattle fed a ration containing 0.67% P, on a dry matter basis, excreted over 6 g of P per day in the urine (Knowlton and Herbein, 2002). Berry et al. (2001) reported cattle grazing alpine pastures excreted 56 to 76% of ingested P most of which was excreted in the feces.

IMPACTS OF GRAZING MANAGEMENT ON PLANT AND SOIL CHARACTERISTICS

Grazing Impacts on Forage Characteristics

Productivity and botanical composition of pastures can be rapidly and substantially altered by grazing animals through defoliation, selective grazing, trampling, deposition of dung and urine, and dispersal of seeds (Williams and Haynes, 1993). These alterations can have significant impacts, both positive and negative, on pastures and
grasslands. The 3 main actions of grazing animals on pasture are defoliation, treading, and the removal of nutrients from a pasture or translocation of those nutrients within a pasture (Greenwood and McKenzie, 2001). Heavy grazing pressure reduced vegetative cover, decreased noncapillary porosity of the soil and increased bulk density at the soil surface (Alderfer and Robinson, 1947).

**Forage Sward Characteristics.**

**Sward Height.** Forage sward height is important in the control of intake by grazing animals (Barrett *et al.*, 2001; Tharmaraj *et al.*, 2003). Clary and Leininger (2000) reported that forage intake will be limited at sward heights less than 10 cm. When offered immature swards of different sward heights, cows will select a taller sward (Griffiths *et al.*, 2003). In addition to the importance of maintaining adequate forage sward height to control of forage intake, maintaining a minimum sward height helps preserve forage plant vigor, stabilize sediment, limits stream bank trampling, maintains cattle gains, and provides an easily communicated management criterion (Clary and Leininger, 2000). Forage mass has been shown to be highly correlated with forage height \( R = 0.95; \) Coleman and Forbes, 1998). Papanastasis (1985) showed that surface cover and forage mass decreased as stubble height decreased from 15 to 0 cm in grasslands clipped to simulate cattle grazing.

Forage sward height is not uniform across a pasture. Spatial heterogeneity in grass swards occurs because of spatial variation in the plant species present, the nutritional and environmental factors that influence grass growth, the consumption of grass by the grazing animal and the plant's response to that consumption (Hutchings and Gordon, 2001). When a pasture is continuously stocked during the growing season at a moderate stocking rate, grazing patches will result from an excess of forage supply caused by selective grazing by herbivores (Willms *et al.*, 1988). If forage supply exceeds demand, the regrowth that occurs on previously grazed patches becomes higher in quality, because it will be more immature than the surrounding ungrazed matrix. Over time animals will continue to select forage from the grazed patch, leading to death of plants in the selected patch (Coughenour, 1991). A large amount of spatial and temporal variability of forage sward height was observed in pastures continuously stocked to maintain an average forage sward height of 5 or 10 cm with greater variability observed in the 10 cm forage height treatment (Correll *et al.*, 2003). Papanastasis (1985) observed that grasslands deteriorated when forage stubble height decreased below 3 cm.
Nutritive Value. The nutrient content of forages is affected by growth dynamics of the plant. Neutral detergent fiber (NDF) and acid detergent fiber (ADF) concentrations of forages increase and crude protein (CP) concentrations decrease as forage plants mature from the vegetative, to heading, to anthesis stage (Ferdinandez and Coulman, 2001) resulting in a reduction in forage quality. A reduction in mineral content of the forage is associated with the reduction in forage quality (Kincaid and Cronrath, 1983). The digestibility of NDF, ADF, CP, cellulose, and hemicellulose decrease with increased forage maturity (Barns et al., 1997). These changes in nutrient value of forages as they mature are partially related to changes in the leaf to stem ratio, with the amount of stem increasing with plant maturity (Baron et al., 2000). Crude protein and in vitro organic matter disappearance (IVOMD) concentrations are greater in leaf than stem tissue. Newman et al. (2003) observed average CP concentrations of 12 and 5\% in leaf and stem tissues in a warm-season grass pasture, respectively. In addition to the shift to greater proportion of stems, a reduction in nutritive value of the individual leaf and stem components also occurs as the plant matures. Newman et al. (2003) reported that as forage canopy height increased, the concentrations of CP and IVOMD of the leaf tissue decreased linearly.

Harvest management of forage will also have an impact on its nutrient content. Increasing animal density may initially increase digestible organic matter intake in individual animals because the increased grazing pressure will maintain forage in an immature vegetative state. At some point, animal density increases so that intake becomes limited by forage availability (Hutchings and Gordon, 2001). Correll et al. (2003) reported more young and highly digestible forage was available in areas of the pasture grazed to an intermediate sward height (5 to 10 cm). But at lower sward heights, total mass of forage limited intake and at greater sward heights, the greater maturity of the forage limited its digestibility. Barrett et al. (2001) reported that when dairy cattle were introduced to a new pasture each day, forage sward height and leaf to stem ratio decreased as the day progressed. Additionally, the bite mass decreased as the day progressed with no corresponding increase in bite rate to compensate for the smaller mass. These two factors, a reduction in forage quality (more stem) and forage quantity (reduced forage per bite) consumed, resulted in a reduction in nutrient intake later in the day.

Surface Cover. Over-stocking of livestock on pastures can result in a decrease in surface cover. Mwendera et al. (1997a) reported that in plots stocked at 0, 1.8, 3.0, or 4.2 AUM (AUM = Animal Unit Month) \cdot ha\(^{-1}\), the percentage of surface cover decreased from 100\% to 90\% as stocking rate increased from 0 to 4.2 on
pasture with a low slope (0 to 4%). Surface cover on higher slope (4 – 8%) pastures decreased to 80% at the high stocking rate. With stocking rate held constant at 62.7 steer-days·ha⁻¹ across management systems, surface cover was lower in a season-long continuous stocking system than in either a 4-paddock rotationally deferred or 8-paddock time-controlled rotational grazing system over 12 years under rangeland conditions (Manley et al., 1997). Schulz and Leininger (1990) reported 5 times more bare ground in a pasture stocked at 600 AUM than in an adjacent ungrazed pasture.

Elliot and Carlson (2004) observed that as the amount of surface cover in a pasture grazed by sheep decreased, the concentrations of sediment, total P, and total Kjeldahl N in runoff from the pastures increased linearly. Elliott et al. (2002) reported a linear increase in sediment loss as surface cover decreased in pastures stocked with cattle. Even when surface cover was 100%, some sediment loss still occurred. Total surface cover explained 74% of the variability in sediment yield from plots with less than 30% surface cover (Linse et al., 2001). However, in plots with greater than 30% cover, ground cover was not significantly related to sediment loss. It should be noted that sediment loss from grasslands cannot be explained by vegetative cover alone. Surface cover, slope gradient, ground surface roughness, soil depth, and soil infiltration rate are some of the factors that regulate sediment loss during and after storm events (Moir et al., 2000).

**Botanical Composition.** Grazing pastures by cattle can have a significant impact on the botanical composition of the pasture. Manley et al. (1997) noted a shift in botanical composition over the course of a 12-year grazing study when rangeland was stocked at a high rate. A consistent response to grazing pressure appeared to be selection for plant species with low growing, prostrate growth forms, likely as an avoidance mechanism by the plant to being grazed (Milchunas and Lauenroth, 1993). Forage species diversity of grazed plots was reported to increase compared to ungrazed plots (Bai et al., 2001). The types and relative abundance of species will vary with location and management. Bai et al. (2001) and Schuman et al. (1999) reported a decrease in the amount of grass in response to grazing pressure, but an increase in forbs has been reported with grazing (Manley et al., 1997, Bai et al., 2001). Schulz and Leininger (1990) observed that cattle grazing at a stocking rate of 600 AUM resulted in 9 times more clover than the ungrazed area. Similarly, Bakker et al. (2004) observed that exclusion of cattle from a previously grazed pasture resulted in the disappearance of legumes and an increase in forbs.
Botanical composition of a pasture can impact the amount of surface runoff that occurs from a pasture. Less surface runoff occurred from smooth bromegrass pastures than from alfalfa pastures at 4 and 6 weeks post-harvest and from a smooth bromegrass-alfalfa mixed pasture at 4 weeks post-harvest in the year following establishment. However, by the second year after establishment, no differences were observed in runoff volume between the smooth bromegrass, alfalfa, and mixed pastures (Zemenchik et al., 1996). The authors speculated that these differences in the first year were possibly related to expansion of rhizomes from the bromegrass into the space between seed rows, protecting the soil surface from direct impact of raindrops.

Forage Phosphorus Content and Uptake. Phosphorus in plants may exist in either an inorganic or organic form. Inorganic forms of P found in plants are orthophosphate and pyrophosphate. Organic forms of P in plants include phospholipids, phosphosugars, ADP, ATP, nucleic acids, and phytate (Cole et al., 1977; Valk et al., 2000). Pierzynski and Logan (1993) reported the average P content and P removal in harvested forages for several common forage species (Table 1). The actual P content of forage will vary with seasonal growth dynamics (Saunders and Metson, 1971; Cole et al., 1977; Greene et al., 1987; Grings et al., 1996), P content of the soil (Nash and Halliwell, 1999), soil P desorption kinetics (Raven and Hossner, 1994), available N in the soil (Belanger et al., 2002), application of fertilizer P (Hemingway, 1999; Picone et al., 2003; Toor et al., 2004), soil moisture and ambient temperature (Cole et al., 1977; Kerrigde et al., 1990; Wadsworth et al., 1990), maturity of the plant (Kerrigde et al., 1990; Grings et al., 1996; Wilman, 2004), grazing pressure (Chaneton et al., 1996), forage species (Pierzynski and Logan, 1993; Gaston et al., 2003), and cultivar of the plant (Belanger et al., 2002). The variation in P content under different conditions is related to the ability of the plant to take up P across the root, an energy dependent process (Cole et al., 1977), the presence of actively growing meristematic tissue (Wilman, 2004), the leaf to stem ratio of the plant, and the amount of dead tissue in the plant (Greene et al., 1987).

Plants grown in soil with high levels of plant available P contain more total P and a greater percentage of P in a water-soluble form than do plants grown in low P soils (Nash and Halliwell, 1999). Coates and Ternouth, (1992) determined that P intake by cattle grazing pastures annually fertilized with 10 kg P·ha⁻¹·yr⁻¹ for 9 years was 60% greater than cattle grazing an unfertilized pasture. The difference in P intake was attributed to greater P content of forage from the fertilized pasture, as forage intake was similar between treatments. Annual
P uptake of pasture forage fertilized at 0 to 200 kg P·ha\(^{-1}\) as triple superphosphate varied between 9 and 34 kg P·ha\(^{-1}\)·yr\(^{-1}\) with P concentrations ranging from 0.2 to 0.3% (Picone et al., 2003). During and Weeda (1973) showed that P application as either dung or superphosphate fertilizer were equally as effective at increasing the P concentration of forage compared to the P concentration of forage in a pasture receiving no supplemental P.

Table 1. Phosphorus content, forage yield, and P removal in selected forages. Adopted from Pierzynski and Logan, 1993.

<table>
<thead>
<tr>
<th>Forage</th>
<th>P conc., %</th>
<th>Forage Yield, tons·ha(^{-1})</th>
<th>P removal at harvest, kg·ha(^{-1})</th>
<th>P removal, kg P·ton forage(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alfalfa</td>
<td>0.25</td>
<td>13.4</td>
<td>34.7</td>
<td>2.6</td>
</tr>
<tr>
<td>Bluegrass</td>
<td>0.22</td>
<td>6.7</td>
<td>14.6</td>
<td>2.2</td>
</tr>
<tr>
<td>Corn Silage</td>
<td>0.06</td>
<td>67.2</td>
<td>39.2</td>
<td>0.6</td>
</tr>
<tr>
<td>Fescue, Tall</td>
<td>0.44</td>
<td>9.0</td>
<td>39.2</td>
<td>4.4</td>
</tr>
<tr>
<td>Orchardgrass</td>
<td>0.32</td>
<td>10.0</td>
<td>31.4</td>
<td>3.2</td>
</tr>
<tr>
<td>Red Clover</td>
<td>0.25</td>
<td>9.0</td>
<td>22.4</td>
<td>2.5</td>
</tr>
<tr>
<td>Timothy</td>
<td>0.22</td>
<td>6.7</td>
<td>14.6</td>
<td>2.2</td>
</tr>
</tbody>
</table>

Nitrogen availability is known to have a positive relationship with P content and uptake by forage (Cole et al., 1977; Hemingway, 1999; Belanger et al., 2002). However, under conditions of low available soil P, the application of N fertilizer with no P fertilization has been shown to result in decreased P concentrations of the forage. The subsequent addition of either 112 or 224 kg P·ha\(^{-1}\) under low P, high N conditions increased the P content of forage to an adequate level to meet the maintenance requirements of cattle (Black and Wight, 1979). Belanger et al. (2002) speculated that the increase in P uptake by plants when N is not limiting is related to an increased efficiency of P uptake by roots when N was sufficient since they observed no increase in root mass production with increased N application.

The increased uptake of P by forage with increases in soil P and N has not been observed in all studies. When dairy manure was applied at different rates (N application ranged from 0 to 672 kg·ha\(^{-1}\) and P application ranged from 0 to 136 kg·ha\(^{-1}\)) to an orchardgrass (Dactylis glomerata) pasture, forage P concentration averaged 0.46% and was not significantly affected by slurry application rate (Soder and Stout, 2003). Weeda (1977) reported no increase in P content of legumes fertilized with animal manure, but did observe an increase in P content of grasses under the same conditions.

**Forage productivity.** Milchunas and Lauenroth (1993), in an analysis of forage productivity of 127 grazed grassland studies from around the world, found mean annual net primary productivity (ANPP) in the
studies to be $257 \pm 154 \text{ g m}^{-2} \text{ yr}^{-1}$ with 44% of this ANPP consumed by grazing livestock. Seventeen percent of the studies showed a positive relationship between grazing and ANPP. Studies that showed improvements in ANPP were at low levels of consumption with a limited number of years of treatment and many of the increases were small. On average for all of the studies, grazing of livestock resulted in a 23% reduction in ANPP. In some instances, grazing may stimulate productivity through enhanced nutrient cycling brought on by the physical deterioration, soil incorporation, enhanced rate of decomposition caused by consumption and excretion of forage nutrients and the action of hoofs on the soil surface (Schuman et al., 1999).

The grazing optimization hypothesis of DeMezancourt et al. (1998) states that primary productivity or plant fitness at first increases with grazing pressure, reaches a maximum at a moderate rate of herbivory, and then declines as grazing pressure increases. Net primary productivity was reported to decline as intensity of grazing increased from 0 to 4.2 AUM·ha$^{-1}$ in the highlands of Ethiopia (Mwendera et al., 1997). Forage production was greater in grazed (1174 kg·ha$^{-1}$) than ungrazed (1074 kg·ha$^{-1}$) mixed prairie plants (Dormarr et al., 1997). Gillen et al. (2000) observed no stocking rate by year interaction on forage productivity in pastures stocked at various stocking densities ranging from 23 to 51 AUD·ha$^{-1}$ over a 7-year period. This result implies that stocking rate had neither a positive nor negative impact on herbage production potential of the pastures.

It is generally accepted that a rotational stocking system will allow a greater number of animals to be stocked on a unit of land than if that land is managed in a continuous stocking system. The biological reason for this improvement in stocking rate is controversial (Heitschmidt et al., 1987a) and seems to be in disagreement with the evidence that increasing stocking rate reduces ANPP. Heitschmidt et al. (1987b) concluded that a small (10 to 15%) increase in carrying capacity could be expected following establishment of a properly managed rotational stocking system. This increase is related to better livestock distribution and improved forage utilization rather than an increase in ANPP. Based on modeling of rotational stocking systems, Woodward et al. (1995) determined that subdivision of pastures will generally not increase the productivity of a pasture. These results are supported by Gillen et al. (2000) who reported that the amount of herbage transferred from live to dead was greater at a low than at a high stocking rate because of greater forage consumption associated with greater livestock numbers at the high stocking rate. But no evidence that plant productivity was affected by stocking rate at the community level was observed. Similarly, Mapfumo et al. (2002) reported
greater litter accumulation (less forage utilization) under light grazing than under medium or high levels of stocking because of greater forage removal in the heavy treatment.

Grazing not only impacts aboveground productivity, but belowground productivity as well. Grazing pressure increased root mass in more sites than it decreased root mass in the analysis conducted by Milchunas and Lauenroth (1993) with a 20% average increase in root mass with grazing across all studies. No relationship between aboveground production and root mass in response to long-term grazing was observed in this analysis (Milchunas and Lauenroth, 1993). Root masses of smooth bromegrass pastures were reported to be maximum under medium grazing pressure (compared to heavy or light grazing pressure) in the upper 60 cm of soil surface, while root mass of meadow bromegrass was greatest under light grazing pressure (Mapfumo et al., 2002). Chaneton et al. (1996) reported little difference in root biomass between an ungrazed pasture and one grazed with a stocking rate of 0.55 cattle·ha⁻¹.

**Grazing Impacts on Soil Chemical and Physical Characteristics**

Livestock can affect soil quality through compaction, erosion, and changes in plant community. Inappropriate grazing practices may accelerate erosion and sediment transport to water, alter stream flow, and disrupt aquatic habitats (CAST, 2002b). The effects of livestock grazing on soil physical properties were reviewed by Greenwood and McKenzie (2001). Soils from ungrazed grasslands have a higher proportion of pores, greater hydraulic conductivity, and a greater proportion of small-sized aggregates than do grazed pastures (Singleton and Addison, 1999). Some sediment and nutrient loss occurs from ungrazed pastures with complete ground cover, but grazing of sheep increased the sediment load lost from pasture by both increasing the concentration of sediment in runoff and the amount of runoff (McDowell et al., 2003; Elliot and Carlson, 2004). Manure deposition by grazing livestock can have a positive impact on pasture soil through the addition of organic compounds in animal manure which adds to soil OM, increasing the water holding capabilities and structure of the soil (Hubbard et al., 2004).

Changes in soil physical characteristics caused by grazing may be slow or rapid. Mapfumo et al. (2000) reported negligible changes in soil physical characteristics after three years of grazing at different stocking rates. Elliot and Carlson (2004) showed that 3-days of intensive stocking of sheep resulted in a 66% reduction in infiltration rate. Once the soil in a pasture is affected by grazing, it may require weeks (Elliot and
Carlson, 2004), months (Nguyen et al., 1998), or years (CAST, 2002b) to restore depending on severity of damage and environmental conditions.

**Soil Phosphorus.**

*Forms of Phosphorus in the Soil.* The P concentration in the soil can vary widely, from 100 to 2500 ppm, depending on parent material, texture, and management factors. Only a small fraction of the total P in the soil is available to support plant growth at any time (Daniel et al., 1994). Phosphorus in the soil exists in several forms. Primary mineral P exists as apatites, which are acid extractable and are the ultimate source of all soil P.

The release of P from primary mineral is slow, occurring on a geological time scale. Secondary mineral P is chemically adsorbed to the surface of minerals in the soil, most commonly Fe, Al, or carbonates (Smeck, 1985). In calcareous (basic) soil, P is generally found as calcium phosphate. In noncalcareous (acidic) soils, P is generally found as iron and aluminum phosphate (Blanchar and Caldwell, 1964). The fraction most available to plants is termed soluble P and can be extracted from the soil with water or a dilute salt solution. Labile P is in equilibrium with the soluble soil P pool and is isotopically exchangeable or anion resin extractable P. Organic P exists primarily as ester linkages on inositols with lesser amounts in phospholipids and nucleic acids, usually present as microbial sources, and is generally turned over on an annual basis. Occluded P is physically encapsulated by minerals that are structurally devoid of P and is physically restricted from interacting with other forms (Smeck, 1985).

**Soil Test Phosphorus.** Several laboratory tests exist for the determination of the amount of plant available soil P that will be available to support plant growth at any time. Soil extractable P has been shown to be highly correlated with animal liveweight gains, forage P concentration, and dietary P intake in grazing cattle (Kerrigde et al., 1990). However, a limitation of these agronomic soil tests is that the processes that control plant uptake of P are different from the processes controlling P removal in runoff which decreases the usefulness of these tests to determine environmentally sound soil P concentrations (Kleinman et al., 2000).

Sharpley et al. (1993) stated that agronomic soil test P methods might not be appropriate to relate soil P level to enrichment potential of bioavailable P in runoff. Climatic and soil characteristics, and agronomic factors that influence the amount of surface runoff play a larger role in determining P loss to surface waters than does the
soil test P content of surface soil (Sharpley et al., 1996). Even though soil test P level is not considered to be a good indicator of P losses to surface waters, Cornish et al. (2002) observed a trend for increased total and soluble P concentrations in surface runoff as soil Bray-1 P increased and soil P sorption decreased. It has also been observed that runoff plots with lower soil test P had lower ratios of dissolved reactive P (DRP) to total P in the runoff (Schroeder et al., 2004). The greatest relationships between soil test P and P concentration in surface runoff have been derived in pure soil-pasture systems at small scales where uniformity is high and variability associated with plant biomass, livestock, and manure effects are eliminated (Dougherty et al., 2004).

In a 2001 survey of soil testing labs in the United States and Canada, it was reported that 53% of soil samples tested high or very high in soil P (Fixen, 2002), an increase from 49% of samples evaluated in a 1997 survey (Fixen, 1998). The same survey indicated that 61% of soil samples from Iowa in 2001 tested high or very high in P (Fixen, 2002). Once soil test P levels exceed crop requirements, the potential for P loss in runoff and erosion is greater and no improvement in crop yield can be expected (Sharpley et al., 1993; Sharpley et al., 1994; Nelson, 1999; Pautler and Sims, 2000). Additionally, soils with greater soil test P levels generally have higher concentrations of soluble and desorbable P (Pautler and Sims, 2000). These soils will presumably have greater potential to release P into runoff waters or into surface waters following erosion of P-rich particles.

Phosphorus Distribution. Within states and regions, distinct areas of general P deficits and surpluses in the soils exist (Daniel et al., 1998). In an Arkansas study, mean soil Mehlich-3 P ranged from 7 mg·kg⁻¹ in pastures that had never been fertilized with manure to 437 mg·kg⁻¹ in pastures that had received long-term treatment with poultry litter (Daniels et al., 2001). On a statewide scale, the increasing disintegration between row-crop and animal production is producing an uneven distribution of P in the soil. In the state of Arkansas, soil test P data shows that median Mehlich-3 extractable P for warm and cool-season grass pastures has increased by 2.5 mg P·kg⁻¹·yr⁻¹ between 1995 and 2002, with no change in soil P for crop ground (Slaton et al., 2004).

Within a pasture, the soil surface generally has higher soil P content than lower strata in the soil profile. This is caused by surface fertilizer application, defecation by animals, and plants acting as biological pumps to bring nutrients to the soil surface (Nash and Halliwell, 1999). Weeda (1977) showed that the upper 3.8 cm of the soil surface under a dung pile had an elevated level of plant available P compared to areas not
directly under manure piles. Olsen extractable P has been shown to increase in the upper 5 cm of the soil surface under dung piles within 20 days of dung deposition (Aarons et al., 2004a). Extractable P in the top 15 cm of the soil surface was greater in pastures with heavy stocking rates than light or medium stocking rates after 4 years of management (Baron et al., 2001). When forage is harvested as hay, plant available nutrients have been shown to decline in the upper 15 cm of the soil profile (Mathews et al., 1994).

Accounting for spatial variability of soil P within pastures is an important consideration for nutrient management strategies (Daniels et al., 2001). Within a pasture, some areas will tend to concentrate nutrients, while other areas will become depleted of nutrients over time. Bray-I P was observed to be greater at lower portions of a hill slope than at higher portions of the slope, possibly related to transport of P from higher to lower areas in runoff over time (Cornish et al., 2002). In some dry grassland environments, the accumulation of nutrients below a particular plant, called resource islands, may develop as a result of biological accumulation of above and below ground litter by plants and the physical erosion and deposition processes that redistribute material within the landscape (Burke et al., 1998). Sauer and Meek (2003) reported greater variability in soil P in a pasture that had never received P fertilizer than in a pasture receiving annual application of poultry litter.

Cattle may also be responsible for the redistribution of nutrients within a pasture. Extractable N, P, and K concentrations in the soil were observed to be greater near shade and water sources in a pasture than areas at greater distances from shade and water. Visual observation of cattle location in this study indicated that cattle spent a majority of mid-morning to late-afternoon near shade and water sources allowing greater time for nutrient deposition in urine and feces (Mathews et al., 1994). Potassium, P, and Mg were found to be 5.0, 2.4, and 1.1 times greater 1 m from a shade or water source than at greater distances from these concentration areas within a pasture (Schomberg et al., 2000). Sauer and Meek (2003) reported elevated soil P levels near shade trees, but not near water sources. Carran and Theobald (2000) observed that total C, N, and P and plant available (Olson) P were lower in the soil along the fence line of a pasture than in the middle of a pasture that had been grazed for a 23 year period. The authors speculated that this difference was caused by animals facing the fence while defecating, creating a zone that did not receive nutrients from the feces.

When sheep were allowed free access to a pasture with slopes ranging from 0 to 45 degrees, P was transported from high slope to low slope areas by the animals (Gillingham et al., 1980). A greater amount of
time was spent at low slope areas (0 to 15 degrees) where animals rested and ruminated, resulting in greater fecal deposition in the low slope than high slope areas. A positive P balance occurred in the low slope sites (+60.7 kg·ha\(^{-1}\)) and negative balance occurred at high slopes; -19.5 kg·ha\(^{-1}\) at 25 degrees and -15.3 kg·ha\(^{-1}\) at 35 degrees.

Amount and availability of nutrients returned to the pasture in dung and urine are influenced not only by the concentration and forms of nutrients, but also by the number of excretions, size of each excretion, and surface area covered by excreta (Williams and Haynes, 1993). During and Weeda (1973) estimated that 40% of the surface area of a pasture will receive dung in one year and 75% of a pasture will receive dung in 3.5 years, assuming that cattle defecate 12 times per day and a stocking rate of 4 cattle per hectare. Cattle average 8 to 12 urinations per day with 1.6 to 2.2 L per urination, covering an area of 0.16 to 0.49 m\(^2\) per urination and 11 to 16 defecations per day at 1.5 to 2.7 kg of DM per defecation, covering an area of 0.05 to 0.09 m\(^2\) per defecation. Sheep average 18 to 20 urinations per day with 0.1 to 0.18 L per urination, covering an area of 0.03 to 0.05 m\(^2\) per urination and 7 to 26 defecations per day at 0.03 to 0.17 kg per defecation, covering an area of 0.008 to 0.025 m\(^2\) per defecation. The area affected by nutrients in the urine is often twice the area wetted and the nutrients in dung can influence an area 1 to 6 times the area of the dung pile (Williams and Haynes, 1993).

**Fecal Decomposition and Phosphorus Release to the Soil.** Nutrient cycling in a grazing environment can be through either a slow or fast cycle. The slow nutrient cycle occurs when ungrazed plant material dies and the plant litter slowly degrades, returning nutrients to the soil. In the fast cycle, herbivores consume plant material that is degraded during digestion and excreted in a readily available form in the feces and urine of the animal (Bakker *et al.*, 2004). The distribution of fecal and urine deposition within the pasture and the incorporation of excreted nutrients into the soil will determine the availability of those nutrients to support plant growth or to impact water quality. Feces are the major path of P deposition from cattle (Knowlton and Herbein, 2002). Weeda (1977) reported that 24% of P applied in feces to a pasture was recovered in forage in the first 1.5 years following application and 37% was recovered in the forage over a 3.5-year period (Weeda, 1977).

The incorporation of fecal P into the soil may occur by either physical degradation and incorporation of feces into the soil or by leaching of P from feces caused by precipitation; the importance of either route
depending on the amount of precipitation that occurs (Williams and Haynes, 1993). The rate at which sheep feces decomposed was faster under the wetter conditions typical of New Zealand's winter than during the dryer summer months, requiring 28 and 106 days during the winter and summer, respectively (Rowarth et al., 1985). Aarons et al. (2004a) reported that the concentration of P in feces from dairy cattle applied to pastures decreased by about 26% over the course of a study, indicating that P was leached out of the dung at a faster rate than dung decomposition. In another study, Aarons et al. (2004b) reported some leaching of P occurred from the fecal piles, but that the primary mode of P deposition in the soil from dung was through incorporation as the dung pile was degraded as a result of the actions of soil fauna. Rowarth et al. (1985) observed no change in total P concentration of feces with time, demonstrating physical rather than chemical decomposition of the feces. The influence of feces on soil nutrients spreads about 15 cm beyond the boundary of the fecal pile possibly because of rain-wash, root spread, and the presence of stoloniferous plants such as white clover in the pasture (During and Weeda, 1973). The influence of a single fecal excretion may be increased in the event of liquid stools or the action of hoof traffic in breaking down and spreading out the fecal pile (During and Weeda, 1973).

In addition to being a source of P to pasture soils, feces have several other benefits compared to the application of superphosphate. Feces may increase the pH, CEC, total N, organic C, and exchangeable Mg and Ca of the soil (During et al., 1973). This higher pH decreased soil sorption of P, making it more available to the plant (During et al., 1973). The higher organic matter may have improved soil water-holding capacity, also contributing to plant growth and P uptake (During et al., 1973; During and Weeda, 1973). A negative effect of fecal deposition in pastures is that herbage production under the fecal pile will be stopped in the first year following application of dung. This loss of forage may be compensated by increased production around the edge and in the following year. Over a 1.5-year period following application of feces to a pasture, forage yield increased by about 50% within 12.7 cm of the boundary of the dung patch (Weeda, 1977).

**Soil Moisture.** Soil moisture has been reported to be lower in grazed pastures than in ungrazed pastures (Naeth et al., 1991a; Milchunas and Lauenroth, 1993; Donker et al., 2002) and as grazing pressure increased, soil moisture declined further (Naeth et al., 1991a; Teerdoff et al., 1999). Decreased soil moisture is likely caused in part by the combined effects of lower infiltration and higher evaporation associated with
grazing (Greenwood and McKenzie, 2001; Pires da Silva et al., 2003). Additionally, a reduction in soil organic matter resulted in a reduction in water holding capacity of the soil (Naeth et al., 1991b; Betteridge et al., 1999) and a reduction in leaf litter resulted in a reduction of aboveground water-holding capacity (Naeth et al., 1991c). Water-holding capacity of standing leaf litter, roots and course litter has a greater impact on total water-holding capacity of a soil than does medium, fine, and very fine organic matter particles (Naeth et al., 1991c). Early season and high intensity grazing reduced litter and organic matter accumulation in the soil more than late season intensive grazing or light intensity grazing during either season (Naeth et al., 1991b).

During parts of the year characterized by periods of low precipitation, the effects of grazing on soil moisture become less obvious (Naeth et al., 1991a). Low soil moisture can be a factor limiting pasture growth during parts of the year (Alderfer and Robinson, 1947; Greenwood and McKenzie, 2001). Therefore, by managing to maintain adequate forage cover during wetter periods, reductions in forage production caused by low soil moisture can be minimized. A great deal of spatial variability can exist in the moisture content of the upper soil surface layers which may be attributed to variation in soil characteristics, topography, and water routing processes (Merz and Plate, 1997).

When soil moisture content is high, greater particulate matter loss may occur in runoff because of an increased volume of surface flow from these sites (McDowell and Sharpley, 2002). In general, the time to peak flow of surface runoff decreases as soil moisture increases. However, when soil moisture is very low, time to peak flow may be rapid because of the development of a hydrophobic organic layer on the soil surface preventing infiltration initially (Fraiser et al., 1998).

**Treading Damage and Surface Roughness.** Treading damage in pastures results from the grazing of livestock on moist soils. Nguyen et al. (1998) reported that damage caused by 2 to 3 days of winter grazing on wet soils required 6 months to recover to the point that surface runoff characteristics returned to pre-grazing conditions. Singleton and Addison (1999) reported that soils that had experienced treading damage (pugging) 18 months previous and visually appeared to have returned to a normal pre-pugging conditions, still exhibited greater soil bulk density than soil that had not experienced pugging damage. Random surface roughness was shown to increase when cattle were allowed to graze pastures with moist soil, while grazing of sheep under the same conditions resulted in no increase in surface roughness (Betteridge et al., 1999). The presence of bare
ground in a pasture will increase the amount of treading damage caused by grazing animals (Betteridge et al., 1999).

Treading damage caused by livestock resulted in reductions in soil macroporosity and water infiltration rate causing increased surface runoff and total P loss (Nguyen et al., 1998; Elliot and Carlson, 2004). As the amount of treading damage measured as hoof prints per square meter increased, the amount of sediment loss increased (McDowell et al., 2003). Sediment loss associated with treading damage is greatest immediately following the treading event. Elliott et al. (2002) reported 500 to 2200 g·m⁻² of sediment was lost during simulated rainfall events immediately following a treading event, but two months later, sediment loss was reduced to less than 100 g·m⁻². By four months following the treading event, sediment load was negligible and by one year, pastures had returned to pre-treading conditions. Elliot and Carlson (2004) reported that after 6 weeks of exclusion of sheep from a pasture that had experienced treading damage, the soil surface had recovered so that the severity of initial treading damage was not correlated with the infiltration capacity of the soil.

**Soil Compaction.** Penetration resistance and bulk density are the most common parameters to evaluate the extent of soil compaction (Mapfumo et al., 1999). Soil compaction is caused by an applied stress in excess of the bearing strength of the soil that causes the soil to fail, resulting in increased bulk density with smaller, fewer, and less continuous macropores (Greenwood and McKenzie, 2001). Penetration resistance gives an indication of shear resistance of the soil, which in turn depends on strength properties of the soil (Ayers and Perumpral, 1982). Penetration resistance in the range of 1000 to 2000 kPa can restrict root growth (Whalley et al., 1995). Penetration resistance, a measure of the ease at which an object can be pushed into the soil, is affected by characteristics of the penetrometer, such as angle, diameter, and roughness of the cone, the rate of penetration and characteristics of the soil such as soil moisture, soil bulk density, and soil type (Ayers and Perumpral, 1982; Perumpral, 1987; Vas et al., 2001). Bulk density is a measure of the dry weight of a given volume of soil and is often expressed as g·cm⁻³ (Murphy et al., 2004). Penetration resistance has been shown to be a more sensitive measure of soil compaction than is bulk density (Naeth et al., 1990a; Rodd et al., 1999).

Soil compaction, measured by either penetration resistance or soil bulk density, associated with grazing livestock has been reported to be limited to the upper 2.5 cm of the soil surface (Mapfumo et al., 1999).
greater in the top 6 cm of the soil profile (Rodd et al., 1999), greater in the top 10 cm of the soil surface (Gilley et al., 1996), greatest between 7 and 10.5 cm in a congregation area of a pasture (Mulholland and Fullen, 1991), and to increase to a depth of 30 cm (Krenzer et al., 1989). The differences in soil depths at which cattle cause compaction in the various studies is likely a result of soil characteristics, animal characteristics, and management practices. Soil properties such as texture, organic matter, water content and other factors like environmental conditions and grazing intensities govern the degree to which compaction occurs resulting in the great variability in bulk densities that can occur within the same pasture (Mapfumo et al., 1999). Treading effects of cattle on soil is a function of the animal's mass, hoof size, and kinetic energy resulting in greater pressure on the soil while walking as opposed to standing. The pressure exerted on the ground by hooves has been reported to range from 98 to 192 kPa for cattle and from 48 to 83 kPa for sheep (Greenwood and McKenzie, 2001).

Plant material on the soil surface provides protection to the soil and may be important in resisting compaction and preserving aggregation of the soil under grazing (Greenwood and McKenzie, 2001). The use of longer rest periods between grazing periods in a rotational stocking system allows the pasture to recover and decreases soil penetration resistance (Rodd et al., 1999). Once soil compaction has occurred, recovery of the soil is controlled by wetting/drying cycles, freeze/thaw cycles, growth/decay of roots, and action of soil animals such as dung beetles and worms (Mapfumo et al., 1999; Rodd et al., 1999; Greenwood and McKenzie, 2001). Soil bulk density and penetration resistance have been reported to be greater in the fall than in the spring. This difference may be related to greater soil moisture in the spring or to the natural freezing and thawing action of on the soil during the winter (Mapfumo et al., 1999; Donker et al., 2002). Relatively small changes in soil moisture are able to significantly alter penetration resistance of the soil (Pires da Silva et al., 2003). The degree to which penetration resistance is affected by soil moisture is influenced by the texture, as determined by clay to sand ratio, of the soil (Ayers, and Perumpral, 1982). Maximum penetration resistance is not associated with the minimum amount of moisture, but usually occurred between 5 and 10% soil moisture (Ayers and Perumpral, 1982).

Lower soil bulk density leads to greater water infiltration through better soil structure (Teerdoff et al., 1999). Grazing of elk at a stocking rate of either 4.16 or 2.08 AUM·ha⁻¹ resulted in an increased bulk density of
the soil and a decrease in soil moisture related to the lower infiltration rate (Donker et al., 2002). Root growth of forage plants may be restricted when penetration resistance exceeds 2 MPa (Mapfumo et al., 1999). Both a reduction in soil moisture and restricted root growth can result in a reduction in forage productivity.

High stocking rates have been shown to have greater compacting effects on pasture soils compared to low stocking rates (Naeth et al., 1990a; Mapfumo et al., 1999; Teerdoff et al., 1999; Donker et al., 2002). The greater soil compaction in pastures with heavy stocking was likely related to the greater surface area trampled and greater forage removal, reducing the cushioning effect of the forage (Naeth et al., 1990a). Weigel et al. (1990) reported greater penetration resistance in grazed than in ungrazed paddocks. While Greenwood et al. (1997) reported greater soil bulk density in grazed than in ungrazed pastures, but increasing stocking rate of grazed pastures (10, 15, or 20 sheep·ha⁻¹) did not result in increased soil bulk density. When soil strength within grazed paddocks was divided into ‘within a hoof print’ and ‘outside of a hoof print’, the soil strength outside of the hoof print in a grazed paddock was not different from that of an ungrazed paddock during three of the five sampling months of the study. In the month of April, penetration resistances averaged 1.07, 0.59, and 0.53 MPa within hoof print, in a grazed paddock but outside of a hoof print, and in an ungrazed paddock, respectively.

Infiltration Rate and Hydraulic Conductivity. Hydraulic conductivity of the soil, the ability of the soil to transmit water, decreased linearly with increasing soil damage from cattle treading (Elliott et al., 2002) because of a reduction in pore space under hoof prints (Mulholland and Fullen, 1991). As treading with a mechanical cow hoof on soils from cultivated land and grassland increased from 0 to 120 imprints·m⁻², soil microporosity and soil infiltration rate decreased, resulting in an increase in the amount of overland flow. Microporosity of the grassland soils was greater than microporosity of cultivated soils following treading, indicating a greater ability for grasslands to withstand treading damage (McDowell et al., 2003).

Infiltration rate tends to be higher in intensively managed pasture systems than in cropping systems because of the maintenance of ground cover and favorable soil structure resulting from higher levels of organic matter in pasture systems (Dougherty et al., 2004). Naeth et al. (1990b) reported a decrease in infiltration rate of grazed paddocks compared to an ungrazed area, even at low stocking rates. Heavy stocking rates in the early spring (the wetter period of the year) resulted in the greatest reduction in infiltration rate. Mapfumo et al.
reported a reduction in soil water-holding capacity at heavy stocking rates after 3 years of management while light and medium stocking rates resulted in a slight increase in water-holding capacity over the same period. Hydraulic conductivity of soil was reduced by 80% in a soil that had experienced pugging damage compared to adjacent soils, of the same soil type that had never been grazed (Singleton and Addison, 1999). Mulholland and Fullen (1991) measured soil bulk density and infiltration rate at different distances from a livestock congregation area (portable feeder). Bulk density was greatest near the feeder (1.48 g·cm⁻³) and decreased to 1.22 g·cm⁻³ as distance from the feeder increased. Infiltration rate followed an opposite trend being lower at the feeder (1.06 mm·hr⁻¹) and increasing to 72.95 mm·hr⁻¹ at greater distances from the feeder.

In addition to grazing pressure, infiltration rate is affected by soil moisture. Naeth et al. (1990b) reported infiltration rate to be inversely related to the antecedent soil moisture. Infiltration rate during rainfall simulations was higher in summer (dry season) than in winter (wet season) in a New Zealand pasture stocked by sheep (Elliot and Carlson, 2004).

Grazing methods that allow litter to accumulate on the soil surface (Naeth et al., 1991b) and limit grazing on wet soils (Naeth et al., 1990b) can limit reductions in infiltration rate associated with grazing. Once damage has occurred, the actions of earthworms and plant roots can act to increase the hydraulic conductivity of pasture soils (Betteridge et al., 1999). Cooper et al. (1995) compared the hydraulic conductivity of a pasture to a portion of a pasture from which the animals were fenced out and an ungrazed area of native vegetation. Infiltration rates of the soils were 15, 6340, and 4769 mm·hr⁻¹ for the pasture, former pasture, and native area, respectively.

**Rainfall - Surface Runoff Interactions**

**Rainfall Simulations vs. Natural Rainfall.** Small plot rainfall simulators are a common tool in the study of the impacts of land use practices on surface runoff characteristics. The benefit of rainfall simulators over natural rainfall in runoff studies is that the researcher has greater control, allowing rainfall to be produced when and where needed and to be of the character and for duration required for the study. The limitations associated with rainfall simulators include limited plot size, edge effects at the plot boundary, differences in drop size distribution and energy characteristics of natural and simulated rainfall, and the intricate variability of natural rainfall compared to simulated rainfall (Bowyer-Bower and Burt, 1989). The short flow length of
rainfall simulators may have a greater proportion of larger sediment particles than would be expected in runoff from an entire pasture because of the selective transport of smaller particles over long distances (McDowell and Sharpley, 2002). The use of small plot size rainfall simulators may result in high amounts of variability between plots within the same field because of differences in hydrology (Delaune et al., 2004). The limits and inherent errors in small plot rainfall simulation studies should be recognized and not applied to an entire watershed (Bowyer-Bower and Burt, 1989; Nash and Halliwell, 1999; Humphry et al., 2002).

Results from small plot erosion studies must be carefully analyzed when extrapolating them to the watershed scale. Soil types, slopes of the land, topography, and location of small plots within the landscape can all be factors in the amount and character of runoff generated (Meyer et al., 1999). Nash and Halliwell (1999) found that the use of small plot runoff studies might not be good indicators of catchment-scale sediment and P losses, however, small plot rainfall simulations do allow for relative comparisons of different land use practices and are sufficient in runoff studies relating soil P and runoff P (Cornish et al., 2002; Delaune et al., 2004).

Many processes are scale dependent and residence time of surface runoff in most plot studies would be lower than that in a catchment because of the reduced time for dissolution and desorption processes to occur. Elliot and Carlson (2004) stated that sediment and nutrient losses from small-scale rainfall simulators are much smaller than what would be expected to occur at a catchment scale. Stream bank erosion is predominantly responsible for the loss of sediment and nutrients to surface waters and that overland flow, which is measured with the use of rainfall simulations, is only a very minor contributor to sediment and nutrient loss to surface waters (Elliot and Carlson, 2004).

Rainfall-Surface Soil Interactions. In a review of the processes that control P loss in runoff from pastures, Dougherty et al. (2004) stated that different processes control the transport of P in runoff at the paddock, farm, and watershed scales. Runoff coefficients (runoff coefficient = 1 - (evapotranspiration / precipitation); Savenije, 1996) are high and runoff residence times are short in the small areas used in rainfall simulation studies. Under natural rainfall conditions at the paddock and watershed scale, runoff coefficients are relatively low and residence times relatively long (Dougherty et al., 2004). Similar to sediment erosion, a few rainfall events will often account for the majority of the nutrient movement within the landscape during the year (McColl and Gibson, 1979).
The first steps in the transport of dissolved P in runoff are desorption, dissolution, and extraction of P from plant material and soil. These processes occur when rainfall interacts with the surface soil before leaving as runoff (Sharpley et al., 1993). Rainfall and the subsequent runoff interacts with all of the P pools on the soil surface, including plant, manure, fertilizer, microbial and soil P (Dougherty et al., 2004). The depth at which rainfall interacts with the soil has been reported to be 2 to 5 cm, but the actual depth at a given location will vary with soil type and condition, rainfall kinetic energy and intensity, and slope of the land (Sharpley, 1985; Ahuja, 1986; Sharpley et al., 1996). The effective depth of interaction of precipitation with soil was shown to increase from 1.3 to 37.43 mm as rainfall intensity increased from 50 to 160 mm·hr⁻¹ and soil slope increased from 2 to 20% (Sharpley, 1985).

Cornish et al. (2002) stated that the duration of a rainfall event was the best indicator of soluble P concentration in surface runoff. Longer events resulted in higher soluble P concentrations from paddocks, but lower soluble P concentrations from a farm, while shorter events had the opposite trend. These trends were possibly related to areas near streams playing a more important role in soluble P transport in short events.

As the length of the flow path of runoff increases, so does the selective erosion of fine materials (<20 μm), as larger particles tend to drop out of and fine particles tend to remain in the runoff (McDowell and Sharpley, 2002). Smaller particles have a greater surface to volume ratio allowing proportionally greater surface area to transport P (McDowell and Sharpley, 2002). As the runoff passes over the surface profile, P becomes more bioavailable by sorption-desorption processes and preferential transport of P on clay-sized material as it moves through the landscape (Sharpley et al., 1993).

As surface runoff concentrates over longer flow lengths, the energy of the water per unit of soil area tends to increase, resulting in less physical and chemical removal of nutrients from the water and greater mass movement of soil through soil erosion (Nash and Halliwell, 1999). The amount of overland flow and particle loss increased as plot length increases. As a result, a longer flow path will produce a larger amount of total P to dissolved P than will a short flow path (McDowell and Sharpley, 2002).

**Soil Erosion.** Soil erosion is a natural process that will occur in the absence of human activity, but the rate of erosion can be accelerated by agricultural and other land management practices (Trimble and Mendel, 1995; Uri, 2000). Uri (2000) reviewed problems associated with soil erosion, soil conservation programs in the
United States, and regional trends in rates of soil erosion in the United States. Sediment erosion is the
detachment of soil particles from the soil mass by rainfall impact and runoff shear and transport of the resulting
sediment by raindrop splash or flowing runoff. Different soils have different levels of susceptibility to erosion,
referred to as the erodibility of the soil and represented as the K factor in the universal soil loss equation
(USLE). Soil erosion has been shown to increase with slope steepness (Shainberg et al., 1992) and increasing
rainfall intensity (Huang, 1998). Seasonal variability has been observed in the amount of sediment eroded from
a site, usually reaching a peak during the wetter portion of the year (Elliott et al., 2002). A greater percentage
of sediment loss will occur near the beginning of a storm, with the most highly erodible particles being removed
first and the more stable particles remaining (Huang, 1998; Linse et al., 2001).

Mean suspended sediment in surface runoff has been shown to be greater from cultivated land than
from grassland (McDowell et al., 2003). Burwell et al. (1975) compared erosion losses in surface runoff under
different land use practices and found that fallow, cultivated fields lost 14 ton sediment·ac⁻¹·yr⁻¹, compared to a
field in continuous corn production losing 6.7 tons sediment·ac⁻¹·yr⁻¹, and a hay field losing 0.008 tons
sediment·ac⁻¹·yr⁻¹. Increasing stocking density of cattle from 0 to 1.8 and 4.2 AUM·ha⁻¹ has been shown to
result in an increase in erosion from pastures (Mwendera and Saleem, 1997). If adequate forage canopy and
base cover are maintained in hay fields or pastures, excessive soil erosion will be avoided (Gilley et al., 1996).
In landscapes with permanent vegetative cover, such as forest and pasture, the primary source of sediment is not
from surface losses, but from erosion of the stream bank (Sharpley et al., 1993; Daniel et al., 1994).

Once a sediment particle has been dislodged from the soil surface, the distance it moves downslope is
related to the percentage of vegetative surface cover, amount and type of aboveground biomass, surface
roughness coefficient of the soil, and the size of the individual soil particles (Pearce et al., 1998b). Decreasing
surface cover exposed the sediment to greater raindrop impact, increasing sand movement through splash
erosion. A reduction in surface roughness resulted in increased sediment runoff because of a lack of surface
depressions to catch sediment particles. Larger particles do not move as far downslope as do smaller particles
(Pearce et al., 1998b). Most sediment deposition from runoff water to the surface of pastures occurred in the
first meter downslope of application. When simulated rainfall was applied to a pasture at 100 mm·hr⁻¹ and
simulated runoff was applied at 25 mm·hr⁻¹, nearly 90% of the sediment was filtered in the first 10 m in plots
that were either ungrazed or mowed to 10 cm, 84% of the sediment was filtered in the first 10 m in a plot that was trampled, but not grazed by cattle, and 77% of the sediment was filtered within 10 m when cattle were allowed to both trample and graze the plot (McEldowney et al., 2002). Using regression analysis, Pearce et al. (1998a) were not able to produce a model for the accurate prediction of sediment yield based on vegetative characteristics of the site including vegetative cover, vegetative density, litter cover, and species composition.

One difficulty in predicting sediment erosion from a site is that a few large rainstorms are often responsible for the majority of the erosion that occurs in a given location. Daniels and Gilliam (1996) reported that a single storm was responsible for 60 to 70% of the sediment erosion that occurred over a 2-year study period to evaluate the effectiveness of vegetative buffers in the control of sediment loss. Similarly, Moir et al. (2000) reported that much of the sediment loss that occurred in a mountain grassland occurred during the early portion of a multi-year study.

**Surface Slope.** It has been widely demonstrated that as the slope of the land increases, so does the rate of sediment loss (Liu et al., 1994; Bradford and Foster, 1996; Mwendera and Saleem, 1997; Moir et al., 2000; Russell et al., 2001). In a rainfall simulation study on a packed clay loam soil, sediment delivery increased as slope increased from 5 to 10%, but not when it increased from 10 to 15% (Huang, 1998). The author stated that these results indicate that as slope increased from 5 to 10%, sediment loss was limited by the transport capabilities of the runoff. But as slope increased from 10 to 15%, sediment transport was limited by detachment of sediment particles (Huang, 1998).

**Phosphorus Cycling in a Pasture System**

Smil (2000) presented the P cycle on a global scale; this discussion will focus specifically on the cycling within a pasture system. Three main P pools exist in the P cycle of a pasture system, including the animal, the soil, and the forage. However, many additional inputs and outputs exist (Figure 1). Approximately 1.0% of the live weight of a beef cow is P; a 550 kg animal, therefore, contains 5.5 kg of P in her body (Armsby and Moulton, 1925). This same animal requires 13 g P per day to support her maintenance and additional P to support growth, gestation, and lactation, depending on her stage of production (NRC, 1996). Much of this P can be provided by forages growing in the pasture. Under some conditions, additional P supplementation will be needed to support adequate growth and production by the animal or P may be provided in energy or protein.
supplements fed to the animals (Karn, 2001), resulting in an input to the pasture P cycle. Phosphorus in the ‘cow P pool’ can exit the pasture system through in the milk, meat, and calves of grazing cows (Watson and Foy, 2001).

Figure 1. The Phosphorus cycle in a pasture system.

Cattle excrete approximately 60 to 80% (Berry et al., 2001; Knowlton et al., 2004) of dietary P to pasture in their feces and urine, primarily in the feces (Berry et al., 2001). Annual P deposition on pastures by grazing livestock can range from 1 to 12 kg ha⁻¹, depending on stocking rate and intake (Smil, 2000). As feces
decompose, P enters into the soil P pool (Williams and Haynes, 1993; Aarons et al., 2004b). In addition to P deposited by grazing animals, P can be introduced to pasture soil from the application of organic and inorganic fertilizers (Slaton et al., 2004) and from precipitation or dry atmospheric deposition associated with dust from wind erosion (Smil, 2000). In a study of soil P distribution and fertilizer application to agricultural land in Arkansas, Slaton et al. (2004) reported that 66 to 82% of the P applied to agricultural land in regions dominated by pastures was organic P, primarily as poultry litter and swine manure. In temperate zones, rainfall contains between 0.01 and 0.06 mg P·L⁻¹, resulting in 0.5 to 0.7 kg P·ha⁻¹ deposition to pastures per year in rainfall (Smil, 2000).

Phosphorus in soil occurs naturally at levels between 300 and 1200 mg·kg⁻¹ (Daniel et al., 1994). Phosphorus in the soil pool exists in several organic and inorganic forms, the amount of each form of P and the rate of P cycling between these different forms is controlled by the presence of other minerals in the soil, soil pH, environmental factors, and biotic factors in the soil (Smeck, 1985; Smil, 2000). Smeck (1985) reviewed the abiotic and biotic processes that regulate P cycling within the soil pool. Losses of P from pasture systems occurs bound to sediment particles in eroding material, in a soluble form in surface runoff, and by leaching (Smil, 2000; Havlin, 2004). These losses of P from the system are natural processes, but can be accelerated by management practices administered to the land (Smil, 2000). In a pasture system, these P losses are generally small resulting in the accumulation of applied P in the soil (Watson and Foy, 2001).

A portion, generally less than 10%, of the total soil P pool is available for uptake by forages to support plant growth at any time (Smeck, 1985; Daniel et al., 1994). Typical cool-season grass and legume forages contain 0.22 to 0.44% P (Pierzynski and Logan, 1993). Phosphorus within forage can enter a fast cycle, being consumed by the grazing animal, or a slow cycle, being returned to the soil during the decomposition of dead plant material, both of these pathways will maintain the P in the cycle (Bakker et al., 2004). Forage can also be harvested as hay, removing 13 to 35 pounds P per acre from the pasture system per year (Pierzynski and Logan, 1993).

**Controlling P loss from Pastures**

Three general principles must be considered in the control of P losses from pastures or other land management practices. Phosphorus movement in the landscape follows the direction of surface and subsurface water flow
(Smeck, 1985). Near streams, soil P concentration controls P export from a watershed (Gburek et al., 2000). Seasonal differences exist in the concentration and mass of P lost to surface waters between watersheds and seasonal fluctuations in P movement (Sharpley et al., 1993).

**Phosphorus Indices.** The P index was originally developed to provide a P assessment tool for estimating the risk that exists for P leaving a landform site and traveling toward a water body, to aid in identifying the critical parameters that most strongly influence P load, and to select management practices that would significantly decrease the site’s vulnerability to P loss (Lemunyon and Gilbert, 1993). The original P index considered several site characteristics and assigned a weighting factor to each characteristic depending on its potential to allow P movement from the site to surface waters. Site characteristics included a soil erosion factor, an irrigation erosion factor, runoff class, soil test P level, P fertilizer application rate and application method, and organic P application rate and application method (Lemunyon and Gilbert, 1993). These eight characteristics in the original P index were chosen because the required inputs could easily be obtained and because of their impact on P movement (Stevens et al., 1993). Based on the assessment characteristics and weighting factors, a site’s vulnerability number was determined. Sites with values less than 8 were considered low risk for P transport, 8 to 14 was considered medium risk, 15 to 32 was considered high risk, and greater than 32 was a very high risk area (Lemunyon and Gilbert, 1993). Management practices for the control of P losses to surface waters were not included in this early version of the P index (Lemunyon and Gilbert, 1993).

Using the P index to determine the potential for P transport from fields in an Oregon watershed, Stevens et al. (1993) determined that the P index was sensitive to nutrient management factors, such as soil test P concentrations and P fertilization application rates and methods in differentiating sites according to potential P loss. It was less sensitive to differences in erosion and runoff related site characteristics. Stevens et al. (1993) concluded that the P index was a useful tool to assess sites for potential harm to surface water quality under local conditions in the study area.

The original P index was meant to be modified so that it could be used under different conditions. These modifications were based on the condition of local water bodies, specific soil characteristics, climatic conditions, and management practices in a given watershed (Stevens et al., 1993). Examples of these modified P indices include the Iowa P index (USDA, 2004) and the Arkansas P index for pastures (Delaune et al., 2004).
The Iowa P index has been adapted to fit local conditions using regional and in-state research to assess the risk of P delivery to surface waters. The P index is meant to be used by the USDA-NRCS in nutrient management planning for land users and resource planners. It is not intended to be an evaluation scale for determining whether land users are complying with water quality or nutrient management standards established by local, state, or federal agencies. The Iowa P index uses source and transport factors to approximate P loads contributed to surface waters from a field, expressed in lbs·ac⁻¹·yr⁻¹, and group each field into one of five risk classes (very low, low, medium, high, very high) based on its potential to contribute P to water source (USDA, 2004). The Iowa P index is calculated as:

- P index = erosion component + runoff component + subsurface drainage component
  - Erosion component = gross erosion x (sediment trap factor or sediment delivery ratio) x buffer factor x enrichment factor x soil test P erosion factor
  - Runoff component = runoff factor x (soil test P runoff factor + P application factor)
  - Subsurface Drainage component = precipitation x flow factor x soil test P drainage factor

Using the Iowa P index, a value from 0 to 15 is calculated and classified as being very low, low, medium, high, and very high depending on its potential to contribute P to surface waters. An index value of 0 to 1 indicates very low risk and is associated with fields in which, if soil conservation and P management practices are maintained, impacts to surface water resources from P losses from fields will be small. An index value of 1 to 2 indicates a low risk and is associated with fields in which current soil conservation and P management practices will keep the risk of water quality impairment low. An index value of 2 to 5 indicates a medium risk to surface waters and occurs in fields in which P delivery potential to waters may produce some water quality impairment. In fields with a medium risk, consideration should be given regarding the implementation of further soil conservation and P management practices that do not increase P delivery to surface water. An index value of 5 to 15 indicates a high risk of P contamination of surface waters and the requirement of remedial actions to reduce P movement to surface waters. New soil and water conservation and/or P management practices are necessary to reduce offsite P movement and water quality degradation in these high-risk sites. An index value of greater than 15 indicates a very high risk to surface waters and occurs in fields in which the risk of impacts on surface water resources are extreme. Remedial actions are required to
reduce P delivery to surface water from fields in the very high-risk group. These actions would include all necessary soil and water conservation practices and a P management plan to reduce water quality impairment under this condition (USDA, 2004).

A P index has been created in the state of Arkansas specifically for the development of manure management plans for the safe application of poultry litter to pastures (Delaune et al., 2004). The index is unitless and is calculated as:

\[
\text{P index} = \text{P source} \times \text{P transport} \times \text{precipitation} \times \text{best management practice}
\]

The P index for pastures provides a simple assessment tool with readily available input parameters that can easily be used by nutrient management planners to determine the potential risk to surface waters. Pastures are classified as low (less than 0.6), medium (0.6 to 1.2), high (1.2 to 1.8), or very high (greater than 1.8). At a low or medium P index, manure application to pastures is based on N requirements of the pasture, while at high or very high P index values, manure application is based on the rate of P removal in forage (Delaune et al., 2004).

Delaune et al. (2004) conducted a rainfall simulation study to evaluate the role of poultry litter in contributing to P movement in pastures. Prior to the application of poultry litter to a pasture, average soluble reactive P concentrations in the runoff water were significantly correlated \((R^2 = 0.61)\) with soil test P concentrations. After poultry litter application, soluble reactive P load in surface runoff were poorly correlated \((R^2 = 0.051)\) with soil test P, but was more highly correlated \((R^2 = 0.66)\) with the amount of soluble P applied in the litter.

In a separate study using two small watersheds, measured P losses and P index values were highly correlated \((R^2 = 0.83)\) and the P content in surface runoff was more closely related to P index value than to soil test P concentration in pastures treated with poultry litter. These results indicated that the P index value was an acceptable estimate of annual P loss from pastures receiving litter application (Delaune et al., 2004).

**Riparian Buffers.** The quantity of sediment P delivered to and beyond a field’s edge is a function of the soil erosion rate, the amount of sediment deposited within the field, and the quantity of P adsorbed to the
eroding soil particles. Riparian buffers are one method of controlling sediment and P from entering surface waters once they have passed beyond the field edge. Unlike sediment-bound P, there are few conservation practices that can reduce soluble P from reaching a stream’s edge. However, management practices which increase infiltration of runoff can reduce the total load of soluble P reaching surface water sources (Havlin, 2004).

Riparian buffers are an effective management tool for the physical removal of sediment from surface runoff to prevent its transport to surface waters (Nash and Halliwell, 1999; Lee et al., 2000). Over a twenty-year period, greater than 50% of the sediment displaced from a crop field was deposited within a grassy area within 100 m of the field edge (Cooper et al., 1987). Because P loss from pastures is primarily in the soluble forms, Nash and Halliwell (1999) proposed that riparian buffers might not be effective at controlling P loss from pastures. Lim et al. (1998) found that approximately 75% of both the total P and soluble P loads in runoff from a pasture were removed with a 6.1 m vegetative filter strip when artificial rainfall was applied to an area treated with cattle manure. This result implied that buffers are able to filter soluble P as well as particle-bound P in precipitation runoff.

Even though riparian buffers can act as traps for particulate-bound P, microbial processes within the buffer zone can release nutrients previously trapped in the buffer, enriching runoff water with soluble P, which is more biologically available to aquatic plant life (Dillaha et al., 1989; Cooper et al., 1995). Uusi-Kamppa et al. (2000) reported that a grass buffer site that was harvested had a 14% lower soluble P load leaving the buffer while an unharvested site had a 64% greater soluble P load in the runoff leaving the buffer compared to runoff entering the buffer. Riparian buffers should not be considered a bottomless sink for high nutrient inputs. If long-term water quality benefits are to occur, riparian buffers should be accompanied by improved land use practices over the larger landscape with periodic harvest from set aside used to maintain nutrient cycling (Cooper et al., 1995; Uusi-Kamppa et al., 2000).

**Controlling P Loss with Pasture Management.** Controlling sediment (and nutrient) loss from grasslands cannot be achieved by managing for a single factor such as vegetative cover (Moir et al., 2000). Several factors interact in the loss of sediment (and nutrients) from pastures during and after storm events. Included in this list are surface cover, slope gradient, ground surface roughness, soil depth, and soil infiltration
rate (Moir et al., 2000). Proximity to a water body (Gburek et al., 2000), season of the year (Elliott et al., 2002), and soil P levels are also important (Cornish et al., 2002) to consider in the control of sediment and P losses from pastures.

Management of Critical Areas. Phosphorus transport in surface runoff generally occurs from well-defined areas within a watershed, usually within 60 m of a stream or water body (Sharpley et al., 1999). These critical areas should be the focus of management practices meant to control P contamination of surface waters (Daniel et al., 1994). Management practices meant to minimize P losses must reduce surface runoff and particulate transport, particularly in areas that have the most potential to be major contributors of P (Gburek et al., 2000). For example, paddocks lower on the slope of a hill have greater potential to transport P to surface waters because of their closer proximity to water bodies (Cornish et al., 2002). Congregation areas, such as shade, water, and supplement feeders can be sites for nutrient concentration (Mathews et al., 1994) and, if these areas are near surface waters, they will have a greater potential for contributing sediment and nutrients to surface waters. Not only do nutrients build up in congregation areas, but these areas also have greater soil bulk density and a lower infiltration rate for precipitation than the remainder of a pasture. Mulholland and Fullen (1991) observed that soil bulk density was greater in a congregation area (1.48 g·cm⁻³) in a pasture than in the remainder of the pasture (1.22 g·cm⁻³). Infiltration rate followed an opposite trend being lower in the congregation area (1.06 mm·hr⁻¹) than in the rest of the pasture (72.95 mm·hr⁻¹). Areas of high slope are also likely to have higher soil erosion rates and greater surface runoff than low slope areas (Shainberg et al., 1992).

Several methods exist for managing these critical areas for sediment and nutrient loss in runoff. One method that has been proposed is the use of a fence to restrict cattle access to water bodies and riparian areas (Bryant, 1982; Owens et al., 1996). This solution creates traps for particulate P that is digested by microbial processes enriching runoff water with soluble P that is more biologically available to aquatic plant life (Dillaha et al., 1989; Cooper et al., 1995). Phosphorus from decaying leaf litter in these areas may also contribute nutrients to surface waters. The use of rotational stocking rather than continuous stocking can reduce the concentration of nutrients at rest areas or near water sources (Williams and Haynes, 1993). If the riparian area is included as a paddock in a rotational grazing system, it can be grazed for short periods by cattle at times when it is less likely to contribute nutrients and sediment to surface waters (Clary and Leininger, 2000). The use of
portable shade, water, molasses supplements, and mineral sources within paddocks may also encourage animals to not congregate in a single area, thereby preventing the concentration of nutrients and the reduction in water infiltration rates typical of these areas (Godwin and Miner, 1996; Porath et al., 2002; Bailey, 2005; Launchbaugh and Howery, 2005). Regular soil P tests are also important in the control of P losses (Daniel et al., 1994).

Soil Phosphorus Levels. One way to control P load in surface runoff is by focusing on the control of soil P concentrations and management of fertilizer and manure applications in areas most likely to produce surface runoff (Gburek et al., 2000). As mentioned previously (see section on soil test P), several laboratory tests exist for the determination of plant available soil P, but these tests do not necessarily indicate the potential for P to be lost in surface runoff (Kleinman et al., 2000). Although soil test P level is not considered to be a good indicator of P losses to surface waters, Cornish et al. (2002) observed a trend for increased total and soluble P concentrations in surface runoff as soil Bray-1 P increased and P sorption to the soil decreased. Additionally, once soil test P levels exceed plant requirements, no additional benefit from greater P application exists for the producer (Nelson, 1999; Pautler and Sims, 2000).

Surface Cover. The proportion of bare ground can be minimized in pastures by using low stocking rates or properly managed rotational stocking systems (Manley et al., 1997; Mwendera and Saleem, 1997). Even at 100% surface cover, some erosion and nutrient transport in surface runoff will take place in pastures (Elliott et al. 2002). Elliot and Carlson (2004) observed that as the amount of surface cover in a pasture decreased, the concentrations of sediment, total P, and total Kjeldahl N in runoff from pastures increased linearly. Over-stocking of livestock on pastures, as affected by location and climatic conditions, can result in a decrease in surface cover. Mwendera et al. (1997a) reported that the percentage of vegetative surface cover decreased from 100% to 80% and erosion rate increased as the stocking rate increased from 0 to 4.2 AUM·ha⁻¹ on pastures (Mwendera and Saleem, 1997). Converting from a continuous stocking system to a rotational stocking system at a constant stocking rate can prevent patches of bare ground from developing within a pasture. Manley et al. (1997) observed that at a stocking rate at 62.7 steer-days·ha⁻¹, surface cover was lower in pastures grazed by a season long continuous stocking system than in either a 4-paddock rotationally deferred or 8-paddock time-controlled rotational stocking system over 12 years under rangeland conditions.
Forage Sward Height. Forage sward height is easier to measure and communicate to land managers than is forage utilization and other pasture measures, making stubble height a worthwhile measure (Clary and Leininger, 2000). Nutrient concentrations in surface runoff from pasture tend to be inversely related to grass length (McCull and Gibson, 1979). Clary and Leininger, (2000) recommended a 10 cm residual stubble height as a starting point for improved riparian grazing management but stated that monitoring of the pasture condition should be conducted to determine if forage sward height should be maintained at greater or lesser sward heights. Average forage height of a pasture is a common measure used in determining the impact grazing has had on a pasture (Turner and Clary, 2001) and it plays an important role in the control of intake by grazing animals (Barrett et al., 2001; Tharmaraj et al., 2003). In addition to the importance of maintaining adequate forage sward height in the control of forage intake, maintaining adequate sward height helps preserve forage plant vigor, stabilize sediment, limit stream-bank trampling, maintain cattle gains, and provide an easily communicated management criterion (Clary and Leininger, 2000).

Infiltration Rate. Phosphorus losses from pastures can be reduced by management practices that slow or reduce the total volume of surface runoff and/or encourage infiltration or sediment trapping (Guberek et al., 2000). Singleton and Addison (1999) reported an 80% reduction in hydraulic conductivity of soil as a result of pugging damage caused by grazing of wet soils. Reduced hydraulic conductivity leads to a reduction in infiltration rate and a corresponding increase in surface runoff and sediment and nutrient transport. Even when pugging does not occur or pastures are grazed at low stocking rates, infiltration rate on grazed pastures will be less than that on ungrazed grasslands (Naeth et al., 1990b).

High stocking rates have been shown to have greater effects on soil compaction and infiltration rates in pastures than low stocking rates (Teerdoff et al., 1999; Donker et al., 2002). The greater soil compaction in pastures grazed at heavy stocking rates was likely related to greater surface area trampled and greater forage removal, reducing the cushioning effect of the forage (Naeth et al., 1990a). The act of grazing can affect litter decomposition rate and organic matter mass in a pasture by the direct consumption of forage by the grazing animal and by the actions of hoof traffic degrading leaf litter. Naeth et al. (1991b) reported a reduction in coarse, fine and very fine litter with increasing grazing intensity.
Mapfumo et al. (2000) reported a reduction in soil water-holding capacity at heavy stocking rates after 3 years, while light and medium stocking rates resulted in a slight increase in soil water-holding capacity over the same period. Grazing methods that allow litter to accumulate on the soil surface (Naeth et al., 1991b) and limit grazing on wet soils (Naeth et al., 1990b) can limit reductions in soil infiltration rate and may even increase infiltration. The use of longer rest periods between grazing periods in a rotational stocking system allows the pasture to recover and decreases soil penetration resistance (Rodd et al., 1999) while increasing infiltration rate. During wet periods of the year, it may also be beneficial to designate one paddock for grazing to protect other paddocks from being damaged (Greenwood and McKenzie, 2001).

Seasonal Considerations. The condition of the pasture and its propensity for runoff and sediment and nutrient loss changes with season. Elliott et al. (2002) reported seasonal variability in the amount of sediment eroded from a pasture, with peak erosion occurring during the wetter portion of the year (Elliott et al., 2002). Infiltration rate, during a rainfall simulation study, was higher in summer (dry season) than in winter (wet season) in a New Zealand pasture stocked by sheep (Elliot and Carlson, 2004). Naeth et al. (1990b) reported that heavy stocking rates in the early spring (wetter period of the year) resulted in the greater reductions in infiltration rate than grazing in other seasons. Not only does runoff increase during wetter periods of the year as a result of soil saturation (Trimble and Mendel, 1995; Elliott et al., 2002), but the saturated soils are also more fragile and likely to be damaged by the presence of grazing animals (Greenwood and McKenzie, 2001). During wet periods, it is advantageous to use low stocking densities and short periods of occupation on a given paddock so that adequate forage cover is maintained to reduce runoff. The use of a sacrificial paddock may also be advised, so that damage to moist soils can be limited (Greenwood and McKenzie, 2001).

LITERATURE CITED


CHAPTER 3. GRAZING MANAGEMENT EFFECTS ON SEDIMENT AND PHOSPHORUS IN SURFACE RUNOFF

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ABSTRACT

Sediment and phosphorus (P) in runoff from pastures are potential non-point source pollutants in surface waters. Sediment and P loads in runoff from pastures may be influenced by surface cover, sward height, treading damage, surface slope, soil moisture, and soil P. The objectives of the current study were to quantify sediment, total P, and soluble P loads in runoff produced during simulated rainfall from pastures and evaluate their relationships with the physical and chemical characteristics of the soil and sward. Five forage management treatments; ungrazed (U), hay harvest/fall stockpile grazing (HS), continuous stocking to a forage height of 5 cm (5C), and rotational stocking to forage heights of 10 cm (10R) or 5 cm (5R), were established in 0.40-ha paddocks in three smooth bromegrass (Bromus inermis Leyss.) pastures and maintained for three years. Rainfall simulations were conducted at a rainfall intensity of 7.1 cm·hr⁻¹ for 1.5 hours over a 0.5-m² area in three locations at two slope ranges in each paddock in June, August, and October of each year and the subsequent April. Forage management did not affect mean sediment load (7.3 ± 5.0 kg·ha⁻¹·hr⁻¹). Mean total P load was greatest from 5C treatment (0.071 ± 0.011 kg·ha⁻¹·hr⁻¹), did not differ between U, HS, and 10R treatments (0.019 ± 0.011 kg·ha⁻¹·hr⁻¹) and was intermediate in 5R (0.053 ± 0.011 kg·ha⁻¹·hr⁻¹). Mean soluble P load was greatest from 5R and 5C treatments (0.037 ± 0.004 kg·ha⁻¹·hr⁻¹) and did not differ between U, HS, and 10R (0.011 ± 0.004 kg·ha⁻¹·hr⁻¹) treatments. Of the soil and sward characteristics measured, percentage surface cover was most highly related to sediment load ($R^2 = 0.16$) and total P load ($R^2 = 0.10$). Surface runoff from pastures managed to maintain adequate residual forage cover did not contribute greater sediment or P to surface waters than an ungrazed grassland.

Key words: cattle, non-point source pollution, nutrients, rainfall simulation, water quality
INTRODUCTION

The erosion of sediment and transport of nutrients in surface runoff are natural processes that can be accelerated by land management practices (Smeck, 1985; Trimble and Mendel, 1995). The amounts of sediment and phosphorus (P) in surface runoff from agricultural lands are of concern because of the potential they pose for contributing to siltation and eutrophication of surface waters (CAST, 2002). Phosphorus is usually considered to be the limiting nutrient controlling the eutrophication of fresh water lakes and streams (Correll, 1998).

Soil losses from forage systems are generally considered to be low (Gard et al., 1943). Forage production may reduce sediment and phosphorus loading of pasture streams through improved soil structure related to higher soil carbon and nitrogen (Entz et al., 2002), increased soil microporosity (McDowell et al., 2003), increased rainfall infiltration (Alderfer and Robinson, 1947), protection of the soil surface from the force of raindrop impact (Pearce et al., 1998), filtration of sediment particles in surface runoff that have dislodged from other locations (Pearce et al., 1998), and fragmentation of compacted soil layers by forage roots, which enhances water and air infiltration into the soil (Hubbard et al., 2004). Despite the potential benefits associated with forage production systems, reports from the U.S. (CAST, 2002) and New Zealand (Gillingham and Thorrold, 2000) have implicated livestock grazing in the degradation of surface water quality.

Several factors related to forage, soil, and site characteristics, and environmental conditions influence the amounts of sediment and P that can potentially be dislodged from a pasture and transported by surface runoff, contributing to a reduction in water quality. These factors include the amount of ground cover (Thurow et al., 1988), treading damage (Nguyen et al., 1998), slope steepness (Shainberg et al., 1992) and length (McDowell and Sharpley, 2002), antecedent soil moisture (Merz and Plate, 1997), and soil P content (Sharpley et al., 1996). Precipitation rate (Sharpley, 1985; Huang, 1998) and seasonal variability (Sharpley et al., 1993; Elliott et al., 2002) can also influence surface runoff.

Forage on the soil surface provides protection to the soil from the impact of raindrops, cushions the soil to resist compaction and preserves aggregate stability of the soil under grazing conditions (Naeth et al., 1990; Pearce et al., 1998; Greenwood and McKenzie, 2001). Grazing can increase litter decomposition rate and decrease organic matter mass in a pasture through direct consumption of forage by the grazing animal and by the actions of hoof traffic degrading leaf litter (Naeth et al., 1991b). A reduction in leaf litter can result in a
reduction of aboveground water-holding capacity (Naeth et al., 1991a). A reduction in soil organic matter results in poorer soil structure and a reduction in water holding capacity of the soil (Naeth et al., 1991b; Betteridge et al., 1999).

Proper management of grazing lands to enhance vegetative cover helps to hold soil in place, filter water, and recycle nutrients (CAST, 2002; Hubbard et al., 2004), resulting in improved water quality. Maintaining optimum pasture cover and forage sward height and allowing litter to accumulate on the soil surface can preserve forage plant vigor, improve soil structure, stabilize sediment, and reduce the movement of nutrients from pastures into streams (Naeth et al., 1991b; Nelson et al., 1996; Clary and Leininger, 2000). The use of rotational stocking systems and hay harvest have been shown to reduce the amount of bare ground and maintain an adequate forage canopy in pastures to prevent excessive soil erosion and nutrient transport to surface waters (Gilley et al., 1996; Manley et al., 1997; Mwendera and Saleem, 1997; Hubbard et al., 2004). Clary and Leininger, (2000) recommended a 10 cm residual stubble height as a starting point for improved riparian grazing management, but stated that the pasture condition should be monitored to determine if forage sward height should be maintained at greater or lesser sward heights. The objectives of the current study were to quantify the losses of sediment, total P, and total soluble P in surface runoff produced by simulated rainfall in pastures with different management practices and to determine the relationships among the soil and forage characteristics of the pastures with those losses.

MATERIALS AND METHODS

Site Description
For 3 years, 2001 - 2003, pastures located at the Iowa State University Rhodes Research and Demonstration Farm (lat 42°00' N, long 93°25' W) were managed to determine the impacts of beef cow grazing on sediment and P losses in surface runoff from pastures. Pastures had slopes of 0° to 15° and were primarily composed of smooth bromegrass (Bromus inermis Leyss). Soils at the study site were characterized as Downs silt loam (fine-silty, mixed, mesic Mollic Hapludalf), Gara loam (fine-loamy, mixed, mesic Mollic Hapludalf), and Colo-Ely complex (fine-silty, mixed, mesic, Cumulic Haplaquoll, and fine-silty, mixed, mesic, Cumulic Hapludoll). Thirty-year average annual precipitation in the area was 891 mm, with the first (932 mm) and third year (965 mm) of the study being slightly above and the second year (716 mm) slightly below the 30-year average (Fig. 1;
NOAA, 2001, 2002, 2003, 2004). Prior to initiation of the study, all pastures were managed as a single unit for grazing of beef cattle and hay harvest.

Three pastures of approximately 2.2 ha were located on three hillsides with a north, south or east aspect and subdivided into five 0.4-ha paddocks with a 6-m wide lane at the top of the hill for cattle movement. Soil samples were taken prior to the initiation of stocking in April 2001 for determination of plant available (Bray-I) P and potassium (K) by the Iowa State University Soil Testing Laboratory. In April 2001, P was applied as diammonium phosphate to two paddocks in one pasture to bring all paddocks to at least the level of 11 to 15 mg·kg⁻¹ considered optimum for cool-season grass pastures in Iowa (ISU, 2002). Soils in all paddocks contained an optimum level of 111 to 150 mg·kg⁻¹ or greater of K; therefore, no additional K was applied. Neither P nor K was applied for the remainder of the study period. In all three years, N was applied to all paddocks as urea at a rate of 200 kg urea·ha⁻¹ before the initiation of grazing in the spring and at 112 kg urea·ha⁻¹ in August. Sandbags were placed around the perimeter of the pastures and adjacent paddocks to prevent contamination from surface runoff during natural rainfall events from outside the experimental area and between adjacent paddocks.

**Forage Management**

Forage management treatments were randomly assigned to one of 5 paddocks in each pasture. Treatments included: an ungrazed control (U), summer hay harvest with fall stockpile grazing to a residual sward height of 5 cm (HS), continuous stocking to a residual sward height of 5 cm (5C), and rotational stocking to a residual sward height of 5 cm (5R) or 10 cm (10R). Paddocks were initially stocked with 3 nonpregnant mature Angus cows (mean body weights of 657±84, 613±94, and 625±53 kg in year 1, 2, and 3, respectively) in May of all three years (Table 1). Animals had access to salt, but received no supplemental P while on the research pastures.

In the continuous stocking system, cattle were removed from the paddocks when the sward height, measured with a falling plate meter (4.8 kg·m⁻²; Hermann et al., 2002), decreased to 5 cm. Paddocks were allowed a short rest period of 7 to 10 days to allow limited regrowth. These short rest periods in the continuous stocking system were considered to be representative of cattle distribution patterns in a larger pasture. Cattle allowed continuous access to a large pasture will avoid areas of low forage availability in favor of areas of
greater forage availability (Pinchak et al., 1991), effectively providing the area of low forage availability a short rest period for forage regrowth to occur. In the rotational stocking systems, cattle were removed from the paddocks when the sward height decreased to 5 or 10 cm for the 5R and 10R treatments, respectively. In both rotational stocking systems, paddocks were allowed 35-day rest periods for plant regrowth before being restocked.

Hay was harvested from the HS treatment in June of each year (Table 2). Regrowth from these paddocks was mowed in early August of each year to improve forage quality during the stockpile-grazing period (Fribourg and Bell, 1984), but the yield of clipped forage was inadequate to harvest. Paddocks in the HS system were stocked in mid-November of each year, following a killing frost, with 3 cows that had been used during the previous summer grazing period and grazed to a residual sward height of 5 cm.

Rainfall Simulations
Rainfall simulations were conducted in the late spring, mid-summer, and autumn of each year and in the early spring the following year to determine infiltration rate, percentage of precipitation lost as surface runoff, and amounts of sediment and P lost in surface runoff. Six simulation sites were selected within each paddock so that three in a low (1° to 7°) slope area and three were in a high slope area (7° to 15°). Rainfall simulation sites were marked with fiberglass posts so that the same locations were measured during each sampling period during the three years of the study. Rainfall simulations were conducted with drip-type simulators measuring 1 x 0.5 m (Bowyer-Bower and Burt, 1989) onto plots running with the hill slope. Simulators were assembled so that the uphill side of the simulator was 1 m above the soil surface, allowing simulated rainfall to reach 56% of terminal velocity (Gunn and Kinzer, 1949). Each rainfall simulation ran for 1.5 hours at a precipitation rate of 7.1 cm·hr⁻¹, corresponding to a storm event with a 50-year recurrence interval (Huff and Angel, 1992). Municipal water that had been filtered through a 0.45 µm filter to remove particulate matter, (pH 7.89, electrical conductivity 316 µS) was used as the source water for rainfall simulations. During simulations, amounts of rainfall and runoff were measured at 10-minute intervals and runoff was composited by simulation site over the simulation period. Rainfall infiltration rate was calculated as the volume of rainfall applied minus the runoff volume divided by the area of the simulation site per hour and percent surface runoff was calculated as the volume of surface runoff collected divided by the volume of rainfall applied over the 90 minute rainfall simulation period.
multiplied by 100. At the completion of rainfall simulation, collection tanks were agitated and a 1 L subsample was retained for analysis.

Prior to rainfall simulations, surface roughness was measured by digital photography of a 41-pin meter with a length of 2 m and calculated as the standard deviation of the length of adjacent pins on the pin meter (Betteridge et al., 1999) determined by image analysis using SigmaScan Pro 5.0 software (SPSS inc., Chicago, IL). Vegetative ground cover was determined as one minus the percentage of pins on the pin meter striking soil at each site (Betteridge et al., 1999). During simulations, soil samples were taken adjacent to each simulation site at depths of 0 to 5 cm for determination of Bray-1 P and antecedent soil moisture. Penetration resistance was measured adjacent to the simulation site at 3.5 cm intervals to a depth of 35 cm using a Bush Recording Penetrometer (Findley, Irvine, Midlothian, Scotland) with a 12.9 mm diameter cone. When penetration resistances exceeded a value of 4572 kPa; the maximum resistance that the instrument could measure, this value was used for missing values for statistical analysis. Sward height was measured with a falling plate meter (4.8 kg·m·2, Hermann et al., 2002), and a forage sample was clipped to a height of 2.5 cm from a 0.25-m2 area adjacent to the rainfall simulation site, with the same sward height as the simulation site.

Laboratory Analysis

Water samples were stored at 4°C until analysis for sediment, total P, and total soluble P. Sediment and P concentrations in the input water were subtracted from the runoff samples. Sediment was determined by filtering a 100 ml water sample through a pre-weighed 0.45 µm filter paper (Fisher Scientific, Pittsburg, PA). The filter paper was oven-dried (APHA, 1995) at 100°C for 24 hrs and weighed. Total P concentration was determined by digestion, followed by colorimetric analysis with the ascorbic acid method (Hach Company, Loveland, CO). Total soluble P concentration was determined by filtering the runoff samples through 0.45 µm filter paper (Fisher Scientific) followed by digestion and colorimetric analysis with the ascorbic acid method (Hach Company). Sediment, total P, and total soluble P loads, kg·ha⁻¹·hr⁻¹, were calculated by multiplying the concentration of each component by the total runoff volume from each rainfall simulation area.

Soil P concentrations were determined using the Bray-1 P procedure (Bray and Kurtz, 1945). Gravimetric soil moisture was determined by drying samples at 105°C for 24-hours and weighing. Forage samples were oven-dried at 65°C for 48 hrs and weighed to determine forage mass.
Statistical Analysis

Data were analyzed using the PROC MIXED procedure of SAS (SAS, 2001). The model included the fixed effects of treatment, slope, year, season, and their interactions and replicate as a random effect. Because of the non-normal distribution for sediment load, total P load, and soluble P load, a randomization test was conducted to determine statistical significance. Least square means are reported in text and tables. Significance was determined at $P < 0.05$ and a tendency for significance was determined at $P < 0.10$.

Stepwise multiple regressions were performed using the PROC REG procedure of SAS (SAS, 2001) to determine the relationship between the measured forage (sward height, surface cover, forage mass), soil (soil moisture, Bray-1 P, penetration resistance at 3.5 cm intervals to a depth of 35 cm), and site characteristics (surface slope) and the dependent variables of infiltration rate, surface runoff, sediment load, total P load, and soluble P load. Slope was included in the data set as the sine of slope in radians because sine of slope provides a more accurate representation of the flow shear stress of runoff water (Liu et al., 1994). Variables not significant at $P < 0.15$ were excluded from the model.

RESULTS

Year effects

Infiltration rate was greater ($P < 0.05$) and the percentage of runoff was lower ($P < 0.05$) in year two than in either year 1 or 3 (Table 3). These yearly differences in infiltration rate and surface runoff were likely related to variations in precipitation patterns, as opposed to changes in soil condition. Changes in soil characteristics that have an impact on hydrological properties of the soil may take longer to appear than the three years of the current study (Mapfumo et al., 2000). Concentrations of sediment and total and soluble P and sediment load in surface runoff did not differ by year. The greater infiltration rate in year 2 resulted in less surface runoff and a corresponding reduction in total and soluble P movement during rainfall simulations.

Across treatments, Bray-1 P in the soil tended to decrease ($P = 0.06$) over the three years of the study (Table 4). The lack of a year by grazing management treatment interaction seems to indicate that this decrease may not be the result of an actual change in soil P, as no change in soil P was detected when only the Bray-1 P levels from beginning and end of the study were included in the data set for analysis. Soil moisture was greatest during the first year, decreased in the second year, and decreased further in the third year of the study ($P <$
0.05). Even though a greater amount of precipitation fell during year 3 than years 1 and 2, the low soil moisture during this year can be explained by the periods in which rainfall simulations were conducted having below average precipitation while nearly three times the average precipitation fell during November. Penetration resistance was lowest in year 1 (P < 0.05) and greatest in year 2 at all depths; with year 3 being intermediate at most depths. Surface roughness was greatest during the first year of the study (P < 0.05) with no differences between the second and third years.

Mean forage sward height was lower (P < 0.05) in the first year (9.3 cm) than in the second and third years (10.9 and 10.8 cm). Mean forage mass was lower (P < 0.05) in the first and second years (2003 and 2004 kg·ha\(^{-1}\)) than in the third year (2338 kg·ha\(^{-1}\)). Mean vegetative ground cover was lower (P < 0.05) in the first and second year (92.7 and 93.9%, respectively) than in the third year (95.6%).

**Treatment effects**

Forage management practices that maintain high levels of soil organic matter, litter, and vegetative cover have been shown to improve rainfall infiltration (Naeth et al., 1990). In this study, infiltration rate tended to be greater (P = 0.06) in the U paddock than in the harvested paddocks (Table 5). The percentage of precipitation lost as surface runoff from paddocks with the U and 10R treatments was similar and less (P < 0.05) than that of paddocks with the 5C and 5R treatments. The percentage of surface runoff also did not differ between paddocks with the HS and 10R treatments. Sediment concentration in surface runoff did not differ among forage management treatments. The concentrations of total and soluble P in runoff from paddocks with the 5R and 5C treatments tended to be greater (P = 0.08) than in the paddocks with U, HS, and 10R treatments.

Because of the large amount of variation both within and between treatments, sediment load did not differ among forage management treatments, though there was the general trend for sediment load to increase as grazing pressure increased. In contrast, the total and soluble P load did not differ between paddocks with the U, HS, and 10R treatments, but were greater (P < 0.05) than paddocks with the 5C treatment. While total P load did not differ between paddocks with the HS, 10R and 5R treatments, soluble P load was greater (P < 0.05) from paddocks with the 5R treatment than paddocks with the HS and 10R treatments. No significant forage management treatment by year interactions were observed for infiltration rate, percentage of surface runoff,
concentrations of sediment, total P, or soluble P in surface runoff, or for the total amount of sediment or P load in surface runoff.

Bray-1 soil P did not differ among forage management treatments (Table 6). Soil moisture and surface roughness were greater ($P < 0.05$) in ungrazed paddocks than in the other forage management treatments, but no difference was observed among the harvested forage treatments. Penetration resistance was lower ($P < 0.05$) in the upper 14.0 cm of soil in ungrazed paddocks than in paddocks with the other forage management treatments and lower ($P < 0.05$) between 14.0 and 17.5 cm of soil in paddocks with the HS and 5C treatments. Penetration resistance in the upper 7.0 cm of the soil profile in paddocks with the HS treatment was lower ($P < 0.05$) than paddocks with the 5C, 5R, and 10R treatments. At depths greater than 17.5 cm, no difference was observed in penetration resistance among forage management treatments. No treatment by year interactions existed for Bray-1 P or penetration resistance at a depth of 3.5 cm or depths greater than 14 cm. Soil moisture had a tendency ($P = 0.08$) for a forage management treatment by year interaction with the 5C treatment having lower soil moisture in year 3 (16.9%) than did the other forage harvest treatments (18.2% for 5R, 10R, HS) in that year. This effect may be an indication that some change in the water-holding capacity of the soil had begun to occur in the 5C treatment, as a reduction in soil organic matter has been shown to result in a reduction in water-holding capacity of the soil (Betteridge et al., 1999).

Maintaining optimum vegetative surface cover and forage sward height are known to reduce the movement of nutrients entering streams from grazing lands (Nelson et al., 1996). Forage sward height and forage mass were greater ($P < 0.05$) in ungrazed paddocks than in harvested treatments (Table 6). Forage sward heights did not differ between paddocks with the HS and 10R treatments and were greater ($P < 0.05$) than either of the treatments grazed to 5 cm. However, mean forage masses of paddocks with the 10R treatment were greater ($P < 0.05$) than paddocks with the HS treatment. Although the 5R and 5C treatments were grazed to heights of 5 cm, the mean sward heights and forage masses at the time of the rainfall simulations of paddocks with the 5R treatment were greater ($P < 0.05$) than paddocks with the 5C treatment. Forage management treatment by year interactions were observed for forage sward height ($P < 0.05$) and forage mass ($P < 0.05$). Forage sward height in year 1 was greater in the 5C and HS treatments than in years 2 and 3. Paddocks with the 5R treatment had greater sward heights in year 2 than in years 1 and 3, and paddocks with the 10R and
treatments had greater sward heights in year 3 than years 1 and 2. These interactions were likely a result of the temporal variation of rainfall simulations relative to grazing periods. Whether a rainfall simulation was conducted near the beginning or end of the 35-day rest period in one of the rotational stocking systems in a given year would impact forage sward height and forage mass measurements in that year. Forage mass followed the same general trends as did forage sward height.

The mean percentage of vegetative ground cover between paddocks with the U, 10R, and HS treatments (96.5%) was greater \((P < 0.05)\) than the 5C treatment (87.7%). The primary role of forage cover in preventing erosion and nutrient loss is to decrease the kinetic energy of the raindrops before they strike the soil surface (Thurow et al., 1986). A significant forage management treatment by year interaction was observed for vegetative surface cover. Vegetative surface cover of paddocks in the 5C treatment increased from 83.6% in year 1 to 89.1% and 91.5% cover in years 2 and 3, respectively, while surface cover in the other forage management treatments remained constant from year to year.

**Slope effects**

Infiltration rate was greater \((P < 0.05)\) at low slope than at high slope sites (Table 7). Surface runoff was 7.2% greater \((P < 0.05)\) from high slope than low slope sites. Sediment concentration of surface runoff was not affected by the slope of the rainfall simulation site. However, total \((P = 0.08)\) and soluble \((P < 0.05)\) P concentrations in runoff were greater from low slope than high slope sites. Because of the greater volume of surface runoff in high than low slope sites and the similar concentrations of sediment in the runoff between slope sites, sediment load was greater \((P < 0.05)\) from the high slope than low slope sites, this influence of surface slope on sediment loss has been widely demonstrated in other studies (Liu et al., 1994; Russell et al., 2001). In contrast, the greater volume of runoff and the lower concentrations of total and soluble P in runoff from high slope than low slope sites resulted in no difference in total P and soluble P movement from the high and low slope sites. No forage management treatment by slope interactions were observed for infiltration rate, surface runoff, concentrations of sediment or P in surface runoff, or for the total amount of sediment or P load in surface runoff.

The concentrations of Bray-I P in the soil \((P = 0.06)\) and soil moisture \((P = 0.09)\) tended to be greater at low slope than at high slope sites of the paddocks (Table 8). This effect was likely related to prevalence of
low slope sites near hill tops or toe slopes where erosion was less and/or alleviation occurred. The higher soil P at the low slope area may account for the greater total and soluble P concentration in surface runoff. Cornish et al. (2002) observed a trend for increased total and soluble P concentrations in surface runoff as soil Bray-I P increased. In spite of the difference in soil moisture, penetration resistance to a depth of 35 cm and surface roughness did not differ between high and low slope sites. No slope by forage management treatment interactions occurred for Bray-I soil P, soil moisture, penetration resistance in the upper 3.5 cm of the soil, or surface roughness. Significant slope by forage management treatment interactions or the tendency for an interaction were observed for penetration resistance at all depths greater than 3.5 cm. Penetration resistances were greater at low slopes in paddocks with the 5R and 10R treatments and greater at high slopes in paddocks with the 5C, HS, and U treatments at all depths.

Forage sward height and forage mass were less (P < 0.05) at high slope than low slope sites. The percentage of vegetative ground cover tended to be less (P = 0.09) at high than low slope sites. The lower sward height, forage mass, and vegetative ground cover at high slope may partially be related to lower moisture concentrations (Guretzky et al., 2004) on high sloping areas. No slope by forage management treatment interactions were observed for forage sward height, forage mass, or vegetative surface cover.

**Season effects**

Infiltration rate in pastures has been reported to be greater during dry seasons than during wet season (Elliott and Carlson, 2004). This effect was observed in the current study with a greater (P < 0.05) infiltration rate during the summer than during the late spring (Table 9). The percentage of surface runoff was greatest (P < 0.05) during the late spring, intermediate during the fall and early spring, and lowest during the summer sampling periods. Mean concentrations of sediment in the runoff did not differ among seasons. Mean concentrations of total P in runoff during the early spring were lower (P < 0.05) than during the late spring and summer, but did not differ from measurements in the fall. Similarly, the concentration of soluble P in runoff was lower (P < 0.05) during the early spring and fall than during the late spring and summer. The soluble P concentration in runoff was also lower (P < 0.05) in late spring than summer. The high concentration of soluble P in the summer may be partially related to P release from the die back of microbial growth during dry season or the death and wilting of forage during the hot season releasing dissolved P into the runoff (Pote et al., 1999).
The amounts of sediment, total P, and soluble P load in runoff from rainfall simulations in the late spring were greater ($P < 0.05$) than during the other seasons. Elliott et al. (2002) reported similar seasonal variability in sediment load in surface runoff from pasture, with peak erosion occurring during the wetter portion of the year.

There were no forage management treatment by season interactions for infiltration rate, sediment concentration in surface runoff, or total sediment load in surface runoff. Significant forage management treatment by season interactions existed for percentage runoff ($P < 0.05$), concentrations of total and soluble P in runoff ($P < 0.05$), and amount of total and soluble P load by surface runoff ($P < 0.05$; Fig. 2).

The concentration of Bray-1 P in soil was lower ($P < 0.05$) in the early spring than in the other seasons of the rainfall simulations (Table 10). The low concentrations of total and soluble P in runoff in the early spring may be partially related to the lower soil Bray-1 P during this season. The moisture content of soil in the upper 5 cm was lower ($P < 0.05$) in the summer than in other seasons. Relatively small changes in soil moisture are able to significantly alter penetration resistance of the soil (Pires da Silva et al., 2003). Therefore, the differences in soil moisture likely resulted in soil penetration resistances at all depths being greater ($P < 0.05$) during the summer season than other seasons. No forage management treatment by season interactions existed for Bray-1 P, soil moisture or penetration resistance at a depth of 10.5, 14, or 35 cm. Penetration resistance in the HS treatment was similar to that of the U treatment during the late spring, while it was similar to that of the grazed paddocks at the other times of the year (treatment x season, $P < 0.05$) at the 3.5, 7.0, 17.5, 21, 24.5, 28.0, and 35.0 cm depths.

Mean forage sward height was greater ($P < 0.05$) during the late spring than during the other seasons and decreased through the summer, fall, and following early spring. In spite of the difference in sward height, mean forage mass in paddocks with the 5 treatments did not differ between seasons. Sward height and forage mass were similar during the late spring and summer periods in the 5C and 5R treatments, but decreased more in the fall and early spring in the 5C treatment than in the 5R treatment as a result of forage regrowth that occurred during the rest periods in the 5R treatment. Sward height and forage mass of paddocks with the HS treatment were lowest during the early spring as a result of hay harvest and increased throughout the year, while the grazed paddocks had greater sward heights and forage masses in the late spring, which decreased through the remainder of the grazing season (treatment x season, $P < 0.05$). Mean vegetative ground cover was lower ($P$
Vegetative surface cover of the U treatment remained constant throughout the year (99%) while the harvested treatments remained constant from late spring into summer, increased in the fall and remained constant into the early spring (treatment x season, \( P < 0.05 \)).

**Regressions predicting infiltration rate, surface runoff and sediment, total P and soluble P load**

Of the characteristics measured, a model including soil moisture, forage mass, surface cover, and penetration resistance at 3.5 and 30.5 cm was the best predictor of infiltration rate \( (R^2 = 0.11; \text{Table 11}) \). Of the variables measured, forage mass \( (R^2 = 0.10) \) and sine of surface slope in radians \( (R^2 = 0.07) \) were most highly related to surface runoff. The vegetative surface cover was the best predictor of sediment load \( (R^2 = 0.17) \) and total P load \( (R^2 = 0.13) \) in surface runoff. The only variables that were significantly related to soluble P load \( (P < 0.15) \) were penetration resistance at a depth of 3.5 cm, surface cover, forage mass, and sine of slope. However, the inclusion of all these variables only resulted in an \( R^2 \) of 0.06. The model with the greatest \( R^2 \) (0.23) generated for any of the dependent variables was for surface runoff, but this model included 8 independent variables. The number of variables included and the relatively low \( R^2 \) stresses the complex nature of the forage systems as they relate to hydrological processes and sediment and P load. These results are similar to work by Pearce et al. (1998) who concluded that a variety of soil and forage characteristics are important in the control of hydrological processes. Even though many factors are important, soil moisture appears to be important in controlling infiltration and surface runoff, whereas and the amount of vegetative surface cover, penetration resistance in the upper 3.5 cm of the soil, and the slope of the land were important in the control of surface runoff and sediment and P load in surface runoff.

**DISCUSSION**

Sustainable management of grazing lands requires management of vegetative cover, not only to provide feed for grazing livestock, but also to hold soil in place, filter water, and recycle nutrients (CAST, 2002). Some sediment and nutrient loss occurs from ungrazed pastures with complete ground cover (McDowell et al., 2003; Elliot and Carlson, 2004). Livestock grazing increases the likelihood of higher concentrations of nutrients in surface runoff because of soil disturbance, plant damage, and the deposition of dung on the soil surface (McColl and Gibson, 1979; Gillingham and Thorrold, 2000). In the current study, concentrations of total and soluble P
in surface runoff from a rotational stocking system with 10 cm of residual forage and summer hay harvest with winter grazing of stockpile forage were not greater than from an ungrazed pasture.

Grazing livestock may reduce the hydraulic conductivity of the soil, decreasing infiltration rate and increasing the volume of surface runoff, as a result of treading damage (Elliott et al., 2002), decreasing soil organic matter (Naeth et al., 1991b; Betteridge et al., 1999) and reducing leaf litter resulting in a reduction of aboveground water-holding capacity (Naeth et al., 1991a). Management practices that reduce the total volume of surface runoff and encourage infiltration will reduce the potential for sediment and P losses from pastures (Gburek et al., 2000). Grazing practices that allow litter to accumulate on the soil surface can improve infiltration rate (Naeth et al., 1991b). The proportion of rainfall lost as surface runoff and total and soluble P load from the 10R treatment was not different from the U treatment, suggesting that leaving approximately 10 cm of residual forage was sufficient to maintain the hydraulic conductivity of the pasture.

In addition to grazing pressure, infiltration rate and surface runoff are affected by antecedent soil moisture. When soil moisture is high, infiltration rate decreases with a corresponding increase in surface runoff (Naeth et al., 1990; Merz and Plate, 1997). These differences in soil moisture are often seasonal in nature (Elliot and Carlson, 2004) and the greater volume of surface runoff which occurs during wet periods is responsible for transporting a relatively large proportion of the sediment lost each year (Elliott et al., 2002). In the current study, the lowest infiltration rate and greatest surface runoff and sediment, total P and soluble P loads occurred during the late spring sampling period in the month of June. This is the month when the greatest amount of rainfall occurred and when soil moisture was greatest. Soil moisture was inversely related to infiltration rate being the most important factor in determining infiltration rate.

Even at 100% surface cover, some sediment and nutrient transport in surface runoff will take place in pastures (Elliott et al., 2002). However, the sediment and nutrient loads in surface runoff will increase as vegetative cover decreases with grazing (Elliot and Carlson, 2004). In the current study, the percentage vegetative cover on the soil surface was significantly correlated with infiltration rate, surface runoff, and sediment, total P, and soluble P load in runoff. Decreasing surface cover exposes the soil surface to greater raindrop impact and increases sediment movement through splash erosion (Pearce et al., 1998). The lowest amount of vegetative surface cover occurred in the SC treatment in the current study. Switching from
continuous stocking (5C) to rotational stocking (5R) with the same target forage sward height of 5 cm allowed for forage regrowth, prevented bare patches from developing within the pasture and helped to maintain vegetative cover at greater than 90%. The 10R and HS treatments maintained surface cover at approximately 95% and did not differ from an ungrazed pasture.

Areas of high slope are likely to have greater surface runoff and higher soil erosion rates than do low slope areas, partially due to increased flow velocity at higher slopes (Shainberg et al. 1992). In the present study, the proportion of rainfall lost as surface runoff was 7% greater from high slope plots than from low slope plots. This greater volume of water allowed greater transport of sediment and nutrients and resulted in sediment loss from low slope plots that was 42% of that lost from high slope plots. Mwendera and Saleem (1997) found that across a wide range of grazing intensities, surface runoff increased when slope increased from 0 to 4% to 4 to 8%. At a heavy stocking rate, 17% of rainfall was lost as surface runoff on high slope plots, while only 6% of rainfall was lost as runoff on low slope plots.

MANAGEMENT IMPLICATIONS

Some sediment and P loss will occur in surface runoff from pastures even in the absence of forage harvest by either grazing or hay harvest. Losses will be accelerated under certain management practices, site conditions, and during certain periods of the year. Forage management practices that leave adequate forage residue on the surface, such as the 10R and HS treatments, will improve infiltration rate and protect the soil surface from the force of raindrops impacting the soil surface. These factors will result in a reduction of sediment, particularly from areas of high slope, and nutrient transport from the soil surface. Maintaining adequate surface cover is the most important factor in limiting sediment and P load in surface runoff from pastures. However, the poor correlation between surface cover and sediment and total P load indicates that other forage, soil and site characteristics such as forage mass, sward height, soil moisture, soil organic matter, soil P, and slope are also important in the control of rainfall infiltration and sediment and P losses. Areas of high slope have greater potential to generate surface runoff and sediment loss than do areas with low slope. Managing these areas separately, at a lower stocking rate, may be necessary to reduce sediment loss from hilly pastures. Greater surface runoff, sediment loss, and total and soluble P losses occurred during the late spring than at other times.
of the year. Avoiding grazing of pastures near surface waters during this period may be necessary to minimize surface runoff, sediment load, and P load.

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LITERATURE CITED


