CHAPTER

5

Nutrient Management on Pastures and Haylands

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“…nutrient management of pastures and haylands has enormous production, economic, and environmental implications”

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INTRODUCTION

Judicious use of nutrients is critical for management of the 74 Mha of U.S. pasture and haylands (Fig. 1.1) owing to its agronomic, economic, and environmental implications. The primary goal of nutrient management is to promote biomass productivity that provides profit for producers while minimizing negative environmental impacts. Additional goals include improvement of soil quality, increased soil carbon (C) sequestration, and providing important ecosystem services.

The scientific literature is replete with examples of forage response to fertilization that increase agronomic yield. However, when fertilizer costs are considered, maximum forage yields are often not in the best interest of producers; aiming for maximum economic yield with less nutrient inputs is desired. This is especially true in today's economic climate because fertilizer costs, especially nitrogen (N), are directly tied to energy costs.

Although production and producer profit are important, protecting the quality of soil, water, and air resources is imperative

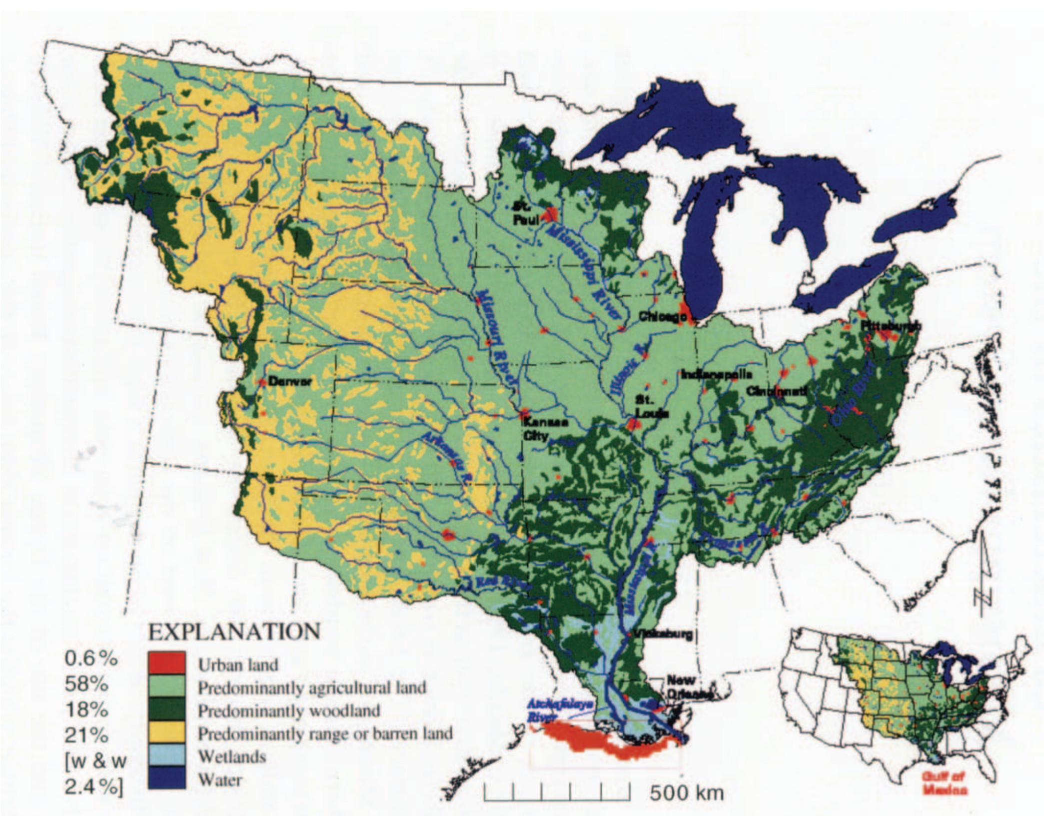

FIGURE 5.1. Mississippi River drainage basin showing major tributaries and the general location of the hypoxic zone south of New Orleans in midsummer, 1999. Reprinted with permission from Goolsby and Battagli, 2000.

Proper nutrient management on pastures and haylands allows for healthy aquatic ecosystems.

to sustain the human race. The public has increased interest in having agricultural land provide ecosystem services like wildlife and plant diversity. According to the 2004 national water quality inventory report to Congress, the U.S. Environmental Protection Agency (USEPA) reported nearly 44% of U.S. rivers and streams; 64% of lakes, ponds, and reservoirs; and 30% of bays and estuaries waters are too impaired to meet one or more of their designated uses (USEPA, 2009). The report implied that agriculture negatively affected 38% of impaired rivers and streams; 16% of impaired lakes, ponds, and reservoirs; and 10% of impaired bays and estuaries. Nutrients were specifically listed as a cause of impairment for approximately 16% of impaired river and stream banks, 19% of impaired lake areas, and 14% of impaired estuary areas.

Pasture and haylands comprise 6% of U.S. lands (Fig. 1.2), most of which are in the Mississippi River Basin (Fig. 5.1), and thus their management has a large potential to impact environmental quality in the central USA. Nutrient runoff from the Mississippi Atchafalaya River Basins (MARB) (Fig. 5.1) produces the second largest coastal hypoxic zone in the world (Rabalais et al., 2002), which is detrimental to commercial and sport fisheries in the northern Gulf of Mexico. Although attention has been focused on losses of N from row crops (National Research Council [NRC], 2008), recent evidence suggests animal manure from pastureland contributes nearly as much P as row crops to the Gulf (Alexander et al., 2008). Proper nutrient management on U.S. pasture and haylands can help reduce hypoxic conditions in the Gulf of Mexico and other U.S. coastal zones.

Nutrient management affects important soil–atmospheric interactions. Processes that remove and store carbon dioxide (CO_2) from the atmosphere and/or retard release of CO_2, methane (CH_4), and nitrous oxide (N_2O) to the atmosphere can help mitigate global climate change. Soil organic matter contains the largest terrestrial pool of C (Lal, 2004), and soils can be managed to sequester greater amounts of C, which improves soil quality in addition to lowering CO_2 content of the atmosphere. Nutrient management exerts control over sequestration of soil C under pasture and haylands via its influence on net primary productivity. Moreover, N management directly affects amount of N_2O emissions from pasture and haylands. Lastly, nutrient management influences ammonia (NH_3) volatilization from soils, which has important N-use efficiency, air quality, and ecological implications.

It is clear that nutrient management of pastures and haylands has enormous production, economic, and environmental implications. Thus, it is imperative that national policy on nutrient management as outlined by NRCS in Practice Standard 590 be supported by science and implemented. In this synthesis of U.S. scientific literature we ask the question "Does the scientific literature on nutrient management of pasture and hayland support the purported benefits outlined in Practice Standard 590?" Table 5.1 shows these purported benefits, the criteria used to assess the benefits, and the relative strength of research support for each criterion.

To assist in determining the scientific underpinning of the above benefits we downloaded the Conservation Physical Practices Effects (CPPE) matrix from the NRCS website and considered the hypothesized responses relative to the NRCS Nutrient Management Practice Standard (590) (Table 5.2). We bound our literature synthesis to managed pastures used for grazing or fields used for hay production. We searched for U.S. literature addressing nutrient inputs to pastures/haylands and practices designed to retain nutrients in these agroecosystems.

BUDGET AND SUPPLY OF NUTRIENTS

Most grassland soils in the USA require nutrient additions to obtain optimum forage production and maintain desired plant species. Nutrient management in grasslands begins with budgeting nutrients based on the difference between the amounts of nutrients expected to be taken up by forage and the amounts made available within the soil. The difference is used to estimate the rates of nutrients to be supplied. Fertilizer recommendations developed by research at land grant universities are used almost exclusively to determine fertilizer rates to apply, although evidence from row crops suggests that these recommendations lead to overapplication of nutrients. This may also be the case with pastures.

Nutrient supply is only part of the picture—factors affecting forage uptake of applied nutrients (yield) are equally important, as it is the balance between supply and uptake that determines the potential for nutrient losses. Grazing and grazing management can affect the rate of forage growth, hence the extent of nutrient uptake and the potential for loss. And grazing animals recycle nutrients to the pasture and need to be considered. Once application rates are determined, decisions regarding source, timing, and placement method are needed to develop optimum nutrient management strategies. These strategies are commonly reported in land-grant university fertilizer recommendation bulletins. Source and placement may be generalized for most grasslands, but timing is specific to the species grown. Most nutrient additions should be made just before the forage starts rapid growth.

In addition to affecting forage production, decisions about nutrient source, timing, and placement affect physical, chemical, and biological conditions of the soil. They also affect air and water quality and atmospheric concentrations of greenhouse gases. Therefore, nutrient management decisions should include these multiple goals. In this section, we first review information related to budgeting and supplying nutrients to grasslands, followed by a scientific assessment of the nutrient management criteria listed in Code 590.

"...nutrient management in grasslands begins with budgeting nutrients"

TABLE 5.1. Purposes of the Nutrient Management Practice Standard (Code 590) and criteria for assessing achievement of the purposes.

Purposes of the practice standard	Criteria for assessing achievement of the purpose	Support by the literature
Budget and supply nutrients for plant production	by developing a nutrient management budget using all potential sources of nutrients, including crop residues, legume credits, and irrigation water	Strong support for hayland, but need manure credits for pastures and research on phytoavailability.
	by establishing realistic yield goals based on soil productivity information, historical yield data, climate, management, and local research	Moderate support, more research needed on lower quality land sites.
	by specifying the source, amount, timing, and method of applying nutrients to each yield goal while minimizing movement of nutrients and other potential contaminants to surface or ground waters	Strong support for application ahead of growth, more research needed for offseason applications.
	by restricting direct application of nutrients to established minimum setbacks (e.g., sinkholes, wells, gullies, surface inlets, or rapidly permeable soil areas)	Strong support, but mainly based intuitively from other studies. More research needed for pastures and haylands.
	address the amount of nutrients lost to erosion, runoff, drainage, and irrigation	Strong support that this is critical, but need more soils and sites, perhaps models.
	applications be based on current soil (within 5 yr) and tissue test results according to land grant university guidance	Moderate support, current soil tests do not report P or N indices.
Properly utilize manure or organic by-products as a plant nutrient source.	by reducing animal stress and death from toxic or poisonous plants	Moderate support, but not a major problem in humid areas.
	by improving and maintaining plant health and productivity	Strong support, except on roles of organic by-products.
	by basing management on target levels of forage utilization or stubble height as a tool to help ensure goals are met	Moderate support showing principles; little on specific management practices.
	by locating of feeding, watering, and handling facilities to improve animal distribution	Strong support that would benefit from quantitative models to better define.
Minimize agricultural nonpoint source pollution of surface and ground water resources.	by improving or maintaining riparian and watershed function	Moderate support, research needed on more soils and sites.
	by minimizing deposition or flow of animal wastes into water bodies	Strong support, but would benefit from models.
	by minimizing animal effects on stream bank stability	Strong support.
	by providing adequate litter, ground cover and plant density to maintain or improve infiltration capacity of the vegetation	Strong support in concept, but responses need to be quantified for a range of soils and sites.
	by providing ground cover and plant density to maintain or improve filtering capacity of the vegetation	Strong support, but responses need to be quantified for a range of species and mixtures.
	by minimizing concentrated livestock areas, trailing, and trampling to reduce soil compaction, excess runoff, and erosion	Strong support and a range of practices to minimize soil damage, but few to restore soil condition.

TABLE 5.1. continued.

Purposes of the practice standard	Criteria for assessing achievement of the purpose	Support by the literature
Protect air quality by reducing nitrogen emissions (ammonia and NOx compounds) and formation of atmospheric particulates.	by reducing accelerated soil erosion	Strong support, would benefit from use of models.
	by minimizing concentrated livestock areas to enhance nutrient distribution and improve ground cover	Strong support, but needs to be integrated with plants and their growth habits.
	by improving carbon sequestration in biomass and soils	Strong support, would benefit from use of models to quantify relationships.
	by application of soil nutrients according to soil test to improve or maintain plant vigor	Strong support for most monocultures, need more research on mixtures.
Maintain or improve physical, chemical, and biological condition of the soil.	by applying and managing nutrients in a manner that maintains or improves the physical, chemical, and biological condition of the soil	Strong support intuitively based on annual crops, but needs verification using long-term perennials.
	by minimizing the use of nutrient sources with high salt content unless provisions are made to leach salts below the crop root zone	Strong support, but it does not appear to be a problem unless excess rates applied.
	by not applying nutrients when the potential for soil compaction and rutting is high	No support, research needed because perennials can become compacted, but are not tilled.

Nitrogen

Rates of fertilizer N applications to grasslands depend on N uptake capacity of the forage and N made available within the soil. Nitrogen uptake of forages varies depending on plant species, soil characteristics, and environmental conditions. Annual N uptake of a mixture of smooth bromegrass (scientific names of all plant species used in this chapter are listed in Appendix III) and alfalfa ranged from 90 to 211 kg N ha^{-1}, depending on amount of fertilizer N added (Nuttall, 1980). Annual N removal in New York was 241 kg N ha^{-1} for tall fescue and 205 kg N ha^{-1} for orchardgrass (Cherney et al., 2002). In Texas, annual N uptake of 'Coastal' bermudagrass fertilized with ammonium nitrate ranged from 121 to 409 kg N ha^{-1} depending on N rate used and environmental conditions (Silveira et al., 2007).

In natural systems like permanent pastures, available N for forage uptake is derived from that supplied from mineralization of soil organic matter and plant residues, grazing animal excreta, precipitation, and biological N_2 fixation. Nitrogen mineralized from soil organic matter and plant residues may range from 40 to 230 kg N ha^{-1} yr^{-1} and is positively related to soil organic matter content, residue composition and favorable environmental conditions (Hopkins et al., 1990; Hassink, 1995). In comparison, N derived from deposited animal excreta can be as high as 1200 kg N ha^{-1} in concentrated areas of deposition. This application rate is well in excess of potential forage uptake and can lead to N losses to air and water, although these hot spots of N loss may be distributed widely and comprise only a small percentage of the pasture area. The N received annually in precipitation usually ranges from 3 to 10 kg N ha^{-1} (Whitehead, 1995), and biological N_2 fixation can supply as much as 400–650 kg N ha^{-1} annually (Ledgard and Giller, 1995; Trott et al., 2004), although typical values range from 27 to 141 kg N ha^{-1} (Yang et al., 2010).

Several indices have been developed to evaluate potential N mineralization during a growing season from soil organic matter (Schomberg et al., 2009), but currently there is no method to obtain an accurate estimate. The amounts of N mineralized from soil organic matter

TABLE 5.2. Conservation Physical Practices Effects (U.S. Department of Agriculture–Natural Resources Conservation Service [USDA–NRCS], 2009) on pastures and haylands associated with the NRCS Nutrient Management Practice Standard (Code 590).

Variable	Effect	Rationale
Plant selection or condition		
Plants not adapted or suited	Slight to substantial improvement	Nutrients and soil amendments are optimized to enhance suited and desired species.
Productivity, health, and vigor	Slight to substantial improvement	Nutrients and soil amendments are optimized to enhance health and vigor of desired species.
Forage quality and palatability	Moderate to substantial improvement	Proper management will increase quality and palatability of forage.
Domestic animals		
Inadequate quantities and quality of feed and forage	Moderate to substantial improvement	Nutrients are managed to ensure optimal production and nutritive value of the forage used by livestock.
Stress and mortality	Slight to substantial improvement	Management results in nutritive forage improving livestock health.
Air quality		
Excessive greenhouse gas—carbon dioxide	Slight improvement	Management of nutrients optimizes the storage of soil carbon.
Excessive greenhouse gas—nitrous oxide	Slight improvement	Reduction in N in waste results in less N volatilization.
Excessive greenhouse gas—methane	Slight to moderate improvement	Proper nutrient management reduces methane production.
Ammonia	Slight to moderate improvement	Proper nutrient management reduces ammonia production.
Objectionable odors	Moderate to substantial improvement	Proper management and application/incorporation of manures and some biosolids reduces volatilization, volatile organic compounds, and particle transport.
Water quality		
Excessive nutrients and organics in groundwater	Substantial improvement	The amount and timing of nutrient application are balanced with plant needs.
Excessive salinity in groundwater	Slight improvement	Proper nutrient application should reduce salinity if nutrient source contains salts.
Harmful levels of heavy metals in groundwater	Slight to moderate improvement	The action limits the total amount of heavy metals that can be applied to a site, ensuring that harmful levels are not leached to groundwater.
Harmful levels of pathogens in groundwater	Slight improvement	The action limits the amount of manure that can be applied, thus preventing harmful levels of pathogens.
Excessive nutrients and organics in surface water	Substantial improvement	Source, amount, timing, and method of application are managed to maximize nutrient use efficiency by the crop and minimize potential for nutrient losses in leaching and runoff.
Excessive suspended sediment and turbidity in surface water	Neutral	Proper nutrient application will minimize losses due to runoff.
Excessive salinity in surface water	Slight improvement	Proper nutrient application should reduce salinity if nutrient source contains salts.
Harmful levels of heavy metals in surface water	Slight to substantial improvement	Changing pH will alter the solubility of metals. The action will reduce the application rate of heavy metals if required.
Harmful levels of pathogens in surface water	Slight improvement	Decrease application of pathogens if nutrient source contains pathogens.

TABLE 5.2. continued.

Variable	Effect	Rationale
Soil condition		
Organic matter depletion	Slight to moderate improvement	Applying sufficient nutrients will maintain or enhance biomass production.
Contaminants—N, P, and K in commercial fertilizer, animal wastes and other organics	Slight to moderate improvement	Proper application results in reduced risks of contamination from N, P, and K.
Contaminants—salts and other chemicals	Slight to moderate improvement	Decreased excess nutrients results in reduced salts and other contaminants.
Compaction	Slight to moderate worsening	Field operations on moist soils cause soil compaction.

are usually based on studies of crops grown with and without N fertilizer applications. Mineralization from soil organic matter depends on environmental conditions such as soil temperature and water content, so rates are expected to vary from year to year (Cabrera and Kissel, 1988).

Consequently, research is needed for pastures and haylands to develop appropriate methods to identify pools and rates of mineralizable N from soil organic matter. These data will help develop simulation models that allow estimation of mineralized N using real-time environmental conditions (Schomberg and Cabrera, 2001). Process-based models that can predict N release and uptake have been developed (Zhang et al., 2002) but have not yet been translated into decision support tools. As with release from soil organic matter, release of N from plant residues is also strongly affected by environmental conditions (Lory et al., 1995; Rodriguez-Lizana et al., 2010). Biological N_2 fixation by legumes can contribute significant amounts of N to grasslands.

Alfalfa–orchardgrass pastures in Iowa increased from 15 to 136 kg N ha^{-1} yr^{-1} as the percentage of alfalfa in the mixture increased from 11% to 55% (West and Wedin, 1985). Similarly, alfalfa–bermudagrass pastures in Texas fixed from 80 to 222 kg N ha^{-1} yr^{-1} (Haby et al., 2006). Despite the significant N contributions from legumes, their use has decreased in hayland and pasture systems of the USA, due in part to the difficulty of maintaining legumes in mixed stands and to the availability of low-cost N fertilizers (Howarth et al., 2002). Recent increases in energy costs of producing N fertilizer, fertilizer prices have increased and have led to renewed interest in use of mixed stands of legumes and grasses. Therefore, research is needed to fine-tune management systems that improve the persistence of legumes in mixed stands. Also, research is needed to evaluate effects of legume proportion on N_2 fixation and transfer to grasses, particularly with warm-season grasses (Haby et al., 2006). Research is also needed on genetic selection of legumes for higher N_2 fixation capacity and tolerance to acidity and low P levels, as well as improvement of bacterial strains and inoculant carriers (Graham and Vance, 2000).

Because of the natural variability in environmental conditions, N application rates for grasslands have been typically derived from economic analyses of long-term experiments carried out with plots in which forage is cut instead of grazed (Power, 1985). Mechanical forage harvesting studies are relatively easy to conduct because they avoid dealing with uneven stubble heights and manure deposition from grazing animals. Although the results from these studies are certainly appropriate for hay and silage production, optimum N application rates for grazed grasslands need to be adjusted downward to account for recycling via deposited excreta. Research is needed to develop these guidelines.

Grazing animals return 75–95% of ingested N to the soil via feces and urine, which can become partly available for plant uptake

Proper nutrient management on pastures results in good forage yields and healthy livestock.

(Whitehead, 1995). The N available from cattle excreta for plant growth during the first year is derived mostly from inorganic N present in urine or feces, and from mineralization of some of the organic N. Excreta-derived N in subsequent years is mostly made available through mineralization of organic N. When considering excreta N that becomes plant available, it is necessary to take into account the distribution of N between feces and urine. In general, the percentage of total excreta N deposited as urine increases from 45% to 80% as concentration of diet N increases from 1.5% to 4% (Whitehead, 1995). Consequently, nutrient management should be tightly linked to diet management.

From 2% to 21% of urine N may be lost through NH_3 volatilization (Mulvaney et al., 2008). In contrast, NH_3 losses from cattle feces are usually low, although values as high as 8–13% have been reported (Sugimoto and Ball, 1989; van der Molen et al., 1989). A portion of the urine and excreta that is not lost through NH_3 volatilization may become plant available, depending on composition. Nitrogen in urine of cattle and sheep is mainly in the form of urea, with smaller amounts as allantoin, hippuric acid, creatinine, creatine, uric acid, and ammoniacal-N (Whitehead, 1995). With the exception of ammoniacal N, urine constituents are organic compounds that need to be mineralized before becoming plant available.

In New Zealand, a pasture of perennial ryegrass and white clover recovered 19% of the N from applications of cattle or sheep urine (Williams and Haynes, 1994). In a similar Netherlands study, perennial ryegrass recovered 16% of cattle urine N and 8% of manure N (Deenen and Middelkoop, 1992). Plant recoveries were low because of the presence of urine and manure patches with N concentrations that greatly exceeded plant N requirements. When calculated on an area basis, these deposition rates typically range from 500 to 670 kg N ha^{-1} for urine, and from 1200 to 2000 kg N ha^{-1} for manure (Lantinga et al., 1987; Whitehead, 1995).

Estimations of N available through mineralization of excreted N in years following deposition may need to consider the amount of time during which the specific grassland has been under grazing management, as well as the yearly rate of accumulation and mineralization of organic N. In the Southern Piedmont region, Franzluebbers and Stuedemann (2009) evaluated the linear rate of change in soil organic N in tall fescue/bermudagrass paddocks that were either hayed or grazed (800–1700 steer days ha^{-1} yr^{-1}) for 12 yr, with an average inorganic fertilizer application of 246 kg N ha^{-1} yr^{-1}. The rate of N accumulation in the upper 30 cm was 57 kg N ha^{-1} yr^{-1} in hayed paddocks and 103 kg N ha^{-1} yr^{-1} in grazed paddocks. The extra accumulation of 46 kg N ha^{-1} yr^{-1} in grazed paddocks suggest that grazed grasslands likely require less fertilizer N than hayed grasslands in similar soils.

One of the greatest challenges associated with nutrient crediting from manure deposition is the lack of uniformity of nutrient distribution from deposited manure, especially in pastures with low animal densities. This issue is described in more detail later, but is a major reason why there is a general lack of U.S. research data on N (and other nutrients) requirements of grazed pastures. Consequently, research is needed to characterize the nutrient distribution from deposited manure in grazed grasslands, especially with different grazing methods, to fine-tune fertilizer N recommendations.

When natural sources of N are not sufficient for optimum grassland performance, N sources such as commercial fertilizers, animal manures, and biosolids can be used. Livestock manures and biosolids are generally lower cost per unit of nutrient than commercial fertilizers and they are readily available for most producers.

Livestock and poultry in the USA excrete 1.6 billion tonnes of fresh manure annually (Table 5.3), and this excreted manure contains approximately 75% of the commercial fertilizer N consumed in the USA each year. In addition, there are 40% fewer livestock and poultry farms than 30 yr ago, but the number of animals has increased (U.S. Department of Agriculture–National Agricultural Statistics Service [USDA-NASS], 2009). Although most of the manure produced by cattle in the USA is excreted directly and nonuniformly, onto pasture and rangelands, the great majority of the manure generated by pigs and poultry is collected and can be managed as a crop nutrient resource.

Manures that are collected and managed are important nutrient sources for pastures and haylands, especially in the lower midwest and southern and eastern USA. Total N excreted in the collectable fraction of livestock and poultry manure is nearly 28% of U.S. commercial fertilizer N consumption, but 30–85% of this N may be lost to the atmosphere during manure storage and application depending on the manure management system. Therefore, the potential N replacement value of collectable manure is likely less than 15% of U.S. commercial fertilizer N consumption.

Biosolids are another potential source of nutrients for pastures and haylands. Approximately 57 million dry tonnes of biosolids are generated annually from the treatment of over 45 trillion liters of wastewater (USEPA, 1999). Roughly 40% of these biosolids are beneficially reused via land application. Biosolid applications to pastures are also highly regulated. The USEPA requires a 30-d interval between application of biosolid and hay harvest or grazing to minimize the potential for direct ingestion of the biosolids and any pathogens present in the material (USEPA, 1994).

A significant challenge facing producers is that the nutrient composition of manures and biosolids generally is not balanced relative to crop nutrient requirements. Applying these materials at rates needed to meet plant N needs creates overenrichment of soil P. Although substantial variations in manure nutrient composition exist, manure $N:P_2O_5:K_2O$ is typically around 1:1:1 (Council for Agricultural Science and Technology [CAST], 2006),

“…nutrient composition of manures and biosolids generally is not balanced relative to crop nutrient requirements”

TABLE 5.3. Estimated quantities of manure and manure nutrients produced annually in the USA compared to consumption of commercial N, P (as P_2O_5), and K (as K_2O) fertilizer.

	Animals[1]	Manure[2]	N[2]	P_2O_5[2]	K_2O[2]
	Millions	Millions of tonnes	Thousands of tonnes		
Beef cattle	60	1191	5199	2424	4299
Milk cattle	17	257	1597	617	839
Horses	4	37	133	44	48
Hogs	68	74	596	256	338
Layers	350	11	203	146	90
Pullets	106	1	17	12	8
Broilers	1603	44	485	324	330
Turkeys	107	8	119	80	66
Ducks	4	< 1	2	1	1
All animals	2319	1623	8,351	3904	6019
Collectable manure[3]		424	3098	1471	2037
Deposited manure[4]		1,199	5253	2433	3982
Fertilizer consumption[5]			11,117	3940	4365

[1]Number of animals in inventory from 2007 Census of Agriculture (U.S. Department of Agriculture–Natural Agricultural Statistics Service [USDA-NASS], 2009). [2]Excretion values from USDA–Natural Resources Conservation Service (USDA-NRCS) Agricultural Waste Management Field Handbook (USDA-NRCS, 2008). [3]Includes 100% of manure from beef finishing cattle, 15% of manure from other beef cattle (not including beef cows), 50% of manure from lactating dairy cows, 15% of manure from other dairy cattle, and 100% of manure from hogs and poultry. [4]Includes 100% of manure from beef cows, 85% of manure from other beef cattle (not including beef finishing cattle), 50% of manure from lactating dairy cows, 85% of manure from other dairy cattle, and 100% of manure from horses. [5]Average 2000–2009 U.S. commercial fertilizer consumption (American Plant Food Control Officials and The Fertilizer Institute [AAPFCO-TFI], 2011).

whereas removal rates from grazed pastures are closer to 3:1:3 (Joern et al., 2009). When animal manures, biosolids and most other nutrient rich by-products are applied at N-based rates to pastures, P applications are usually 2–4 times greater than crop-P removal. Repeated applications of these materials to pasture and hayland result in soil-test–P levels that are greater than those needed for optimum forage yields. High soil-test–P levels result in elevated soluble-P concentrations in water runoff from pastures. The effect of elevated soil-test–P levels and nutrient runoff from manured pastures and haylands on surface-water impairment are discussed in later sections.

Manure also lacks consistency in both nutrient content and particle size. Despite attempts to standardize procedures for both manure sampling and analysis (Peters, 2003), a review of numerous state Extension bulletins found on the Web clearly showed there is neither a single, widely accepted strategy for obtaining a representative manure sample for analysis nor uniform procedures for analyzing manure nutrient content.

Manure spread on pastures and haylands in the USA is usually surface applied with rear- and side-discharge spreaders, application uniformity with solid dairy and beef cattle manure had a coefficient of variation greater than 100% (Norman-Ham et al., 2008). Poultry litter is a more consistent material and more uniform rates can be achieved with proper overlap of each spreader pass. Achieving application uniformity with liquid-manure applications is a challenge for liquid tank spreaders and numerous irrigation systems.

One of the most significant challenges to achieve optimum utilization of manure and other organic by-products on pastures and haylands is the wide range of algorithms states use to determine manure nutrient availability. Among 34 states, 27 different variables were used to determine plant-available manure N and individual states use from 5 to 12 variables (Joern et al., 2009; Table 5.4). Although the number of variables required is indeed large, once the manure source is determined, 12 states use only one factor (e.g., total manure N) and 15 states use only two factors (e.g., total manure N and application method). Although all states require manure total N as part of their manure-N–availability algorithm, 14 states do not require the ammonium (NH_4) form of N, which is the manure-N component most likely to volatilize when surface applied to pastures and haylands. In addition, roughly one-third of the states in the database do not account for mineralization of residual N in the years following application. And, for the states that estimate residual manure-N credits, the time frame varies from 1 to 5 yr after application.

All but one state includes method of application to determine plant-available manure N. However, only 13 of 34 states use time (months) between manure application and crop N uptake to determine manure-N availability. It is likely that manure applications made to dormant pastures have significantly greater potential for N loss prior to crop uptake than manure applied to actively growing plants, yet the majority of states surveyed do not include this variable. This lack of a timing variable in many state algorithms is particularly problematic for producers with liquid manure, as storage limitations often require applications to crops that are not actively growing. These large discrepancies in algorithms can result in more than 50% differences among states for estimates of plant-available N from manure, and is probably the single greatest factor affecting optimum utilization of manure N in agricultural fields.

Studies to develop N recommendations for grasslands are usually carried out with commercial fertilizers such as ammonium nitrate that do not undergo significant NH_3 losses in neutral or acidic soils (Knight et al., 2007). Losses through NH_3 volatilization should be taken into account when using sources that lead to such losses, such as animal manures and commercial fertilizers that contain urea, as well as ammoniacal fertilizers applied to soils with pH > 7.5. For example, NH_3 losses ranged from 6% to 14% of the available N when poultry litter was surface applied to tall fescue at 70 kg available N ha^{-1} in Alabama, Georgia, and Tennessee (Marshall et al., 1999). Similarly, NH_3 losses ranged from 12% to 46% of the applied N when urea was surface applied to tall fescue at 50 kg N ha^{-1} in Georgia (Vaio et al., 2008). In

“discrepancies in algorithms can result in more than 50% differences among states for estimates of plant-available N from manure”

Soil sampling and testing is imperative for good forage production and environmental protection.

“methods that reduce both NH_3 loss and surface runoff are desirable for agronomic and environmental reasons.”

TABLE 5.4. Data needed to determine availability of manure N following manure application in 34 U.S. states.[1]

Input data	AL	AR	CA	CO	DE	FL	GA	IA	IL	IN	KY	KS
Number of years credited	1	1	3	3	4	1	1	3	4	1	2	3
Manure total N	R	R	R	R	R	R	R	R	R	R	R	R
Manure NH4–N	R			R	R				R	R		R
Manure dry matter (%)												
Animal type		R	R	R	R	R		R	R			
Manure is from poultry		R	R	R	R	R	R	R	R	R	R	
Manure contains bedding									R			
Storage type		R	R			R	R					R
Manure is liquid				R			R	R	R	R	R	R
Equipment type			R									R
Manure is injected	R	R	R		R	R	R	R	R	R	R	R
Manure is irrigated	R	R	R			R	R	R	R	R		R
Manure is aerially applied												
Days to incorporation	R	R	R		R	R	R	R	R	R	R	R
Application month	R	R								R	R	
Region of state	R									R		
Field is irrigated with water												
Field is artificially drained		R	R									
Field is annually manured			R								R	
Soil organic matter (%)		O										
Soil surface texture												
Soil drainage class		O	R									
Application rate												
Crop canopy/cover/type										R	R	
Application made in year of crop utilization										R	R	
Long-term monthly air temperature												
Average days in month soil is saturated												

[1]R indicates required input value and O indicates optional input value.

the same study, NH_3 losses from a solution of urea–ammonium nitrate applied at 50 kg N ha^{-1} ranged from 6% to 33%. Additional information on NH_3 losses from grasslands is presented in the air-quality section.

Broadcast applications of manures and N fertilizers are common in grasslands, and may lead to significant NH_3 losses, and exposure to surface runoff increases potential contamination of surface waters (Pierson et al., 2001). Consequently, methods that reduce both NH_3 loss and surface runoff are desirable for agronomic and environmental reasons. For example, surface or subsurface banding of N fertilizers may help reduce NH_3 loss and improve N-use efficiency (Raczkowski and Kissel, 1989; Vigil et al., 1993). A recently developed subsurface applicator that applies poultry litter in a band 5 cm deep and 4 cm wide (Pote et al., 2003; Warren et al., 2008) may be useful in reducing the surface area of poultry litter subject to NH_3 volatilization and N contamination of surface runoff. These applicator systems are encouraging, but are in the early stages of development.

Phosphorus

Phosphorus requirements of grasslands depend on soil-P availability and forage uptake capacity. Phosphorus uptake by forages can vary from 9 kg P ha^{-1} for white clover (Brink et al., 2001) to 83 kg P ha^{-1} for johnsongrass (Pierzynski and Logan, 1993), with

MA	MD	MI	MN	MO	MS	MT	ND	NE	NJ	NM	OH	OK	OR	PA	RI	SD	TN	UT	VT	WA	WI
6	4	4	3	3	1	3	3	4	6	3	1	3	1	6	6	2	4	1	3	1	3
R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
R	R	R		R	R			R	R	R	R			R	R	R	R		R		
									R					R					R		
	R	R	R			R	R	R	R	R			R	R			R	R			R
R	R	R	R	R		R	R	R	R	R		R	R	R		R	R	R	R		R
		R		R							R										
	R					R		R	R	R	R			R			R	R	R		
		R		R				R	R	R	R		R	R			R		R		
			R	R		R	R				R						R				
R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
					R	R		R		R			R			R	R	R		R	
										R	R										
R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
R					R			R	R	R	R			R	R			R	R		
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						O				O								O	O		
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R									R					R							
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bermudagrass taking up intermediate amounts of 29 to 73 kg P ha^{-1} (Brink et al., 2004; Read et al., 2006). Most available P in soils originates from mineralization of soil organic matter and plant residues plus P desorbed from clay minerals and amorphous iron and aluminum oxides. Soil-P availability varies with soil type and is commonly evaluated with soil tests such as Bray1, Mehlich-1 and Mehlich-3 (Mehlich, 1953). A small amount of P (0.2 to 1.5 kg P ha^{-1}) is also received yearly with precipitation (Newman, 1995; Owens et al., 2003).

Because of the many factors involved in P availability and P uptake, recommendation rates for P fertilizer to optimize forage production are usually determined empirically by conducting studies for selected forage, soil, and environmental conditions. Two strategies for fertilizing grasslands with P are considered; one emphasizes fertilizing the forage, whereas the other emphasizes fertilizing the soil. Both strategies require determination of the optimum soil-test level for a given soil type and forage. The optimum level is the soil-test level above which forage yield or quality does not respond to added P, and is commonly determined by carrying out studies with two treatments (control without added P, or a high P rate) in soils with different soil-test levels, ranging from low to high. For example, in Florida soils bahiagrass did not respond to added P when Mehlich-1 test levels were above 16 mg P kg^{-1} (Stanley and Rhoads, 2000).

“

In contrast to most commercial fertilizers, ...animal manures and biosolids have variable amounts of plant-available P.”

The strategy that emphasizes fertilizing the forage is based on applying the amount of P fertilizer needed to obtain optimum forage yield at the current soil-test P–level, implying P should be applied when the soil-test P is below the optimum level. To determine the amount of P fertilizer to be added, P-rate studies are conducted at different soil-test–P levels below the optimum level. The other strategy, which emphasizes fertilizing the soil, is based on adding the necessary P fertilizer to bring the soil to, or maintain it at, the optimum soil-test–P level. Field or laboratory studies are usually needed to determine how much P is necessary to add to soils with lower soil tests to bring them up to the optimum level. For example, in Georgia Piedmont soils, it took 4.1 mg P kg^{-1} to increase Mehlich-1 P level by 1 mg P kg^{-1} (Pierson et al., 2001); in Alabama it took an average of 4.5 mg P kg^{-1} per soil-test unit (Cope, 1983). Once the optimum soil-test level is achieved through addition of P fertilizer, this fertilization strategy consists of adding an amount of fertilizer P annually that is equal to that removed from the grassland to maintain the soil test at the optimum level.

Many studies have been conducted to establish P requirements of agronomic crops, but fewer have evaluated P requirements of forage crops and pastures (Pant et al., 2004). Part of the reason is that many grasslands are fertilized with animal manures, which have ratios of available N to available P that are much lower than those required by most forages. Further, as for N, most studies to evaluate P response of forages have been conducted with plots that are cut instead of grazed, which raises the question of their relevance for grazed situations. In general, cattle excrete about 75–90% of P in the diet, with most excreted P found in feces rather than in urine. However, P in excreta is utilized slowly because inorganic P in feces has low solubility and the organic P mineralizes slowly (Whitehead, 2000).

Few U.S. studies have compared P responses under hayed and grazed systems. Forage productivity and P uptake of smooth bromegrass in Iowa were greater in hayed and grazed plots than in nongrazed plots (Haan et al., 2007). In New Zealand, dry-matter production was greater in grazed than in mowed trials (Morton and Roberts, 2001), and in Australia, more frequent mowing resulted in greater response to P than did less frequent mowing (Cayley and Hannah, 1995). These results suggest forage production and P uptake are stimulated by forage harvest, either by grazing or hay harvest. Additional research should be conducted in the USA to confirm these results.

Phosphorus sources for grasslands include commercial fertilizers as well as animal manures and biosolids. The total P excreted in livestock and poultry manure in the USA is nearly identical to the commercial-P fertilizer consumption; but the P that can realistically be collected and managed as a nutrient resource is only about 37% of commercial fertilizer-P consumption (Table 5.3). In contrast to most commercial fertilizers, of which > 90% of the P is in plant-available form (water-soluble P + citrate-soluble P), animal manures and biosolids have variable amounts of plant-available P.

O’Connor et al. (2004) determined total P and percent phytoavailability of 12 biosolids to bahiagrass grown in two soils in the greenhouse compared to triple superphosphate (TSP, 100% phytoavailability). One biosolid had a total-P content of 3.2 g P kg^{-1}, whereas the remaining 11 had total-P contents ranging from 21.5 to 39.0 g P kg^{-1}. The relative bioavailability of two biosolids (which were produced through biological P removal processes without addition of Al) was high, ranging from 74% to 130% in the two soils used. In contrast, a biosolid produced with the above process, but with the addition of Al, had relative phytoavailability ranging from 31% to 63%. On average, pelletizing two of the biosolids reduced their phytoavailability (compared with nonpelletized form) from 80% to 55% and from 40% to 11%. Oladeji et al. (2008) determined P phytoavailability of two biosolids and poultry manure with respect to TSP in a greenhouse study using bahiagrass for 6 mo, followed by ryegrass for 5 mo, and a second bahiagrass for 4 mo. A field study was also carried out with improved pastures for 2 yr. Average P phytoavailability for the combined studies was 49% for poultry manure, 56% for a conventionally processed biosolid, and 84% for a biosolid that underwent a process similar to biological P removal. These results show that bioavailability of biosolid P depends on the

"Should total manure P or available manure-P values be entered into the P index or other risk assessment tool algorithms?"

Accurately calibrated fertilizer spreaders ensure good forage yields without excessive loss of nutrients to the environment.

wastewater treatment used and on subsequent treatments such as composting and pelletizing.

Several U.S. studies have evaluated P uptake in grasses fertilized with animal manures (e.g., Brink et al., 2004; Sistani et al., 2004), but few have compared the phytoavailability of manure P with respect to commercial P fertilizers.

There is variability among states in the proportion of manure P assumed to be plant available. In a survey by Joern et al. (2009), all but 8 of the 34 states surveyed assume that manure total P is 100% plant available relative to commercial fertilizer. Manure-P availability relative to commercial fertilizer P for the other eight states ranges from 60% to 90% and four of these states estimate residual-P availability in subsequent years. States that use less than 100% fertilizer-P equivalent for manure generally allow greater rates when manure is applied based on crop-P removal. Subsequent soil testing can confirm the need for additional P applications, but when manure is applied based on crop-P removal, the soil-test P can rise above agronomically responsive levels. This raises the question for those states that credit manure at less than 100% commercial fertilizer P efficiencies. Should total manure P or available manure-P values be entered into the P index or other risk assessment tool algorithms?

Because P recommendations are usually based on commercial fertilizers with high phytoavailability (> 90%), additional research should be carried out to characterize the P phytoavailability of different animal manures and biosolids with respect to commercial fertilizers such as TSP. Furthermore, depletion of rock phosphate deposits in the not-so-distant future will make it necessary to develop processes to recover as much P as possible from animal manures and biosolids to maintain agricultural productivity (Cordell et al., 2009). There is potential to develop recycled P materials with concentrations similar to those of commercial fertilizers (Szogi et al., 2008; Szogi and Vanotti, 2009), but the phytoavailability of these recycled products needs to be known to optimize application rates (Bauer et al., 2007).

With regard to placement, P fertilizers are usually broadcast onto pastures and haylands and as a result, surface runoff may solubilize some of the fertilizer P and transport it to surface waters (Franklin et al., 2005). Therefore, subsurface banding would be a preferred method for grasslands where surface runoff has the potential to contaminate surface waters with P (Pote et al., 2003).

Potassium

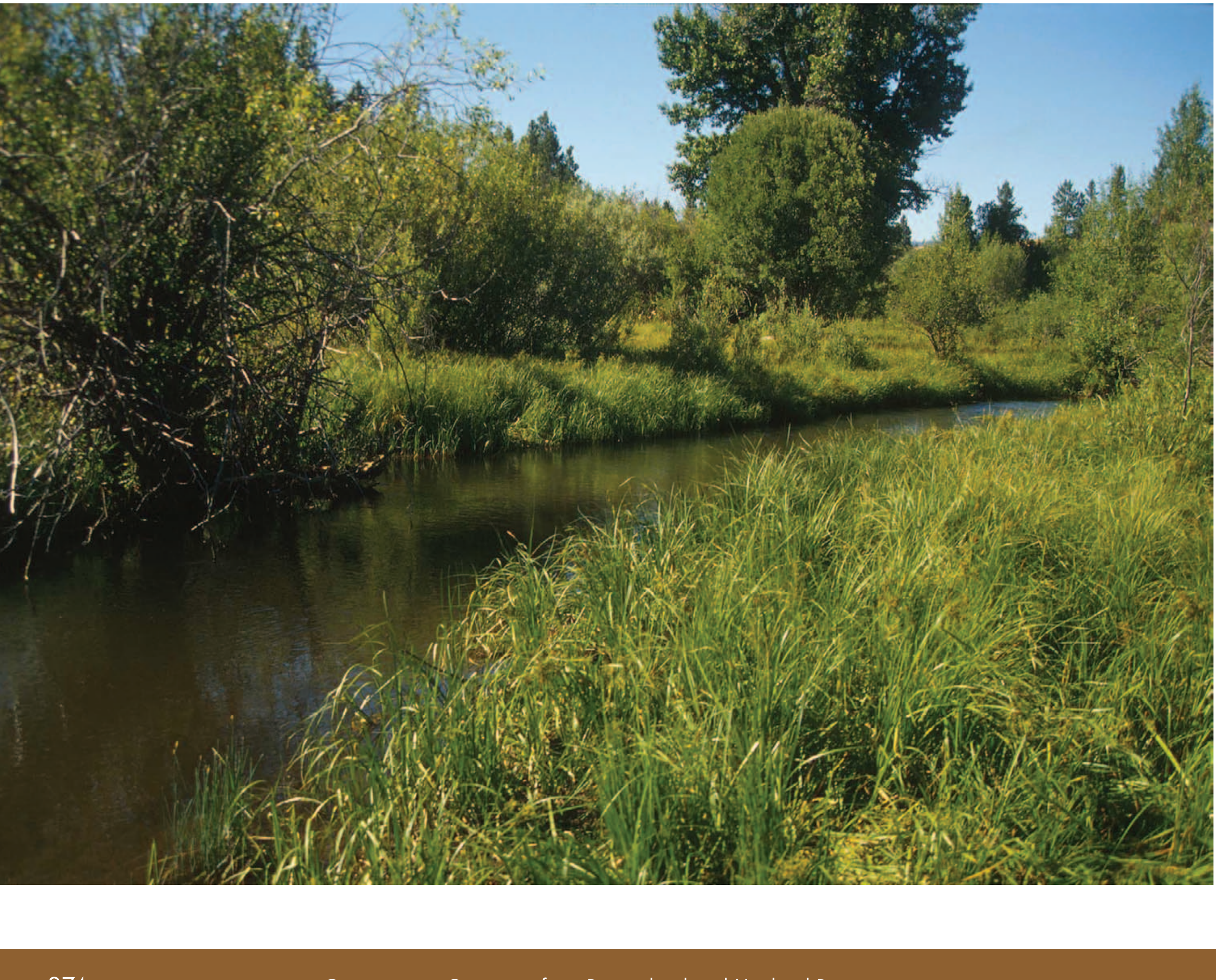
Potassium (K) fertilizer requirements for grasslands are determined from plant-available

K and K uptake by the forage. Forage K uptake can be high, especially when forage is cut for hay or silage. Alfaro et al. (2003) found that plant uptake was 140 kg K ha^{-1} when grazed and 415 kg K ha^{-1} when cut for silage. Because of high removal rates, most grassland requires K fertilizers to obtain and maintain optimum levels of dry-matter production.

Plant-available K in pastures is derived from soil minerals, senesced plant materials, animal excreta, and precipitation. The supply derived from soil minerals varies with soil type and environmental conditions (Havlin et al., 2005), whereas the amount of K recycled from senesced materials can vary with plant species and dry matter production. Uptake by tall fescue in Georgia was 64 kg K ha^{-1} yr^{-1} in the top growth and 34 kg ha^{-1} in the roots (Wilkinson and Lowrey, 1973). Significant amounts of K can be returned in animal excreta. Whitehead (2000) estimated 105 kg K ha^{-1} are returned in urine and 23 kg K ha^{-1} are returned in feces when dairy cattle graze at a density of 700 cow-days ha^{-1} yr^{-1}. In New Zealand, excretal returns of K ranged from 180 to 500 kg K ha^{-1} (Early et al., 1998). In contrast, K received in precipitation is relatively low, ranging from 2 to 20 kg K ha^{-1} (Whitehead, 2000). For example, rates were 5.1 to 6.3 kg K ha^{-1} yr^{-1} in Ohio (Owens et al., 2003) and 6.4 to 9.5 kg K ha^{-1} yr^{-1} in southwest England (Alfaro et al., 2003).

As with P, K fertilization of grasslands can be based on two main strategies: fertilizing the forage or fertilizing the soil. Most university recommendations use the strategy of fertilizing the forage, which is based on results of K response studies under various soil-K

Vegetated riparian areas remove nutrients before they can enter waterways.



levels. These studies are conducted under optimum N fertilization because the response to K depends in part on N availability (Kayser and Isselstein, 2005). In K-response studies forage is cut and removed instead of being grazed, usually ignoring recycling of K through senesced plants and animal excreta (mainly urine). As a result, fertilizer recommendations based on soil-K levels are likely too high for pastures under intensive livestock production (Kayser and Isselstein, 2005). Therefore, additional research is warranted to develop methods to account for K recycling for grazed grasslands. Accounting for K recycling may allow development of K balances (input–output) that are close to zero, thereby reducing K leaching losses. In addition, avoiding large, positive K balances is important to reduce Ca and Mg leaching, thereby reducing the occurrence of hypocalcemia and hypomagnesemia in grazing cattle (Early et al., 1998; Owens et al., 2003).

Potassium amendments for grasslands include commercial fertilizers as well as animal manures and biosolids. Livestock and poultry in the USA excrete 38% more K annually than commercial fertilizer K consumed. Further, the fraction of manure K collected for application is nearly 47% of commercial fertilizer K consumed (Table 5.3). Potassium availability in animal manures is usually assumed to be near 100% because K is present in the inorganic form (Eghball et al., 2002). Availability of K in composted livestock manure and biosolids has also been found to be near 100% (Wen et al., 1997). However, K utilization in a Wisconsin field study was only 73% for injected dairy slurry (Motavalli et al., 1989). Clearly, additional research on K availability in by-products is needed. In terms of placement method, K fertilizers are usually broadcast on pastures because the high solubility of K ensures relatively fast movement into upper soil layers (Sistani et al., 2003; Warren et al., 2008).

Code 590–Criteria Assessment

Can a nutrient budget be developed for N, P, and K based on all potential sources of nutrients including animal manures and organic by-products, wastewater, commercial fertilizer, crop residues, legume credits, and irrigation water? Development of a nutrient budget taking into account all sources of nutrients requires information on the nutrient availability of the different sources. As described above, there is sufficient information on N release from most sources, but there is considerable variability among state laboratory recommendations, especially for N availability in animal manures. Also, average values commonly used for N release from organic sources have been developed from data sets with large variability because of variations in mineralizable N and environmental conditions. Thus, research is needed to develop tools for fast determination of pools and rates of mineralizable N in organic sources, as well as models to estimate N mineralized with the use of real-time environmental data. Although such models have been developed (Zhang et al., 2002) there is a need to translate them into decision-support tools to guide nutrient application.

With regard to P and K budgets, there is information on the average availability from many sources, but additional research is needed to determine phytoavailability of P and K in organic sources such as biosolids, manures, and other organic fertilizers. In summary, nutrient budgets can be developed with the use of average availability values (VanDyke et al., 1999; Shepard, 2005), but additional work is needed to fine-tune N, P, and K availability based on rates of nutrient release. In addition, there is a need to develop on-farm nutrient budgets that quantify multiple pathways of nutrient input and loss over time under different management systems (Vitousek et al., 2009). These needs may be usefully served by developing more comprehensive models.

Another significant challenge in developing nutrient budgets is that, in most states, fertilizer recommendations for pastures are the same or nearly the same as they are for hay fields, even though about 75–90% of the nutrients consumed by grazing animals are returned in the manure (Whitehead, 2000). The lack of differences in fertilizer recommendations between pastures and hay fields is most likely because manure deposited by pastured animals is not uniformly distributed across the field in extensively grazed pasture systems. Numerous studies have shown that manure deposition is concentrated around shade trees, watering sources, and supplemental feed bunks

“fertilizer recommendations based on soil-K levels are likely too high for pastures under intensive livestock production”

“

Nutrient planning should use soil, tissue, and manure samples that are collected, processed, and analyzed...”

(Mathews et al., 1994; White et al., 2001; Dubeux et al., 2009). Therefore, most state fertilizer recommendations assume that lack of uniformity in deposition makes it nearly impossible to credit these nutrients properly to reduce fertilizer recommendations for extensive pastures.

However, a growing number of producers have adopted managed grazing methods where animals have access to limited land areas for short time periods (a few hours to a few days). In these grazing methods, manure deposition is more uniform than in traditional, extensive pasture management. Lory and Kallenbach (1999) estimated it might take more than 25 yr to have at least one manure pile per square meter of a pasture under traditional continuous grazing; with a 2-d grazing period this time would be reduced to less than 2 yr. With 1-d grazing periods, manure should be relatively uniformly distributed throughout the grazing paddock and nutrient credit in the form of reduced fertilizer recommendations should be developed. Currently, very few states have developed fertilizer recommendations for intensively managed grazing systems and it needs more research. It is also important to recognize that different forms of grazing management can affect the rate of forage growth, hence altering plant uptake of applied nutrients and affecting the risk that unused nutrients will be lost to the environment.

Can realistic yield goals be established based on soil productivity information, historical yield data, climatic conditions, level of management and/or local research on similar soil, cropping systems, and soil and manure/organic by-products tests? In several states, fertilizer recommendations for grasslands are based on nutrient addition response studies instead of yield goals. However, those states that use yield goals usually provide guidelines for setting realistic yield goals in extension bulletins or fact sheets. These guidelines range from using the average yield from the last 3–5 yr plus an increase of 5–20%, to a 5-yr average plus one standard deviation, or to 80–90% of an estimated attainable yield based on a simulation model and weather data from several years (Fixen, 2006). The need for long-term yield data for either N response curves or forage yields is based mainly on weather variability across years. Because of this variability and current limitations in long-term weather prediction, a promising approach that needs additional research is the adjustment of yield goals or N rates within the growing season by using real-time tools like sensors and weather-driven models.

Do nutrient management plans need to specify the source, amount, timing, and method of application for each field to achieve production goals while minimizing movement of nutrients and other potential contaminants to surface and/or ground water? The available literature indicates that nutrient management plans that take into account the source, timing, method of application, and level of management for each field (or within a management zone) lead to lower N and P application rates and lower N and P losses (VanDyke et al., 1999; Beegle et al., 2000; Delgado et al., 2005; Shepard, 2005).

Can nutrient planning be based on current soil- and tissue-test analyses that are collected and processed with the use of land-grant approved practices and by labs meeting performance standards? Nutrient planning should use soil, tissue, and manure samples that are collected, processed, and analyzed with the use of practices approved by land-grant universities (Peters, 2003).

Are all soil-test analyses less than 5 yr old useful for nutrient planning? For fields that have been receiving recommended rates of nutrient applications, 5-yr-old soil tests may be adequate for nutrient planning. However, in fields that have been receiving high rates of manure or biosolid applications, soil-test P may be increasing rapidly (Pierson et al., 2001) and as a result, 5-yr-old soil tests may not be adequate for nutrient planning. In those situations, more recent soil tests may be needed. Additional research is needed to determine the short-term increase in soil-test values under those conditions.

Is it possible to predetermine analyses pertinent to monitoring and amending the annual nutrient budget? The use of sensors and inorganic N analysis during the growing season may allow adjustment of the N budget based on forage and soil conditions (Flowers et al., 2004; Meisinger et al., 2008), but additional research would be needed to adapt these tools to use in grasslands.

Is it important or critical to adjust soil pH to an adequate level for (effective) nutrient availability and utilization? Because the availability of some nutrients increases with pH while the availability of other nutrients decreases with pH, a pH range of 6–6.5 is usually where most nutrients are adequately available (Havlin et al., 2005). In addition, forages have different acidity tolerance. Therefore, soil pH should be adjusted, taking into account crop tolerance to acidity and nutrient availability.

Are nutrient application rates based on land-grant university recommendations using soil tests, yield goals, and management capabilities adequate for nutrient planning? Land-grant university recommendations are typically based on many years of research and as such provide an excellent approach to develop adequate nutrient management plans that lead to reduced nutrient losses (Lawlor et al., 2008). However, data collected from row crops suggest that in many cases these recommendations lead to overapplication of nutrients indicating there is the potential to reduce nutrient applications by up to 20% without compromising yields. Further, manures are commonly applied to pastures naturally or from storage areas. These manures vary in distribution on the pasture, quantity per year, nutrient concentrations, and nutrient availability, making planning more difficult.

Should N applications using commercial fertilizers match recommended rates for pastures/haylands as closely as possible? Application of commercial N fertilizers should match recommended rates as closely as possible to minimize nutrient losses (VanDyke et al., 1999; Delgado et al., 2005).

Is conservation management unit (CMU) risk assessment (using the appropriate tool) needed for manure and organic by-products to adjust the amount, placement, and timing of the nutrient application? When manure, organic by-products, or biosolids are applied, it is important to carry out a risk assessment with a tool such as the P index to adjust nutrient management so as to minimize the environmental impact of the added nutrients (Butler et al., 2010).

Are sampling and analytical methods for manures adequate for use in nutrient budgeting? Most methods for manure analysis determine the total amounts of nutrients in the manure, not the plant-available amounts (Peters, 2003). Consequently, in current methods for nutrient budgeting, a certain proportion of the total amount of each nutrient is assumed to be plant available. As discussed above, these proportions are interpreted differently among states, especially for N and P. Although some research has been conducted to develop methods for assessing amounts of available nutrients in manures, none is currently in use.

Research should continue to develop methods that can measure plant-available N and P in manures, organic by-products, and biosolids. Furthermore, methods that measure pools of mineralizable N and P could be coupled to simulation models to estimate rates of release using real-time environmental conditions during the growing season. Process-based models capable of such simulations are under development (Zhang et al., 2002; Giltrap et al., 2010).

Is a cumulative record of manure analyses (3 yr?) adequate for use as a basis for nutrient allocation to pastures and hay fields? Are "book values" on composition acceptable? Animal operations that use consistent forage/rations should achieve consistent manure composition in a relatively short time. Under these conditions, a 3-yr cumulative record of manure analyses should be sufficient to demonstrate consistent manure composition (Moore et al., 1995b).

Do manure analyses need to include nutrient and specific ion concentrations, percent moisture, percent organic matter, and salt concentration? An accurate method to determine mineralizable N in manures is not available, so current manure analyses for N include inorganic N (mainly ammoniacal N) as well as total N (Peters, 2003). For P and K,

"manures vary in distribution on the pasture, quantity per year, nutrient concentrations, and nutrient availability, making planning more difficult."

Nutrient management becomes more complicated when manure sources are used due to uncertainties in nutrient concentration and release rate.

total amounts are usually sufficient to estimate availability, although some research shows considerable variability in use efficiency of P and K (Motavalli et al., 1989; Wen et al., 1997; O'Connor et al., 2004). Determination of salt concentration or osmotic potential would be useful for manures that are to be applied close to the seed or young plants. Percent moisture (or percent solids) in manures and biosolids is essential for determining organic residue additions used when calculating soil erosion in RUSLE2. Otherwise this information would be needed only when nutrients are reported on a dry-matter basis.

Is it critical that the application rate of liquid materials shall not exceed the intake or infiltration rate, minimize ponding, and avoid runoff? Liquid manures applied at rates that exceed the soil infiltration rate can lead to ponding and runoff, as well as preferential flow to subsurface drain tiles, resulting in potential contamination of surface waters. In a literature review, Hoorman et al. (2010) suggested that liquid manure application rates should not exceed the remaining available water holding capacity of the surface 20 cm of soil at the time of application and suggested that application rates should be limited to 120,000 L ha−1 for surface-drained cropland, regardless of the remaining available water-holding capacity of the soil. Because many hayfields and pastures are on sloping land, the rates of intake and runoff may vary with ground cover at the time of application. These relationships need to be understood so better estimates of application rates can be made. Fields subject to cracking or root channeling are potentially more problematic, so great care must be taken when applying liquid materials in these situations.

Are N application rate guidelines for manure and organic materials sufficient? Nitrogen-availability algorithms have been developed to determine application rates for manures and other organic materials, including biosolids and composted materials, in most states.

Availabilities of biosolid N are generally consistent among states and are mandated via state or federal regulations. However, N-availability algorithms used are not necessarily supported by research. Nitrogen-availability algorithms for manures vary greatly from state to state and are probably the greatest single factor affecting optimum N utilization of these materials. In addition, most states do not include time of application relative to crop uptake for any organic materials. More research in this area is needed, especially in-season modeling based on actual weather data, to improve the ability to predict actual crop-N needs from year to year.

Are P application rate guidelines for manure and organic materials sufficient? Phosphorus-availability algorithms for manures and other organic materials are more consistent among states than are those for N. Phosphorus availability in biosolids can be greatly affected by the process used to concentrate solids (i.e., polymers, Fe, Al or Ca flocculating agents, biological recovery), yet these variables are not generally considered. Most states consider manure P to be 100% available relative to commercial fertilizer P, though this is not uniformly true. However, because manures are often applied on an N-rate basis, P deficiencies resulting from manure and other organic materials are rare. In addition, regular soil testing will help determine the impact of these P sources on plant available P. However, more research to determine P availability from manures and biosolids is needed to improve predictions for availability of P in manures, biosolids and other organic P sources relative to commercial P sources.

Are K and other nutrient application rate guidelines sufficient? Nearly all states assume that K availability from organic sources is 100% available relative to commercial fertilizer K, and this assumption is generally supported by research, though the availability of research data in this area is limited. Excessive K applications to grass pastures in the early spring under high soil moisture conditions can result in high levels of nitrate-N and lowered magnesium levels in the forage, which can lead to animal health problems like nitrate poisoning, hypomagnesemia and hypocalcemia in cattle. Although this is generally not a problem with biosolids, because of their low K content, the judicious use of high-K manures on pastures should be stressed.

Should timing and method of nutrient application (especially N) correspond with plant nutrient uptake characteristics considering relevant risk variables, including strategies to minimize nutrient losses? Ideally the timing of nutrient applications should correspond with plant nutrient uptake, especially for N, which is easily accomplished in the cases of commercial fertilizers and poultry litter. However, with liquid manure systems many producers do not have adequate storage to apply these manures with proper timing relative to crop N needs. Most nutrients are applied to the surface of grassland soils, making them subject to runoff losses, though some advances in subsurface application equipment for pastures have been made. Shallow injection would help reduce ammonia volatilization and P runoff potential, but surface applications are far more prevalent, because of equipment availability and the potential damage to plants from injected applications. There is also some risk that subsurface application increases loss of N by leaching.

Summary

The scientific literature supports the principle that judicious nutrient management can improve utilization of manure and other organic by-products while reducing environmental contamination. However, significant advances must be made before the use of these materials can be optimized. For example, although all states provide guidelines and recommendations for how to utilize manures on pastures and haylands best, the recommendations and algorithms lack consistency among states, even those within a similar geographic region. The greatest differences are those related to how the availability of manure N is calculated. These differences generally overshadow those in fertilizer recommendations among states within similar geographic regions, and are due at least in part to the lack of rapid methods to measure pool sizes and release rates of mineralizable N.

Biosolids availability algorithms are quite consistent across all states, but the algorithms are not necessarily supported by the scientific literature. Few states have developed fertilizer recommendations that are suitable for intensively managed grazing operations. Few significant advances have been made in equipment for surface application of manure, so uniformity of distribution remains a significant challenge. Overall, there is a need to develop on-farm nutrient budgets that quantify multiple pathways of nutrient input and loss over time under different management systems. This may require expanded efforts in modeling.

"there is a need to develop on-farm nutrient budgets that quantify multiple pathways of nutrient input and loss over time under different management systems."

MINIMIZE AGRICULTURAL NONPOINT SOURCE POLLUTION OF SURFACE AND GROUNDWATER RESOURCES

Surface Runoff

In many areas of the USA, nutrient runoff from pastureland can lead to water-quality problems including eutrophication of lakes and rivers, one of the most widespread water-quality impairments of U.S. waterways (USEPA, 1996). Several studies show nutrient concentrations in runoff increase with intensity of agricultural land use (e.g., Dillon and Kirchner, 1975; Owens et al., 1996; Pionke et al., 1996; Carpenter et al., 1998). Excessive manure applications were the greatest potential threat leading to eutrophication (Duda and Finan, 1983). Usually P is the limiting element for eutrophication in freshwater systems, whereas N usually limits in brackish and salt water (Schindler, 1977).

Phosphorus in runoff water can be dissolved (soluble) or particulate. Particulate P includes

“…best management practices for reducing P loss should focus on those that are aimed at reducing losses in surface runoff, whereas those for N should focus on reducing losses as leaching.”

all solid-phase forms, such as P associated with soil particles and organic matter. Because most pastures and haylands have very low rates of water erosion, the majority of P in runoff is the soluble form (Sharpley et al., 1992; Edwards et al., 1993; Shreve et al., 1995). Water-soluble P is the form most readily available for algal uptake (Sonzogni et al., 1982). Concentrations of P and N in runoff water from pastures can be very high following manure or fertilizer applications (Edwards and Daniel, 1992a, 1992b).

Although P-induced eutrophication is generally considered a freshwater phenomenon (Correll, 1998), there is strong evidence that P has a greater influence than N on hypoxia in the northern Gulf of Mexico (Lohrenz et al., 1999a, 1999b; Sylvan et al., 2006). In the Mississippi River plume, bioassays indicated P was limiting during periods of highest river flows (March, May, and July) (Sylvan et al., 2006). Likewise, river-influenced waters often exhibited very high N:P ratios (over 50), indicating P limitation of eutrophication during spring and early summer.

Although N limitation occurs in late summer and early fall, the rates of primary productivity are five times lower during this period than in spring (Lohrenz et al., 1999a, 1999b). Sylvan et al. (2006) concluded that P limitation in spring and summer was probably due to increasing N loading in the past 50 yr. As mentioned above, pasturelands are believed to be the source of the greatest amount of P in the Gulf of Mexico (NRC, 2009).

Although there are scant data on N and P losses via groundwater leaching from pastures, it is generally accepted that most N losses occur as nitrate leaching, whereas most P losses from pastures are due to surface runoff during storm events. Hence, best management practices for reducing P loss should focus on those that are aimed at reducing losses in surface runoff, whereas those for N should focus on reducing losses as leaching. The exception would be for pastures on sandy soils, in which case P losses from leaching would also be high.

Groundwater Contamination

Nitrate (NO_3) is the most soluble form of N and leaching can lead to elevated NO_3 levels in groundwater. Legal levels of NO_3–N in drinking water are 10 mg L^{-1} (USEPA), because higher concentrations can cause methemoglobinemia (blue-baby syndrome), a potentially fatal blood disorder in infants under 6 mo old. Groundwater contamination of NO_3 under pasture and hayland can occur from both inorganic-N fertilizers and from manure applications. However, because economic returns from hay or pasture are often lower than for row crops, overfertilization of N with commercial fertilizers is less likely than with manure, especially in areas of concentrated poultry or livestock production (Ritter and Chirnside, 1987; Kingery et al., 1993).

Pastures with a long-term history of poultry litter applications had much higher NO_3 levels in the soil profile than similar pastures that had not been receiving litter (Kingery et al., 1993). Marshall et al. (2001) found an average of 43% of manure N was taken up by plants in typical tall fescue pastures fertilized with poultry litter and concluded that of the remaining 57%, only 6% was lost via denitrification and NH_3 volatilization. Yet, in certain soils, NO_3 levels in groundwater at 1 m often exceeded 10 mg N L^{-1}. These results are consistent with Adams et al. (1994), who found groundwater NO_3 levels could exceed the EPA threshold following litter application rates of 20 Mg ha^{-1}, but remained below the critical level when application rates were below 11.2 Mg ha^{-1}. Similarly, NO_3 concentrations were less than the legal level in groundwater from fields fertilized with poultry litter at moderate rates (Moore et al., 2000).

Groundwater NO_3–N levels exceeded 100 mg L^{-1} for most of the year in two 4-ha fields used for winter loafing areas for 250 cows, stocked at 31 cows ha^{-1} on a dairy farm in NW Arkansas. This level of NO_3-N, which exceeds the EPA standard by ten-fold, may pose a significant health risk to humans if drinking water wells are located near the field. Calculations based upon N excretion rates per cow of 0.23 kg N day^{-1} (USDA-NRCS, 2008) indicated that daily direct deposits of N were equivalent to 7 kg N ha^{-1}. Based on using the area for 3–4 mo, the direct deposits during the winter were in the range of 620–820 kg N ha^{-1}. During warmer months the producer used these same fields as spray fields for effluent applications from his holding pond, with an average annual application of 280 kg N ha^{-1}. Hence, the annual total-N loading to the fields was probably between 900 and 1100 kg N ha^{-1} (Moore and Brauer, 2009). More research is needed to develop process-based models that can predict groundwater losses given known nutrient loadings, soil properties, and climate.

Management Strategies for Improving Water Quality

Best management practices for improving water quality associated with pastures can be of two types: 1) measures of nutrient source control, and 2) measures of nutrient transport control. Source control measures are practices that affect nutrient management planning, such as determining fertilizer or manure rates, timing and method of application, nutrient solubility, crop uptake, manure testing, soil testing, and manure treatments. Transport control measures are basically practices that reduce nutrient transport from the field such as proper grazing management, buffer strips, fencing, and other physical control structures.

Nutrient Source Control

Nutrient Management Planning. Prior to the 1940s, farms tended to be somewhat self-sufficient with respect to nutrients. Manure produced by animals was returned to the land on the same farm to meet crop requirements. This recycling of nutrients resulted in a more sustainable agricultural system than most current animal production systems, which often rely on import of grain produced in other areas of the country. By the 1990s, states responsible for finishing the majority of U.S. animals imported over 80% of the grain for their feed (Lanyon and Thompson, 1996). This disconnect in the nutrient cycle has resulted in transfer of nutrients, like P, from grain-producing areas to animal feeding areas causing P accumulation in those soils. Nutrient imbalances on animal farms can be worse where pastures exist, because most nutrients, like P, tend to be recycled within the system when consumed by grazing animals. Many of the vertically integrated animal production enterprises, such as the poultry industry, have developed in areas where land is not suitable to row-crop production and,

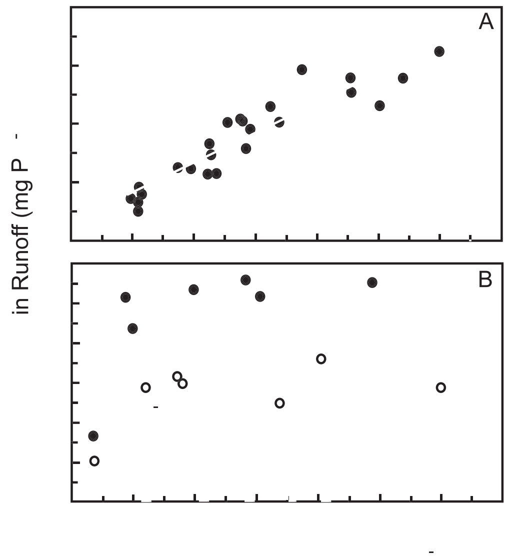


FIGURE 5.2. Effects of Mehlich III soil-test P on soluble-reactive P (SRP) in runoff water before (A) and after (B) litter application. Different rates of alum were added to the litter. Adapted from DeLaune et al. (2004a).

at least partially, where there is availability of lower-cost agricultural labor (Strausberg, 1995). The result is that the poultry industry is more concentrated where pastureland is the dominant agricultural land use.

Soils with elevated soil-test–P levels can contribute P in runoff in both dissolved and particulate forms (Sharpley, 1995; Pote et al., 1999a). Concentrations of P in runoff from pastures are highly correlated to soil-test–P levels (Sharpley, 1995; Pote et al., 1996, 1999a, 1999b; DeLaune et al., 2004a, 2004b; Schroeder et al., 2004). But this

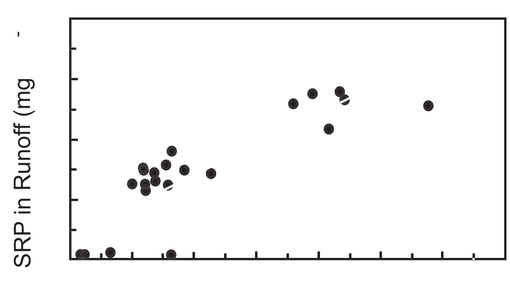


FIGURE 5.3. Effects of amount of soluble P in applied litter and soluble-reactive P (SRP) in runoff water. Adapted from DeLaune et al. (2004a).

"Best management practices for improving water quality....can be of two types: 1) measures of nutrient source control, and 2) measures of nutrient transport control."

"…poultry litter, particularly alum-treated litter, might be a more sustainable fertilizer than ammonium nitrate."

relationship only holds for pastures that have not been fertilized, and the relationship is site specific (Sharpley, 1995; Pote et al., 1996) making it difficult to delineate soil-test–P levels above which P-runoff losses are unacceptable.

When manure or commercial P fertilizers are applied to pastures, there is no significant relationship between soil-test P and P runoff (Sharpley et al., 2001b; DeLaune et al., 2004a). Instead, P losses from fertilized pastures are a function of the amount of *soluble P* applied through manure or commercial P fertilizer (Fig. 5.2). Soluble P is elevated in runoff from pastures fertilized with manure, often for a year or more after application. Dissolved-P values in runoff from pastures fertilized with poultry litter decreased slowly during the 19 mo after application before leveling out to a low rate (Pierson et al., 2001).

The most important factor affecting P runoff from pastures is the amount of water-soluble P applied as either commercial fertilizer or manure (Shreve et al., 1995; Moore et al., 2000; Sharpley et al., 2001b; Kleinman et al., 2002a, 2002b; DeLaune et al., 2004a, 2004b) (Fig. 5.3). Accurately accessing the potential for organic-P sources to contribute to P runoff requires an accurate measurement of soluble-P in manure. Until recently, the standard method of determining water-extractable P (WEP) from manure has been a 1:10 (manure:water) extraction (Self-Davis and Moore, 2000). However, recent work by Kleinman et al. (2002b) indicated there is a better relationship between P runoff and WEP in manure if a wider extraction ratio (1:100, manure:water) is utilized.

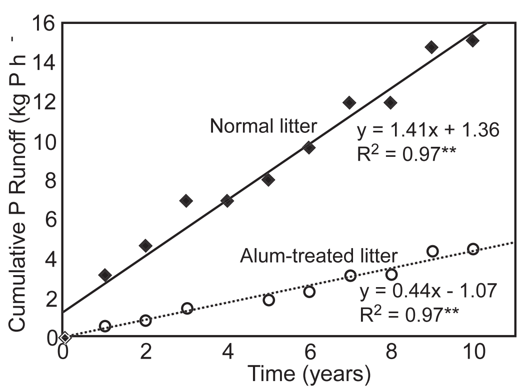


FIGURE 5.4. Cumulative P loads in runoff from paired watersheds fertilized with alum-treated and normal litter. Adapted from Moore and Edwards (2007).

As mentioned above, most P in runoff from pastures fertilized with animal manure is dissolved P. Hence, Moore and Miller (1994) hypothesized that P runoff from pastures could be reduced if soluble P in manure was precipitated using Al, Ca, or Fe amendments. Alum treatment of poultry litter reduced P runoff from tall fescue plots as much as 87% compared to untreated litter (Shreve et al., 1995). Also, forage yields and N uptake by tall fescue receiving alum-treated litter were significantly higher than for areas receiving untreated litter, probably because alum applications reduced NH_3 emissions from litter, which improved the fertilizer value (Moore et al., 1995a, 1996). Alum reduced P runoff from small watersheds by 75% over a 10-yr period (Fig. 5.4) and reduced P leaching compared with untreated litter (Moore and Edwards, 2007). These environmental benefits led the USDA-NRCS to make the use of alum a conservation practice standard (USDA-NRCS, 2009).

In a long-term study using small plots, forage yields were 6% greater with alum-treated litter than with untreated litter, and 16% greater than with an equivalent rate of ammonium nitrate (Moore and Edwards, 2005). Higher yields with alum-treated litter were attributed to the greater N availability, due to less NH_3 loss. Ammonium nitrate resulted in soil acidification and high exchangeable-Al levels in the soil by year 7. In contrast, soil pH increased with both alum-treated and nontreated poultry litter, resulting in lower levels of exchangeable Al than in the nonfertilized control. Aluminum uptake by tall fescue and Al runoff were not affected by fertilizer treatment. They concluded that poultry litter, particularly alum-treated litter, might be a more sustainable fertilizer than ammonium nitrate.

Fertilizer application rate, nutrient solubility in fertilizer or manure, application timing, and application method all influence nutrient runoff from pastures. The largest factor is fertilizer application rate; increasing rates result in more runoff of N and P (Edwards and Daniel, 1992a, 1992b; DeLaune et al., 2004a).

Timing of manure and fertilizer applications can also significantly affect the magnitude of nutrient runoff losses (Westerman and Overcash, 1980; Edwards and Daniel, 1993; Sharpley, 1997; DeLaune, 2002; DeLaune et al., 2004a). Concentrations of P in runoff from tall fescue plots decreased exponentially when the first runoff event occurred on the day of application (18 mg P L^{-1}) or was delayed for 49 d (3 mg P L^{-1}) (DeLaune, 2002). Concentrations of P in runoff were as high as 86 mg P L^{-1} on the day of fertilizer application and also showed an exponential decrease in concentration with time after fertilizer application (Owens and Shipitalo, 2006). One storm occurring soon after manure application accounted for the majority of the annual P load in runoff from a pasture (Edwards et al., 1996). Others have found similar results (DeLaune et al. 2004a; Schroeder et al., 2004).

Highest losses in runoff occur when manure applications are made during periods of the year when nutrient uptake is slow or the soil is frozen (Sharpley et al., 1998). Incorporation of poultry litter on perennial pasture with the use of a knifing technique reduced P and N losses by 95% (Pote et al., 2003). These findings were verified by Sistani et al. (2009), who also showed that losses of an indicator organism (*Escherichia coli*) in runoff were 100 times higher from plots when litter was broadcast on the surface compared with incorporation into the soil.

Most states do not allow application of manure within a certain distance of sinkholes and wells to prevent drinking water from being contaminated with pathogens, NO_3 and/or metals. Although we did not find any published research to support these setbacks, it is understood by the scientific community that such contamination is likely to occur without these measures.

Phosphorus Index. In 1999 the USDA and EPA developed a joint nutrient management strategy that called for comprehensive nutrient management plans for animal-feeding operations (AFOs) by the year 2008 (USEA and USEPA, 1999). The policy states that in fields where P losses are a problem, the management plan should be based on P content rather than N content of the manure. Each NRCS state office was given three management options for land application of P: 1) managing P based on agronomic levels, which are based on crop need, 2) managing P based on an environmental soil-test threshold, or 3) managing P with the use of the P-index approach (USDA and USEPA, 1999).

Although two of the three strategies rely on soil-test–P measurements, it is generally accepted by the scientific community that approaches using the agronomic or environmental soil-test–P threshold provide a poor assessment of risk of P runoff, because many other variables, like P transport, affect P losses from fields (Sharpley et al. 1996; Sims, 2000; Sharpley et al. 2001a; DeLaune et al., 2004b). For example, Pote et al. (1996) measured P loads of 0.05, 0.16, and 0.35 kg P ha^{-1} in runoff from plots with very similar soil-test–P levels (285–295 mg P ha^{-1}) (Sharpley et al., 2001b, reporting data from the study by Pote et al., 1996, although these data are not actually presented in the manuscript). The poor relationship between P loads and soil-test P was attributed to variability in runoff volumes (Pote et al., 1996).

Soil-test–P levels can be extremely high and not cause water-quality problems if leaching and/or surface runoff does not occur from the field. In fact, most of the annual P loads from agricultural lands come from relatively small areas of the landscape (Pionke et al., 1997), demonstrating the need to avoid a simple and universal approach (i.e., one number fits all) to nutrient management on all fields. Even when P transport is taken into account, soil-test P is a good predictor of P runoff only on unfertilized pastures; once manure or commercial P fertilizer has been applied, the soluble P in the applied P overrides P runoff associated with soil-test P (Sharpley et al., 2001a; DeLaune et al., 2004a, 2004b).

Realistic evaluations of potential non-point-source P runoff must consider both P transport (surface runoff, erosion and/or subsurface flow) and P sources (manure, fertilizer, and soil-test P)

"…in fields where P losses are a problem, the management plan should be based on P content rather than N content of the manure."

TABLE 5.5. Phosphorus index for assessing the vulnerability of a land unit. The sum of the weighted rating values is used to determine the site vulnerability. Multiply units of tons acre^{-1} by 0.446 to give MT ha^{-1} and units of lbs acre^{-1} by 1.12 to give kg ha^{-1}. Taken from Lemunyon and Gilbert (1993).

Site characteristic (weighting)	Phosphorus loss (rating value)				
	None (0)	Low (1)	Medium (2)	High (4)	Very high (8)
Soil erosion (1.5)	Not applicable	< 5 ton acre^{-1}	5–10 ton acre^{-1}	10–15 ton acre^{-1}	> 15 ton acre^{-1}
Irrigation erosion (1.5)	Not applicable	Tailwater recovery or QS < 6 for very erodible soils or QS < 10 for other soils	QS > 10 for erosion resistant soils	QS > 10 for erodible soils	QS > 6 for very erodible soils
Runoff class (0.5)	Negligible	Low or very low	Medium	High	Very high
Soil-P test (1.0)	Not applicable	Low	Medium	High	Excessive
P-fertilizer application rate (0.75)	None applied	1–30 lb P_2O_5 acre^{-1}	31–90 lb P_2O_5 acre^{-1}	91–150 lb P_2O_5 acre^{-1}	> 150 lb P_2O_5 acre^{-1}
P-fertilizer application method (0.5)	None applied	Placed deeper than 2 in with planter	Incorporated immediately before crop	Incorporated > 3 mo before crop or surface applied < 3 mo before crop	Surface applied > 3 mo before crop
Organic-P source application rate (1.0)	None applied	1–30 lb P_2O_5 acre^{-1}	31–60 lb P_2O_5 acre^{-1}	61–90 lb P_2O_5 acre^{-1}	> 90 lb P_2O_5 acre^{-1}
Organic-P source application method (1.0)	None	Injected deeper than 2 in.	Incorporated immediately before crop	Incorporated > 3 mo before crop or surface applied < 3 mo before crop	Surface applied to pasture or > 3 mo before crop

Q = flow rate of water introduced into the furrow. S = furrow slope.

in risk assessment. Therefore, it is fortunate that 47 states have opted to use a P index for writing nutrient management plans (Sharpley et al., 2003). These indices account for the risk of P runoff from both source and transport factors.

The P-index, developed by USDA-NRCS to account for risk to water bodies from both P sources and P transport, is a field-scale assessment tool (Lemunyon and Gilbert, 1993). The original P index consisted of an additive matrix of nine site characteristics involving P-source and P-transport factors (Table 5.5). Each of these was weighted numerically, with certain characteristics having a higher weighting factor than others. To obtain a P-index value, the P-loss rating value (Table 5.5) for each site characteristic was multiplied by its weighting factor and all nine characteristics were summed. The authors suggested that modification of the original P-index was needed to reflect local landscape conditions and management practices (USDA Soil Conservation Service, 1994).

Two significant changes have been made to the original P index. First, the relationship between P source and transport was changed from an additive approach to a multiplicative approach to more accurately reflect a field's vulnerability to P runoff (Sharpley et al., 2003). This approach allows fields that have little or no surface runoff or erosion to have a low P-index, even if soil-test P is extremely high. In the original index (Table 5.5), a field could have a very high P-index even though no surface runoff or transport occurs. Second, the risk of P runoff was more accurately quantified such that the P-index actually predicts edge-of-field P losses (DeLaune et al., 2004a, 2004b; Vadas et al., 2009).

Predictive P indices are much more difficult to develop than one that simply gives a relative ranking of risk, but are infinitely more useful. The biggest problem with P indices that include the relative risk of P runoff is that there is no simple way to test their accuracy. However, a predictive P index can be tested for accuracy by comparing the calculated P-index value to measured P loads from fields where runoff data exist.

In 2001, a P-index for pastures was developed for use in Arkansas only (DeLaune et al., 2004a, 2004b). Although most P indices are developed for row crop agriculture or for all agricultural settings, this index was developed specifically for application of poultry litter to pastures. The multiplicative P index combines the effects of P sources (soil-test P and soluble P applied from manure), P transport (soil erosion, soil runoff class, fertilizer application timing and method, flooding frequency, and grazing management), best management plans (fencing cattle out of streams, filter strips, and terracing), and annual precipitation to predict short-term and annual P loads in runoff (DeLaune et al., 2004a, 2004b). This is one of the few P indices in the USA that actually quantifies the risk of P runoff in terms of predicting the annual P-load in runoff from pastures or fields.

The Arkansas P-index for pastures is based on data from several hundred rainfall simulations on small plots cropped to tall fescue (DeLaune et al., 2004a, 2004b). The research was designed to determine how P runoff from pastures was affected by soil-test P, soluble-P levels in manure, poultry manure application rates, poultry diet modification using phytase and/or corn with high available P, and commercial fertilizer. The weighting factors for P sources were determined by multiple regression analysis from the rainfall simulation studies, where P loads in runoff were modeled using soil-test P and soluble P in manure. Weighting factors were 0.000666 for soil-test P and 0.404 for water-soluble P in manure (with both variables reported in lb P acre^{-1}). Commercial fertilizers were not included, based on the belief that their high cost would prohibit overapplication.

Validation data (Delaune et al., 2004b) showed the P index for pastures predicted edge-of-field P losses from paired watersheds accurately over a 6-yr period, with the slope between observed and predicted losses near one with an intercept near zero (Fig. 5.5). This index works well for pastures in western Arkansas, which typically have silt loam soils, and may work directly or be modified for other areas of the country. The irrigation erosion component present in the original P index developed by Lemunyon and Gilbert (1993) was not incorporated into the P index for pastures developed in Arkansas. Irrigation is typically much more important in row crop situations than in pastures.

Nitrate Leaching Index. Leaching of NO_3 through the soil and into groundwater is of concern for human health and the environment. It is impossible to eliminate NO_3 leaching totally (Pratt, 1979), but best management practices such as proper nutrient management can be used to minimize this problem. Some factors that affect NO_3 leaching are similar to those affecting P runoff, including fertilizer and manure application rate (Delgado et al., 1996; Kirchmann and Bergstrom, 2001; Meisinger and Delgado, 2002), type of crop being grown (Delgado et al., 1998a, 1998b) and N content of irrigation water (Bauer et al., 2001).

One tool used for nutrient management planning by NRCS is the NO_3 leaching index (Schaffer and Delgado, 2002). Several NO_3 leaching indices are available that attempt to model leaching as a function of fertilization, climate, soils, crops, and other factors that affect water and nutrient movement in the soil (Pierce et al., 1991; Shaffer et al., 1991; Williams and Kissel, 1991). Although some indices are relatively simple, others are complex and require daily time steps of weather data. As a result, some states like Texas have developed spreadsheets that calculate the N leaching value for the major crops grown in the state. In this case, the leaching index is a relatively simple combination of the percolation index and the seasonal index (USDA-NRCS, 2004).

"It is impossible to eliminate NO_3 leaching totally, but best management practices... can be used to minimize this problem."

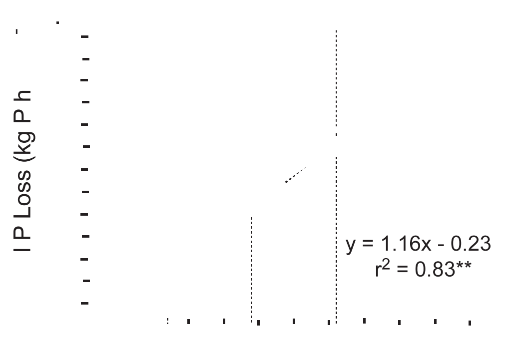


FIGURE 5.5. Relationship between annual-P loss in runoff predicted by the P-Index for Pastures (1:1 line) and that observed from small watersheds receiving natural rainfall and annual applications of poultry litter. Adapted from DeLaune et al. (2004b).

“Phosphorus loads from paddocks harvested mechanically were lower because of reduced runoff volumes compared to grazed paddocks”

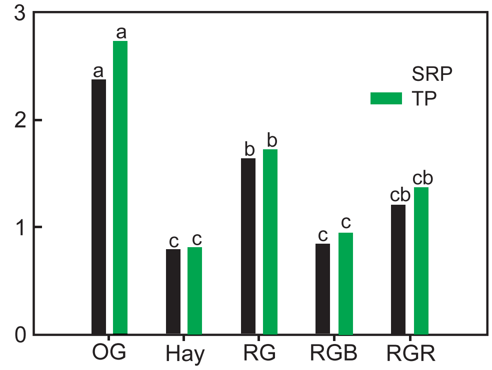


FIGURE 5.6. Effect of grazing management practices and hay harvest on annual runoff loads of soluble-reactive P and total P from small watersheds (OG = overgrazed, HAY = hayed, RG = rotationally grazed, RGB = rotationally grazed with a buffer strip, RGR = rotational grazed with a riparian buffer strip). Adapted from Pennington (2006).

Nutrient Transport Control

Proper Grazing Management. Runoff and P loss from pastures can be reduced by grazing management practices by affecting soil hydrology and chemical properties of the soil and water (Gifford and Hawkins, 1976, 1978; Van Haveren, 1983; Tollner et al., 1990; Owens et al., 1996; Haan et al., 2003). Rotational stocking can reduce negative soil properties such as soil compaction when compared to intensive continuous stocking, resulting in increased forage yield and vegetative cover (Langlands and Bennet, 1973; Gerrish and Roberts, 1999; Franzluebbers et al., 2004). Improved forage growth reduces raindrop impact, increases infiltration rates, nearly stops soil erosion, and improves water quality (Duly and Kelly, 1939; Tanner and Mamaril, 1959; Warren et al., 1986a, 1986b; Owens et al., 1989; Owens et al., 1996).

Runoff of P and N were lower from rotationally stocked pastures than pastures that were overgrazed by continuous stocking, whereas runoff losses were even lower for paddocks harvested mechanically (Fig. 5.6) (Pennington, 2006). Phosphorus loads from paddocks harvested mechanically were lower because of reduced runoff volumes compared to grazed paddocks, which had compacted soil (higher bulk density) at the surface (Pennington, 2006; Pennington et al., 2009). Similarly, Sistani et al. (2009) showed that P and N losses were higher for grazed pastures than for those that were harvested mechanically. Likewise, Schepers and Francis (1982) found nutrient export from pastures increased as grazing intensity increased. However, the effects of grazing intensity on nutrient runoff are relatively minor (Emmerich and Heitschmidt, 2002; Mapfumo et al., 2002; Capece et al., 2007). Capece et al. (2007) and Sistani et al. (2009) concluded that elevated P loads in runoff are probably more related to fertilizer or manure use than to grazing management.

Most grazing studies cited above did not evaluate effects of grazing animals on spatial variability in manure deposition, impacts of animals on stream bank erosion, and/or direct deposits of urine and manure into streams, all of which can increase nutrient transport into aquatic systems (Peterson and Gerrish, 1996; Shirmohammadi et al., 1997; Zaimes and Schulz, 2002). As discussed above, grazing causes compaction, which reduces infiltration and increases runoff. Infiltration rates were lowest along paths made by cattle and areas adjacent to water tanks where animals congregate (Radke and Berry, 1993). Many cow paths lead directly to streams for access to water.

Rotational grazing and intensive grazing are currently gaining greater acceptance in the USA because productivity and profit margins tend to be better compared to continuous grazing (Undersander et al., 1993; Barnhart et al., 1998). Rotational stocking can reduce P runoff compared to continuous grazing (Olness et al., 1980; Ritter, 1988). Rotational stocking also results in more even distribution of manure nutrients compared to continuous stocking in which accumulation of P occurs in areas closest to water sources, shade and feeders (Mathews et al., 1994). A more detailed assessment of the effects of grazing intensity and grazing methods on the environment is in Chapter 3 of this volume.

Field-Scale Best Management Practices (BMPs). Currently, the P index for pastures used in Arkansas is being revised (Moore et al., 2009) to include nine field-scale BMPs, for which growers are given credit to reduce P runoff from pastures via use of fencing, field borders, diversions, ponds, terracing, filter strips, grassed waterways, riparian forest buffers, and riparian herbaceous buffers. The USDA-NRCS conservation practice standard numbers for these BMPs are 382, 386, 362, 378, 600, 393, 412, 391, and 393, respectively (USDA-NRCS, 2009).

Buffer strips or filter strips along a field boundary reduce nutrient runoff by providing a deposition area for sediments, providing an area for infiltration and adsorption of soluble pollutants (Chaubey et al., 1994) and, in some cases, promoting biogeochemical transformation of nutrients to inert forms. In most cases, vegetated buffer strips reduce both sediment and nutrient transport from pastures (Chaubey et al., 1994, 1995; Owens et al., 1996; Blanco-Canqui et al., 2004; Lowrance and Sheridan, 2005; Pennington, 2006). Vegetated buffer strips greatly reduced soluble-P and total-P concentrations in runoff from pastures fertilized with poultry litter (Fig. 5.7).

Riparian buffers have been touted as one of the most important factors for reducing non-point-source pollution in the USA (Gilliam, 1994). Strategically located, they can trap particulate-P and transform N compounds to inert N_2 gas. Cooper et al. (1995) indicated that riparian areas could be sources or sinks for P, because their assimilation capacity is finite. Although most publications indicate that vegetated buffer strips improve water quality, Stutter et al. (2009) found evidence indicating that the strips may have enhanced rates of soil-P cycling, resulting in higher levels of soluble P in the soil and greater P leaching into adjacent water bodies. Fiener and Auerswald (2009) found grassed waterways had little or no effect in reducing soluble-P movement from fields.

Keeping livestock out of waterways, especially during warm periods of the year, can reduce nutrient runoff from pastures. Cattle are 50 times more likely to defecate while standing in water than when on dry land (Davies-Colley et al., 2002). Fencing livestock from streams, a simple BMP, decreases nutrient and pathogen transport by reducing direct deposits of feces and urine into streams (Nelson et al., 1996; Shirmohammadi et al., 1997; Line et al., 2000). Fencing coupled with tree planting (i.e., riparian forested buffer) reduced P loads from pastures by as much as 76% (Line et al., 2000). Likewise, providing alternative watering sources, such as troughs and ponds, reduces direct nutrient deposition to streams. When given the choice, compared with a stream, cattle preferred drinking from a watering trough 92% of the time (Sheffield et al., 1997).

> “In most cases, vegetated buffer strips reduce both sediment and nutrient transport from pastures.”

Summary

There is wide disparity among P indices in the southern USA (Osmond et al., 2006). When P-index ratings for pastures were compared with a Mehlich-3 value of 75 mg P kg^{-1} receiving 9 Mg ha^{-1} poultry litter, Osmond et al. (2006) found five states gave a low rating (GA, LA, MS, NC, and SC), two states a medium rating (AL, FL), four states a high rating (AR, KY, TN, and TX) and one state with a very high rating (OK). In at least four states the P indices would allow application of 18 Mg ha^{-1} of poultry litter, even when buffer strips were not used. Most of these indices are not predictive and have not been adequately verified by science. It is not possible to determine which are reliable and which are not.

Policy makers with NRCS are now considering using an environmental soil-test–P threshold to give more consistency to manure application practices allowed by various state-P indices. Currently, some states allow high manure application rates regardless of soil-test P and/or other field conditions (Osmond et al., 2006). It

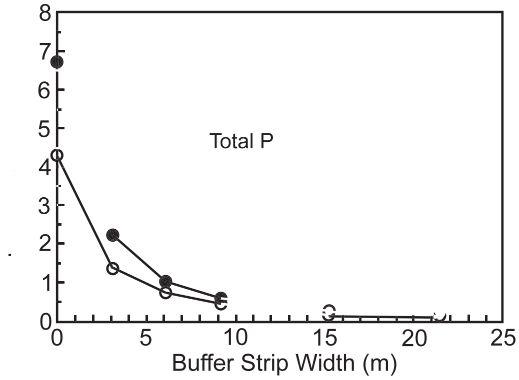


FIGURE 5.7. Effect of vegetative buffer strip width on concentrations of soluble P and total P in runoff water from pastures fertilized with poultry litter. Adapted from Chaubey et al. (1994).

"The best solution…is…a national P-index for application of fertilizers and manures for pastures and haylands"

is clear the soil-test–P threshold, a one-size-fits-all approach to nutrient management planning, is inadequate because it does not consider P transport (hydrology) or other factors. In addition, when manure or commercial-P fertilizers are surface applied on pastures, they totally overwhelm any effect of soil-test P on P runoff for 1–2 yr (Edwards et al., 1996; DeLaune et al. 2004a; Schroeder et al., 2004). Although variability in the way various state P indices function is a problem, scientific evidence does not support a soil-test–P threshold approach for pastures and hay fields that receive chemical or organic forms of fertilizer.

The best solution to this problem is to develop a national P-index for application of fertilizers and manures for pastures and haylands, which would likely be more specific for pastures and haylands. This index should quantify P runoff on an annual time step, similar to the P-index for pastures (DeLaune et al., 2004a, 2004b) and the P-index developed by Vadas et al. (2009). The P-index ratings of low, medium, high, and very high should have annual P loads associated with them, as is done with the P-index for pastures. Although developing the P-index will be a costly and time consuming process, the end product will provide a scientific basis for manure application that will be applicable from state to state.

A major knowledge gap in nutrient management planning for pastures is the lack of data on costs, benefits, and cost effectiveness of various BMPs. At present, agency personnel cannot determine costs to keep a kilogram of P from entering a river or stream with the use of various BMPs. Current methods utilized to determine funding available for cost sharing through EQIP (Environmental Quality Incentives Program) uses little scientific evidence for allocating funds to programs or states. Cost–benefit relationships for sites and regions need to be a part of the process to gain the most ecosystem benefit from the funds allocated to the practice.

Nutrient runoff and leaching from pastures cause water quality impairments such as

Rotational stocking leads to less nutrient runoff from pastures.

eutrophication in both freshwater and saltwater systems. In many cases, high nutrient loads in runoff and high levels of NO_3 in groundwater are associated with animal manure applications. Nutrient losses and soil erosion from pastures are increased by compaction caused by very intensive grazing, which decreases infiltration and increases surface runoff.

Strategies for improving water quality from pastures can be categorized as nutrient source controls or nutrient transport controls. Methods to reduce nutrient sources include practices such as determining proper rates, timing and methods of application of fertilizers or manures, reducing nutrient solubility through manure treatments such as alum, determining realistic crop yields, and using appropriate methods for manure and soil testing. Measures affecting transport of nutrients from the field include BMPs like proper grazing management, buffer strips, fencing, terracing and ponds.

Current P indices utilize information on P sources and transport, along with site characteristics, to determine the risk of P runoff from pastures or crop fields (Osmond et al., 2006). Although some P indices predict edge-of-field P losses, most do not, making it difficult to determine whether they accurately predict the risk of P runoff. There is a need to develop a national P-index that will accurately predict P losses in runoff over a range of crops and soil conditions. Currently, gross differences exist between P indices of various states that should be reconciled by science and not through use of an arbitrary soil-test–P threshold that does not consider hydrology or other transport factors.

The greatest knowledge gaps with respect to water quality from pastures and hayland involve comparative efficacies of BMPs and their cost effectiveness.

PROTECT AIR QUALITY BY REDUCING REACTIVE NITROGEN EMISSIONS AND THE FORMATION OF ATMOSPHERIC PARTICULATES

Loss of N from agricultural to aquatic ecosystems and the atmosphere is a great economic, ecological, and human health concern (Vitousek et al., 1997; Goolsby and Battaglin, 2001; Galloway et al., 2008; Schlesinger, 2009). When applied to agroecosystems but not removed in livestock or crop biomass the surplus N performs no beneficial agronomic function (Jarvis and Ledgard, 2002). Estimates of surplus N in grazed temperate grasslands range from 30% to 50% of N inputs (Carran and Theobald, 1995; Ledgard, 2001), which include N from fertilizer, exogenous manure, biological fixation, and atmospheric deposition.

Gaseous losses of reactive N as either nitrous oxide (N_2O), nitric oxide (NO), or ammonia (NH_3) lead to global, regional, and local environmental problems, respectively (Robertson et al., 2000; Mosier, 2001). Nitrous oxide is a stable greenhouse gas in the lower atmosphere that has been linked to climate change. In the upper atmosphere, N_2O is involved in reactions that deplete the protective ozone (O_3) layer (Crutzen, 1970). In the lower atmosphere NO leads to O_3 formation with negative human health consequences at the regional level. Volatilized NH_3 typically returns to the terrestrial environment within a kilometer of its source, possibly adding to eutrophication of the downwind sink. Ammonia is a strong base whose atmospheric concentrations are highly correlated with a strong odor (Pain and Misselbrook, 1991).

Pathways and Mechanisms of Gaseous N Loss From Pastures and Hayland

Anoxic soil conditions occur when the pore spaces become filled with water, with maximum rates of denitrification occurring at 60–80% water-filled pore space (WFPS). Hence, physical characteristics of the soil are a strong determinant of denitrification. Soils that are high in clay and poorly drained generally have more denitrification than sandy soils with good drainage. Emissions of N_2O occur only where denitrification is incomplete (i.e., NO_3 is not reduced completely to N_2) and a variety of factors, including WFPS, availability of labile C, and availability of NO_3 influence this process.

Given these abiotic controllers, it is clear that management exerts a strong influence on denitrification. In particular, N amendments,

“Strategies for improving water quality from pastures can be categorized as nutrient source controls or nutrient transport controls.”

“

it is desirable to retain as much N as possible in the agroecosystem for future primary production and to minimize the need for N inputs.”

soil disturbance, and compaction have strong direct effects, mainly through chemical and physical alteration of soil conditions. Management that influences plant productivity constitutes an indirect effect by altering the abundance of organic matter as a C source for denitrification. Even in the absence of management that stimulates N_2O fluxes (e.g., additions of N or water via fertilizer application, irrigation, or grazing), significant background levels of N_2O are exchanged between soils and the atmosphere in grasslands. For many years it was believed that N_2O was only emitted as a result of denitrification, but recent evidence points to significant levels of N_2O transfer from the atmosphere into soils under highly reducing conditions (i.e., WFPS > 90%) (Neftel et al., 2007). Conversely, anoxia can be found within aggregates and biofilms in soils that are not saturated, so some N_2O exchange occurs even when bulk soils are relatively dry (e.g., see Neftel et al., 2007).

Because N_2 is nonreactive and constitutes about 78% of the atmosphere, it is often ignored in measurements of denitrification. From an environmental perspective, it is considered to be the most benign loss of N that would otherwise be used by forages or pasture plants. Although N_2 can be fixed by legumes into useful form for return to the crop and soil, from an agronomic perspective, it is desirable to retain as much N as possible in the agroecosystem for future primary production and to minimize the need for N inputs.

Volatilization describes the loss of soil-solution N to the atmosphere from the conversion of NH_4 to gaseous NH_3. This reaction occurs at pH > 7.5. In more acidic conditions N in the NH_4 form is favored over that in NH_3, which reduces the likelihood of conversion and volatilization. Once formed, most NH_3 loss to the atmosphere is by diffusion, which is enhanced by coarse soil texture, low soil water content, high temperature, and high wind speed (i.e., the drivers of evaporative demand). As NH_3 becomes airborne, it either quickly dissolves in water, forming NH_4 that can be taken up directly by leaves or be redeposited to soil, where it is available for biotic uptake (Asman et al., 1998; Ferm, 1998; Aneja et al., 2001).

Conservation Management Practices Affect Gaseous N Loss

Alternative grazing and forage systems affect NH_3 emissions directly by altering vegetation structure and growth. Plant species differ in their ability to absorb or emit NH_3 (McGinn and Janzen, 1998), so forage species composition could affect atmospheric NH_3 concentrations. Grazing ruminants respond to a variety of dietary (e.g., forage N content and degradability) and metabolic factors (e.g., growth promoters, lactation) that affect production of urea, its recycling to the digestive tract, and its excretion in urine.

The nutrient management practice standard (590), which is identified for both N_2O and NH_3 loss, calls for managing the amount, source, placement, form, and timing of the application of plant nutrients and soil amendments. Below we review the scientific support based on research conducted on U.S. pastures and haylands. The *Web of Knowledge All-Database* searches (input terms were pasture, N_2O, NO, NH_3) located 47 studies of N_2O, NO, or NH_3 fluxes from nonrangeland pastures, but only 7 were based in the USA. Five of these were conducted on bermudagrass pastures and two on pastures dominated by tall fescue.

N Inputs

In general, pasture applications of inorganic N consist of ammonium nitrate or ammonium sulfate that are spread in a relatively uniform manner with mechanical devices. Typical recommendations for temperate pastures are 50–100 kg N ha^{-1} yr^{-1} split into early spring and early fall applications to supply nutrients just prior to rapid growth periods. Organic N comes from grazing animals directly as excreta, as manure removed from confined animal areas, or as solid-phase urea (Saggar et al., 2004). Manure, spread mechanically, is distributed more uniformly on the land area than direct excreta, but the actual uniformity depends on the liquid–solid ratio of the manure that changes dispersion properties, which can be quite variable. Nitrogen inputs from excreta and the subsequent effects on N losses to the atmosphere are extremely patchy at scales from centimeters (Koops et al., 1997) to hundreds of meters (Jackson et al., 2007). This spatial heterogeneity makes it very difficult to quantify N losses to the atmosphere accurately (Groffman et al., 2006).

Gaseous N losses from pastures receiving mineral-N fertilizer were 50% of those from urea fertilizer when applied at the same N rate (Colbourn, 1992; Eckard et al., 2003; Muller and Sherlock, 2004). These authors concluded that the additional N_2O loss from urea was likely from nitrification during N transformation from NH_4 to NO_3, whereas most of the loss from the ammonium nitrate fertilizer was from denitrification. A U.K. study on cool-season pastures showed that total denitrification losses were directly related to N application rate, irrespective of the fertilizer type (Ellis et al., 1998). However, N_2O loss was not solely related to N application rate because the type of N applied modified the response over time; N loss as N_2O was greater from manure slurry than from mineral fertilizer.

Diluting swine manure with water, either during or immediately following application to pastures in Nova Scotia, reduced NH_3 emissions by 41% and 45%, respectively (Mkhabela et al., 2009). These results align with many others. Other approaches to reducing volatilization such as manure injection are not feasible on intact pastures because the dense sod inhibits mechanical insertion and increases risk from degradation of soil organic matter and forage productivity. Direct N_2O losses were not affected by several management treatments, but indirect N_2O loss was a large pathway (Mkhabela et al., 2009).

Emissions of N_2O from pasture soils to the atmosphere generally peak soon after N application (Hutchinson and Brams, 1992; Thornton et al., 1998; Sullivan et al., 2005; Bender and Wood, 2007; Tiemann and Billings, 2008), but waned within a few days. Only one article reported that N inputs from broiler litter did not result in increased N_2O emissions (Marshall, 1999). In some cases, applied N did not increase N_2O emission above control plots until subsequent rain occurred (Hutchinson and Brams, 1992). The sum of these pulsed losses always accounted for only 0.3–5% of the N applied. Levels of N_2O–N losses were roughly equivalent to the 1–6 kg N ha^{-1} yr^{-1} entering eastern U.S. ecosystems via wet and dry deposition each year (http://nadp.sws.uiuc.edu/).

Of the studies cited above, only Thornton et al. (1998) and Hutchinson and Brams (1992) measured NO fluxes on pastures, which were greater with N amendments than without. However, Thornton et al. (1998) found < 0.4% of N applied was lost as NO–N, while Hutchinson and Brams (1992) estimated 3.2% of applied N was lost. In contrast, Sullivan et al. (2003) estimated 8–31% of the N applied as swine effluent was emitted from soils as NH_3.

Effects of N Form on Gaseous N Loss

Two U.S. studies explicitly compared N emissions after organic- and inorganic-N amendments to pastures (Sullivan et al., 2005; Bender and Wood, 2007). With equal N application rates of ~ 112 kg N ha^{-1} yr^{-1}, N_2O emitted from pastures receiving swine effluent was double that from ammonium nitrate, which represented about 2% of the total N applied (Sullivan et al., 2005). Bender and Wood (2007) found that 2–3% of the N applied as swine effluent was lost via gaseous emission, whereas < 1% of applied N was lost from pastures receiving ammonium nitrate. Emissions of N_2O and NO were lower when composted poultry litter was applied than when fresh poultry litter or urea was applied, but emissions for all treatments were ≤ 1% of the N applied. From the eight studies, highest NH_3 losses occurred (8–31% of N applied) when liquid swine effluent was supplemented with ammonium nitrate at 112 kg N ha^{-1} yr^{-1} (Sullivan et al., 2003). These studies indicate that NH_3 volatilization is the pathway of greatest gaseous N loss in pasture/hayland systems.

Timing of N Application Effects on Gaseous Loss of N

Bender and Wood (2007) showed that N_2O emissions spiked in the days immediately following application of swine effluent, but returned to background levels within 4–16 d. In contrast to ammonium-nitrate–amended soil, emissions from swine-effluent–amended soil were always greater from lighter than from heavier textured soils. On soils with high clay content, N_2O emissions were similar

"These studies indicate that NH_3 volatilization is the pathway of greatest gaseous N loss in pasture/hayland systems."

"Losses of gaseous N from pastures to the atmosphere are generally ≤ 5% of the N applied."

from swine effluent and ammonium nitrate amendments. No U.S. literature was found on season-of-application effects on gaseous N losses. Extensive New Zealand research shows nitrification inhibitors such as dicyandiamide (DCD) significantly reduced N_2O emissions from pasture (e.g., Di and Cameron, 2006).

Defoliation and Compaction Effects on Gaseous Loss of N

Disturbances such as grazing, harvest for hay, and burning are likely to influence gaseous N loss by affecting the amount of plant biomass, soil N, water-filled pore spaces, and soil temperature (Livesley et al., 2008; Uchida et al., 2008). About 40–80% of cattle excreta N is in urine, of which 10–95% is in the form of urea (Rodhe et al., 1997). Contact with water and the enzyme urease, which is found universally in feces and soils (Hoult and McGarity, 1986), causes rapid conversion of urea to gaseous NH_3. Excretion of urine by livestock creates hot spots where concentrated N combined with high water-filled pore spaces results in significant N_2O loss from nitrification, denitrification, or both (Lovell and Jarvis, 1996; Ambus et al., 2007).

Depending on weather conditions and existing plant cover, which both affect urease activity and the ability of leaves to take up volatilizing NH_3, more than 60% of the urine N can be volatilized to NH_3 (Ryden et al., 1987). Other estimates range from 10% to 90% (Woodmansee, 1978; Rotz et al., 2005) where dry conditions and low canopy volume generally contribute to NH_4 loss. Ammonia losses may be highest on medium-textured soils, because sandy soils result in more and quicker infiltration and heavier soils have greater cation exchange capacity (Whitehead and Raistrick, 1993). Greater standing plant biomass slows wind speed and allows greater plant absorption of the NH_3 volatilized from urine patches (Sommer and Hutchings, 2001).

Uchida et al. (2008) experimentally applied urine to soils with a range of aggregate size classes that had been packed to four levels of density. Emissions of N_2O during 30 d were similar among aggregate sizes if the soils were not packed. When packed, columns with small soil aggregate classes (< 1 mm) emitted more than columns with larger aggregate size classes. Colbourn (1992), using repacked soil cores, concluded that soils receiving no N inputs as urine or fertilizer emitted about 3 kg N ha^{-1} yr^{-1} of N_2O, whereas soils receiving urea or ammonium nitrate at 200 kg N ha^{-1} yr^{-1} emitted about 20 kg N ha^{-1} yr^{-1}.

On grazed cool-season pastures in eastern Kansas, N_2O emissions of about 20 mg N m^{-2} d^{-1} occurred when fertilized with ammonium nitrate whereas hayed plots fertilized the same had significantly lower N_2O emission (Tiemann and Billings, 2008). This was attributed to reduced soil organic-C availability (i.e., reduced capacity for denitrification) under periodic haying.

Plant Species Composition Effects on Gaseous Loss of N

The species mixture of a pasture community affects N dynamics directly, primarily via N_2 fixation by legumes (Ledgard, 2001). Furthermore, legumes and nonlegumes differ in their capacity to absorb soil N (Ridley et al., 1990). In some cases the presence of pasture legume species increases N_2O emissions (Niklaus et al., 2006), whereas there was no effect on emissions due to legume management in Australian temperate pastures (Livesley et al., 2008). No U.S. literature was found addressing this topic on pastures.

Dietary Manipulation Effects on Gaseous Loss of N

Manipulating the quantity and quality of excreta via dietary factors may be a useful approach to mitigating N loss from agroecosystems. For example, hippuric acid in the urine of livestock inhibits denitrification as much as 50% (Kool et al., 2006; van Groenigen et al., 2006). However, adding tannin to the cattle diet on rangelands of the northern Great Plains did not reduce N_2O flux from soils, even though the urine contained only half the N (Liebig et al., 2008). No literature was found addressing this topic on humid pastures of the USA, where the higher soil water content would be expected to increase emissions.

Summary

Losses of gaseous N from pastures to the atmosphere are generally ≤ 5% of the N applied. There is strong literature support that N_2O emissions are stimulated by N inputs and some support for increased NO and NH_3 losses with increased rates of N applications to pastures. There is modest support that organic sources of N promote greater gaseous N loss than does inorganic fertilizer N. No U.S. research was found on the effects of timing of N applications, species composition, or dietary factors on gaseous N loss from pastures. Some evidence exists that haying reduces gaseous N loss from fertilized cool-season pastures.

Significant regional gaps exist in the U.S. literature, because most support for the hypotheses above is from research conducted in the southern USA, mainly on warm-season pastures. This contrasts with western Europe, New Zealand, and Australia, where management effects on N dynamics of cool-season pastures have received two orders of magnitude more attention. Soil characteristics, management peculiarities, and climatic differences certainly will limit extrapolation from these data to the USA. Nonetheless, important principles and subjective generalizations from this vast literature should apply to pastures globally.

MAINTAIN OR IMPROVE THE PHYSICAL, CHEMICAL, AND BIOLOGICAL CONDITION OF THE SOIL

The scientific literature is replete with examples of the nutrient-supplying benefits of inorganic fertilizers, manures, and organic by-products to short-term production of pasture and hay crops. Yet over the long term, application of these nutrient sources may also influence soil physical, chemical, and microbiological properties that, in turn, affect pasture productivity. Fertilization may improve these soil properties indirectly through increases in soil organic matter (SOM) content resulting from greater productivity of shoots and roots. Soil organic matter content can also be increased directly with application of animal manure. Increasing SOM content via soil management is considered a worthy endeavor owing to its contribution to favorable soil fertility and physical properties. Because of its large role in soil function, SOM content is considered to be part of a minimum data set that defines soil quality (Doran and Parkin, 1996)

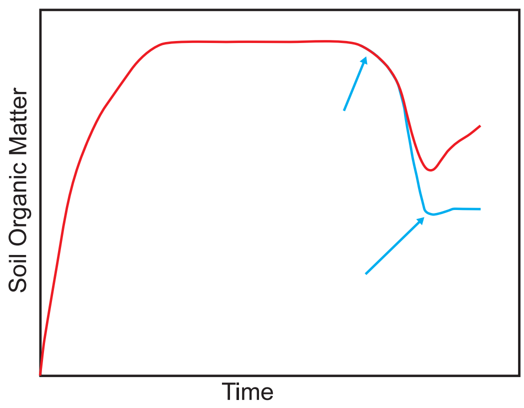


FIGURE 5.8. Conceptual soil organic matter equilibrium shifts with change in soil management. Time responses (over several years) are estimates to show relationships based on numerous published data sets. Adapted from Wienhold et al. (2004).

Long-term increases in SOM, which contains the largest terrestrial pool of C (Lal, 2004), can lower atmospheric-C concentration to mitigate climate change. Conversely, long-term application of nutrients, particularly those contained in animal manures and sewage sludges, at high rates may have negative effects on pasture productivity, the environment and on animal/human health owing to soil accumulation and/or escape of waste constituents to water bodies and the atmosphere.

When pastures are established, and then managed consistently, the underlying soil will shift to a new steady-state content of SOM that remains relatively stable through time under a given set of soil state factors (Jenny, 1980). Whether or not this new steady state is an improvement over the initial state depends on previous management or lack thereof (Fig. 5.8), and each soil has a SOM level that depends on climate and vegetation management. Clearing native grass or forestland to establish improved pasture may result in little change or a slightly lowered SOM steady state. However, pasture establishment on lands previously in row crops will likely result in a higher steady state of SOM concentration. In both cases, nutrient management plays a role in establishment of the new SOM equilibrium,

"When pastures are… managed consistently, the underlying soil will shift to a new steady-state content of SOM that remains relatively stable"

“SOM content may be increased via direct additions of organic matter and/or to increases in biomass production.”

and influences the physical, chemical, and biological condition of the soil. The NRCS CPPE matrix (Table 5.2) identified several ways that nutrient management alters soil properties, including: 1) SOM content; 2) contaminants associated with N, P, and K in commercial fertilizer, animal wastes, and other organics; 3) contaminants associated with salts and other chemicals; and 4) soil compaction. Herein we present U.S. research findings regarding these impacts on pasture soils, and identify knowledge gaps in the U.S. scientific literature.

SOM Depletion

Studies on changes in SOM content owing to management are necessarily long-term, because it takes several years for soil C and N equilibrium shifts to occur and stabilize. Unfortunately, most research grants awarded in the USA are short term (2–3 yr), causing reports on long-term management impacts on soil C and N to be scant. In addition, most U.S. scientific literature on this topic deals with effects of changes in tillage/cropping system or nutrient management, particularly manure sources, in row-crop agriculture. Such studies typically show equilibrium concentrations of SOM increase with reductions in tillage (e.g., Havlin et al., 1990; Wood et al., 1991; Wood and Edwards, 1992), increases in cropping intensity (e.g., Havlin et al., 1990; Wood and Edwards, 1992), and long-term application of manures (Wood and Hattey, 1995). These cropland studies imply that well-fertilized permanent pastures and long-term hay fields should attain higher equilibrium SOM concentrations than row croplands. Several studies outside the USA address these issues for pastures (e.g., Haynes and Williams, 1992; Sparling et al., 1994; Noble et al., 1999; Carran and Theobald, 2000), but little U.S. scientific literature exists regarding nutrient management effects on equilibrium contents of SOM.

The CPPE matrix (Table 5.2) indicates that compliance of the NRCS Nutrient Management Practice Standard (590) should result in slight to moderate improvement in SOM content owing to maintenance or enhancement of biomass production. Liebig et al. (2006), working in North Dakota, studied long-term (> 70 yr) grazing management impacts on soil variables related to SOM including soil organic C, total N, and particulate organic-matter C and N. Treatments were moderately grazed native vegetation pasture (MGP; 2.6 ha steer^{-1}; no fertilizer), heavily grazed native vegetation pasture (HGP; 0.9 ha steer^{-1}; no fertilizer), and fertilized crested wheatgrass pasture (FCWP; 0.4 ha steer^{-1}; 45 kg N ha^{-1} yr^{-1} as ammonium nitrate). Soil organic C and N in surface soil (0–5 cm) under FCWP and HGP were greater than that for MGP. The FCWP treatment also had greater amounts of particulate organic-matter C and N to a 30-cm depth than did HGP or MGP. These findings indicate that fertilization with N maintains to moderately increases SOM content over the long term in northern Great Plains pastures.

Similar results for soil organic C and N were obtained in North Dakota by Wienhold et al. (2001) in a study conducted a few km from the Liebig et al. (2006) site and had similar treatments. Results of both experiments support the notion that prescribed grazing (NRCS Practice Standard 528) interacts with nutrient management to influence SOM contents of pasture soils. In contrast with these studies, a study in Florida (Sigua et al., 2006) compared a natural wetland with a 63-yr-old bahiagrass pasture derived from a natural wetland histosol that had been fertilized every third year with 90, 13, and 37 kg ha^{-1} of N, P, and K, respectively. The SOM content of the surface soil (0–20 cm) beneath fertilized bahiagrass pasture was only 14% of that under reference wetlands, further emphasizing that direction of shift for the SOM equilibrium depends on initial soil condition. In a short-term study, a one-time application of 841 kg 16–20–0 plus 13% S ha^{-1} did not increase soil total N on a meadow renovation project in the Sierra Nevada of California (Kie and Myler, 1987) documenting that equilibrium shifts in components of SOM due to land management treatments may take many years to be expressed.

The SOM content may be increased via direct additions of organic matter and/or to increases in biomass production. After 15–28 yr of broiler-chicken litter (manure, bedding material, wasted feed, and feathers) applications to 12 tall fescue pastures in Alabama, there was significantly higher soil organic C and N, but lower soil C:N in surface soils (0–15 cm) than those found on matched nonlittered pastures (Kingery et al., 1994). In a short-term study, soil organic C and N of soils were studied under bermudagrass pasture either not fertilized or fertilized annually with ammonium nitrate or broiler chicken litter (Wood et al., 1993). Phosphorus, K, and lime were applied in nonlimiting amounts so only N was limiting. After 2 yr the contents of soil organic C and N were not different at any soil depth, although there was a trend for the broiler-litter treatment to have greater soil C and N concentrations than the ammonium-nitrate and control treatments. The results again suggest that equilibrium in soil organic C and N takes several years to show a measurable change.

Staley et al. (2008), working in West Virginia, conducted the only known U.S. study regarding conversion of deciduous forest to silvopasture and its impacts on SOM. Treatments included a forest (about 60 yr old; no fertilizer), a newly established silvopasture (2 yr old; lime, N, P, and K applied) converted from the forestland, and an established pasture (about 40 yr old; lime, N, P, and K applied). Two years after silvopasture establishment, organic C and N concentrations and C:N in the silvopasture soil (0–15 cm) were intermediate between forest and pasture, indicating rapid and substantial forest litter decomposition. However, soil under silvopasture had greater organic P than either pasture or forest, which the authors attributed to immobilization of fertilizer P with incorporation of forest litter.

These results illustrate that direction of SOM equilibrium shift with new soil management depends on initial soil condition and management, i.e., silvopasture established from deciduous forest lowered SOM content but, at least temporarily, increased concentration of soil organic P. Conversely, one could deduce from this study that establishing silvopasture on established pasture or croplands may indeed increase SOM contents. A study in southeastern Nebraska compared 130-yr-old forest established on pasture (not fertilized), present-day conventional corn–soybean rotation established on pasture (fertilized), and a never-tilled or fertilized pasture. Contents of soil organic C and total N increased significantly in the order of cropland < pasture < forest (Martens et al., 2003), which supports the findings of Staley et al. (2008).

Nutrient management affects the amounts of SOM and the amounts of microbial-mediated transformations in soil. Nitrogen mineralization and C emission via respiration are the two most commonly studied SOM transformation variables. In North Dakota, N mineralization rate in the upper 5-cm and the 5–15 cm soil layer of soil under fertilized crested wheatgrass was greater than that under nonfertilized native grass pastures (Wienhold et al., 2001). They suggested that immobilized fertilizer N under crested wheatgrass was more easily mineralized than the mature organic N under the native pasture. In addition, numbers of culturable micro-organisms, microbial biomass C, microbial biomass N, ratio of microbial biomass C to soil organic C, and ratio of microbial biomass N to soil organic N were similar among the pastures. This further suggested that differences in N mineralization were due to organic-matter quality rather than differences in microbial activity.

Contaminants—N, P, and K in Commercial Fertilizer, Animal Wastes and Other Organics

Code 590 indicates that proper nutrient application will provide a slight to moderate reduction of contamination risk from N, P, and K. This implies using appropriate N, P, and K rates according to soil-test laboratory recommendations and timing to match crop nutrient needs. Overapplication and/or poor timing can lead to escape of N, P, and K to the environment, which leads to soil, water, and air contamination. Of the three macronutrients N, P, and K, loss of N and P are of greatest concern because they have detrimental off-site water- and air-quality impacts, such as eutrophication of water bodies with dissolved and particulate N and P, NO_3 leaching to groundwater, and NH_3 and emissions of greenhouse gases.

Potassium generally has much lower environmental impact, although when out of balance with other nutrients it can cause grass tetany (hypomagnesaemia) in cattle (Ball et al.,

“Of the three macronutrients N, P, and K, loss of N and P are of greatest concern because they have detrimental offsite water- and air-quality impacts”

“Phosphorus has no gaseous pathway, which simplifies its nutrient cycle in comparison to N.”

2002). More specifically, high levels of soil K may depress plant uptake of Mg, even if there is adequate soil Mg, which leads to abnormally low levels of blood Mg (Tisdale et al., 1985). High levels of soil ammoniacal N can also depress forage Mg uptake and promote grass tetany (Tisdale et al., 1985). In Alabama, tall fescue pastures receiving long-term poultry litter had a higher ratio of K/(Ca + Mg), which is associated with grass tetany potential, than pastures receiving no litter (Kingery et al., 1993). However, the molar ratio did not exceed 2.2, which is considered the threshold for grass tetany.

When N and P in fertilizers, animal manures, or other waste products are applied in excess of that needed for maximum agronomic yield, or at the wrong time, they have increased risk for escape to the environment. The actual loss of surface applications of N and P can be from that accumulated and perhaps even transformed in the soil or the recently applied materials. Most studies have addressed the buildup of various organic and inorganic forms of soil N and P from applications of animal manures. Much less emphasis has been placed on buildup of these constituents from commercial inorganic fertilizers because they generally are easier to apply at accurate rates and they, unlike animal manures in many cases, are usually considered a direct cost to farmers.

In many areas of the USA, manure generated by concentrated animal feeding operations exceeds local pasture and hayland needs for N and P (Carpenter et al., 1998). Moreover, the bulky and relatively low nutrient concentration of animal manures limits the economic distance they can be hauled (Sharpley et al., 1993). Applications of animal manure N and P that exceed that needed for pasture and hayland production results in buildup of soil N and P. Further, concentrations of N and P in manures usually are not in ratios that match forage needs (Pant et al., 2004). For example, poultry litter applied on an N basis provides excess P to forages, resulting in a buildup of soil P that is subject to loss in surface runoff (Kingery et al., 1993; Sharpley et al., 1993).

Nitrogen can be lost from soil to the environment via NO_3 leaching, runoff of dissolved and particulate N, NH_3 volatilization, and emission of N_2O and dinitrogen (N_2) gas. These processes increase in magnitude as N in fertilizers, animal wastes, and other waste materials are applied beyond forage crop needs. In Georgia, broiler litter was applied for 2 yr to tall fescue pasture at cumulative rates of 0, 2733, 5466, 10,931, and 16,397 kg N ha^{-1} (Jackson et al., 1977). The 2733 kg N ha^{-1} was far beyond the tall fescue N requirement. They found little difference in soil total N due to litter application rate, and no difference when soil NO_3–N was subtracted from soil total N. They attributed the lack of difference among rates to poor incorporation of litter leaving much N on the soil surface. Soil profile NO_3 increased with higher rates and successive applications indicating the increased potential for NO_3 leaching when heavy N rates are applied.

Kingery et al. (1994) used 12 paired tall fescue pastures in the Appalachian Plateau physiographic region of Alabama (average annual rainfall of 1325 mm yr^{-1}) that had received long-term (15–28 yr) application of broiler-chicken litter or no litter to determine accumulation of soil total N and NO_3–N. Soil total N under littered pastures was higher than in nonlittered pastures to a depth of 30 cm. Soil total N under littered soils were elevated to 1 m depth without significant accumulations of NO_3–N (about 3 mg kg^{-1}) that increased with depth to 3 m (about 45 mg NO_3–N kg^{-1}). The data indicated that soils amended with broiler litter were more vulnerable to N-loss via runoff, leaching, and gaseous pathways. The NO_3 was depleted in upper portions of the soil profiles, but that in excess of tall fescue requirements was leached to the lower profile, representing a threat to groundwater quality. In a similar study on bermudagrass in eastern Oklahoma, 12–35 yr of poultry litter applications averaging 270 kg N $ha^{-1}yr^{-1}$ increased total N in the upper portion (0–20 cm) of the soil profiles compared with controls (Sharpley et al., 1993). However, NO_3–N did not accumulate deep in the soil profile, suggesting that litter N was not applied in excess of plant needs. Bermudagrass may have greater N uptake potential than tall fescue.

Phosphorus has no gaseous pathway, which simplifies its nutrient cycle in comparison to N. Owing to its sorption to soil, which is affected by the type of surfaces contacted by P in soil solution (Tisdale et al., 1985), leaching of P to subsurface water occurs in only a few soils. As with N, loss of P increases when P in fertilizers, animal wastes and other waste materials is applied beyond forage crop needs. Runoff losses of P have had much attention in relation to surface application of animal wastes sparking development of the P index for assessing site vulnerability to loss of P (Lemunyon and Gilbert, 1993) and for planning comprehensive nutrient management (Beegle et al., 2000). The index is now used to prevent applications of animal waste that could promote loss of P and N to the environment.

Several studies (Sharpley et al., 1993; Kingery et al., 1994; Lucero et al., 1995; Vervroot et al., 1998; Novak et al., 2000; Sharpley et al., 2004) have shown that application of animal manures to pastures or haylands beyond forage-P requirements builds surface soil-P to high levels. Most of these studies have long-term (decades) applications of manure, but some (Lucero et al., 1995; Vervroot et al., 1998) show soil-P levels build in only a few years when excess manure P is applied. Soil-P buildup also occurs in areas where grazing animals congregate in pastures (Graetz and Nair, 1995; Sigua and Coleman, 2006). As previously mentioned, part of the reason for soil-P buildup is the disparity in N:P ratios between manure and plant tissue, particularly when manure is applied according to forage-N requirements. Switching from nutrient management of poultry litter based on N to that based on P lowered soil-test levels of available P on high-P soils (Maguire et al., 2008). Nutrient management based on P lessens soil-P buildup where manures are applied, and lowers risk for loss of P to surface waters. But then alternatives need to be used to meet N requirements for grass pastures and hay fields, suggesting the value of legumes in these situations.

Cattle traffic in pastures can lead to erosion that results in loss of nutrients and soil quality.

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water-extractable soil P is the preferred analysis if environmental concerns are considered.”

Continuous application of manure alters the amount of soil P, soil-P form, and relative P availability. Sharpley et al. (2004) conducted a comprehensive study involving six grassland sites in New York, eight in Oklahoma, and six in Pennsylvania that received swine slurry, dairy manure, or poultry manure (40–200 kg P $ha^{-1}yr^{-1}$) for 10–25 yr. Compared to areas with no manure applied, manure applications resulted in 1) more soil P being in the inorganic form, 2) a shift in P chemistry from Al- and Fe-dominated complexes to Ca minerals with a concomitant increase in soil pH, and 3) a decrease in the proportion of water-soluble P:Mehlich-3 extractable P. This indicates water-extractable soil P is the preferred analysis if environmental concerns are considered.

Phosphorus leaching has not been considered a problem in most mineral soils where soil-test recommendations have been followed, but downward P movement in soil profiles “can occur in deep sandy soils, in high organic matter soils, and in soils where over fertilization and/or excessive use of organic wastes have increased soil-P values well above those required by crops” (Sims et al., 1998). Subsurface movement of P can contribute to eutrophication of surface waters that have a hydrologic connection with shallow water tables. Surface applications of manures to pasture or haylands cause accumulations of extractable P below the soil surface (Sharpley et al., 1993; Kingery et al., 1994; Lucero et al., 1995; Vervroot et al., 1998; Novak et al., 2000). Most of these studies have shown downward movement of P to 30–40-cm depths, but movement can be deeper, especially in sandy soils (Novak et al., 2000). The increase in Mehlich-3 extractable P after 10 yr of swine-effluent applications was reflected in higher dissolved-P concentrations of shallow groundwater.

The downward movement of surface applied manure-P in a Virginia soil with high P-fixation capacity was attributed to mobility of organic-bound P (Lucero et al., 1995), although only inorganic P was measured. Manure-P additions in Oklahoma lowered the P-sorption index (capacity of the soil to absorb P) below that of untreated soils, which allowed downward movement and increases in Bray-extractable P and total P to a depth of 30 cm (Sharpley et al., 1993). This finding, along with those of Sharpley et al. (2004) regarding the increased ratio of inorganic P:organic P where manures are applied over the long term, suggests that inorganic P moves downward as it fills adsorption sites.

Contaminants—Salts and Other Chemicals

Salts in fertilizers, manures, or other organic nutrient sources can have negative effects on pasture productivity if they reach excessive levels, generally with soil electrical conductivity (EC expressed as Siemans m^{-1}) values > 4 dS m^{-1} (U.S. Salinity Laboratory Staff, 1954). Salt injury from commercial fertilizers usually occurs in row crop situations when fertilizer is banded too close to germinating seeds, which is not a factor in pastures where fertilizer is typically broadcast applied. However, long-term application of manures or biosolids at high rates may build soil salts to levels that exceed the 4 dS m^{-1} threshold.

The CPPE for the NRCS Nutrient Management Practice Standard (Table 5.2) indicates that decreased application of excess nutrients will result in reduced salts. We found only two studies that considered effects of pasture nutrient management on soil salts. Greater soil electrical conductivity occurred after broiler litter had been applied for 15–28 yr to 12 tall fescue pastures in Alabama compared to matched pastures receiving no litter (Kingery et al., 1994). However, average EC in the upper 60 cm of littered pastures (0.08 dS m^{-1}) was well below the 4 dS m^{-1} threshold. After more than 70 yr of grazing nonfertilized native vegetation pasture and fertilized (45 kg N ha^{-1} yr^{-1} as ammonium nitrate) crested wheatgrass pasture, all soil EC values (range = 0.18 to 0.48 dS m^{-1}) were well below the 4 dS m^{-1} threshold (Liebig et al., 2006). These studies suggest that salt contamination is of limited importance in moderately fertilized pastures.

However, salt injury to corn has occurred with heavy applications of poultry manure (Weil et al., 1979), and such injury could occur in pastures following injudicious nutrient management.

Potential heavy-metal contaminants originating in inorganic and organic fertilizers include cadmium (Cd), arsenic (As), chromium (Cr), lead (Pb), mercury (Hg), nickel (Ni), vanadium (V), Cu, and Zn may build up in pastures soils after years of repeated applications (Mordvedt, 1996; Jackson et al., 2003). Of these, Cd is found in phosphate fertilizers and is considered the most important because of its negative human health implications, but estimates indicate that it would take over 1000 yr to reach an intolerable Cd limit (100 mg Cd kg^{-1} soil) at typical P rates (20 kg P ha^{-1} yr^{-1}) (Mordvedt, 1996). Moreover, applications of triple superphosphate or farmyard manure at agronomic rates for more than 60 yr in Missouri had no effect on uptake of Cd by timothy forage, even though slight accumulations of soil Cd had occurred (Mordvedt, 1987).

Animal wastes can contain high concentrations of As, Cu, and Zn owing to their use as biocides or growth promoters, particularly in poultry rations (Jackson et al., 2003). These elements build up in soil after years of manure applications (Kingery et al., 1994), are taken up by forage crops in greater quantities than where manure is not applied (Kingery et al., 1993), and may reach high-enough concentrations in runoff from pastures to cause water-quality problems (Moore et al., 1998). However, we found no reports indicating that soil accumulations of metals owing to manure or fertilizer application resulted in decreased pasture productivity.

Although not a contaminant per se, soil pH is altered by nutrient management in pastures. Ammoniacal fertilizers contribute to surface soil acidity via the nitrification process (e.g., Liebig et al., 2006), which is easily neutralized with agricultural lime (Tisdale et al., 1985). Animal manures contain mostly organic N that is typically easily mineralized to NH_3/NH_4 N plus antecedent ammoniacal N, yet long-term applications of animal manures increase soil pH due to basic cations, bicarbonates, and organic acids having carboxyl and phenolic hydroxyl groups contained in manure (King et al., 1990; Kingery et al., 1993, 1994; Sharpley et al., 2004).

Soil Compaction

Soils in permanent pastures and haylands can become compacted over time, owing to animal and vehicle traffic (Tanner and Mamaril, 1959) that results in retardation of water infiltration, gas exchange, root penetration, and nutrient transformations (Torbert and Wood, 1992; Lee et al., 1996). The CPPE (Table 5.2) for the NRCS Nutrient Management Practice Standard indicates that soil compaction is increased when traffic occurs on moist soils.

The most studied soil physical properties affected by soil management include bulk density, moisture holding capacity, water-stable aggregates, pore space, consistency limits, and strength. But most research involves cropland, particularly under various tillage systems, e.g., conventional tillage versus no-till. However, we found no published studies relating nutrient management to soil physical properties under pasture or haylands in the USA. We also found no published studies regarding amelioration of soil compaction in U.S. pastures or haylands. In Wales, vertical slitting of permanent pasture soils doubled forage production and uptake of N, P, and K (Davies et al., 1989). In New Zealand, mechanical aeration of pastures improved soil physical conditions (Burgess et al., 2000). These studies imply that occasional loosening of soils under permanent pastures will likely improve soil physical conditions, which in turn will increase nutrient uptake and forage production. These types of studies should be conducted on a range of U.S. soils.

Summary

Much more research on soil properties has been done in row crops, and more studies are needed in pastures and haylands, because the literature is not readily transferable. Most available information on soil properties for pastures and haylands evaluates effects from manure application, likely because of uncertainty associated with lack of accuracy in application rates and the imbalanced manure nutrient ratios in relation to plant nutrient requirements. However, the available studies at international locations indicate that nutrient management can impact soil properties under pastures and haylands.

The U.S. literature suggests that pasture fertilization with chemical fertilizers or animal wastes maintains or moderately improves SOM over the long term, which supports the stated impact promulgated by the NRCS CPPE (Table 5.2). The scientific literature

“available studies at international locations indicate that nutrient management can impact soil properties under pastures and haylands.”

Proper nutrient management increases sequestration of soil carbon and improves soil quality.

“In general, the practice standard purposes were supported moderately to strongly by the U.S. scientific literature.”

indicates that application of N and P on pastures in excess of plant needs increases the risk of off-site N and P escape. This may result in negative environmental impacts, indicating that proper application rates result in reduced risks of contamination from N, P, and K (Table 5.1). The CPPE for the NRCS Nutrient Management Practice Standard (Table 5.2) indicates that application of excess nutrients will result in increased salts, which may be true with grain crops, but available literature suggests salt buildup in pastures and haylands from long-term fertilizer or manure applications is negligible.

The limited U.S. literature on heavy metal buildup owing to fertilizer or manure application indicates that metals do accumulate in soils where poultry manures are applied long term, but not enough to reduce pasture productivity. Nutrient management affects soil pH, with ammoniacal fertilizers decreasing pH, whereas manure applications raise pH over the long term. The CPPE (Table 5.2) for the NRCS Nutrient Management Practice Standard indicates that soil compaction increases with traffic on moist soils, which is intuitive, but we found no U.S. studies relating nutrient management to soil physical properties in pastures and haylands. More research is needed for a complete understanding of potential relationships.

CHAPTER SUMMARY

This literature synthesis indicates that proper nutrient management is essential for sustained productivity and environmental compatibility in pasture and hayland systems. We have synthesized the available U.S. literature regarding the purposes of the NRCS Practice Standard 590, which are: 1) budget and supply nutrients for plant production; 2) properly utilize manure or organic by-products as a plant nutrient source; 3) minimize agricultural nonpoint source pollution of surface and groundwater resources; 4) protect air quality by reducing nitrogen emissions (ammonia and NO_x compounds) and formation of atmospheric particulates; and 5) maintain or improve physical, chemical, and biological condition of the soil. The U.S. scientific literature on nutrient management of pastures and haylands focuses mainly on production-oriented uses; there is some on environmental issues, but very little on environmental soil properties.

Along with this synthesis, we made subjective assessments of the level of support provided by the U.S. scientific literature for the purposes and criteria that characterize Code 590. In general, the practice standard purposes were supported moderately to strongly by the U.S. scientific literature (Table 5.1). However, this literature synthesis revealed several areas of nutrient management that require further research and development to ensure sustained and environmentally conscious pasture and hayland production in the USA.

With regard to the budgeting and supplying nutrients for plant production purpose of the 590 Practice Standard, much of the additional needed research and development is entwined with the second purpose, i.e., properly use manure or organic by-products as a plant nutrient source. This research and development need is owing to uncertainty regarding phytoavailability of nutrients contained in such nutrient sources. In particular, research is needed regarding 1) development of annual nutrient application rates for production and environmental preservation that account for excreta deposited on grasslands, 2) nutrient spatial distribution from deposited excreta, 3) tools for rapid determination of pools and rates of mineralizable N and P, and those of phytoavailable K in organic nutrient sources, and 4) improvement in manure application equipment to improve uniformity of distribution. Other issues related to budgeting and supplying nutrients for plant production that need further research include: 1) the impact of forage harvest, either by grazing or haying, on nutrient uptake; 2) fertilizer recommendations for intensively managed pastures; and 3) the use of sensors and weather-driven models for adjustment of nutrient application rates.

Simulation models coupled with aforementioned tools for fast determination of pools and rates of mineralizable N and P, and phytoavailable K, in organic nutrient sources could be powerful decision-support tools. Models would assist in transferability of the data among geographic areas and soil types. Collectively, these tools and analytical procedures would help optimize nutrient management in pasture and hayland systems in the humid areas of the USA.

Similar to the first two purposes of Code 590, uncertainty in minimizing agricultural non-point-source pollution of surface and groundwater resources from pastures and haylands is most prevalent where animal manures and organic by-products are used as nutrient sources. This uncertainty derives from aforementioned reasons, but is also owing to economic reasons, i.e., producers are much less likely to overapply costly commercial fertilizers than manures, especially in areas of intense animal production where finding areas for disposal is a challenge. The literature synthesis indicates that P runoff and NO_3 leaching occur mainly in pasture and hayland systems in regions of concentrated animal production.

It is imperative that BMPs appropriate to a particular CMU (nutrient transport control measures) and decision support tools such as nutrient management planning, the P index and the nitrate-leaching index (nutrient source control measures) be used to retard movement of nutrients out of pasture and hayland systems. A knowledge gap that needs to be bridged is the lack of data on costs, benefits, and cost effectiveness of various BMPs available for retarding nutrient loss from pastures and haylands. With regard to surface movement of P, there is a need to develop a national P index that will accurately predict runoff-P losses over a wide range of conditions. Research is also needed to improve existing models, and to develop new process models that predict nutrient losses from divergent nutrient loadings, soil properties, and climatic conditions.

Synthesis of the limited U.S. scientific literature regarding the Code 590 purpose of protecting air quality by reducing N emissions and formation of atmospheric particulates indicates that gaseous N losses from pastures to the atmosphere are ≤ 5% of the applied N. The literature suggests that these losses increase with increasing rates of applied N, and that organic N sources result in greater gaseous-N losses than do inorganic-N sources. More research is needed in various regions of the USA regarding pasture and hayland management impacts on N emissions, as most of the work to date has been done in the southeastern region.

Little research exists on effects of nutrient management in U.S. pastures and haylands on soil properties. Available U.S. scientific literature suggests that pasture and hayland fertilization maintains or moderately improves SOM contents over the long term. The U.S. literature also indicates that over application of N and P to pastures and haylands results in a buildup of these nutrients in the soil that may promote their escape to surface water and groundwater, and escape of N to the atmosphere. Salt buildup was found to be negligible and of no consequence in studies where nutrients were applied to U.S. pastures and haylands. The literature indicates that heavy metals do build up in pasture and hayland soils where animal manures are applied, but no influences from heavy metals on plant productivity were reported. There is some indication that manure applications have a slight liming effect.

No U.S. study was found that evaluated effects of nutrient management on soil physical properties on pasture or hayland. Some of these are known for row crops, but transferability of the effects to perennial pastures and haylands would be difficult. Thus, research on the influence of nutrient management in U.S. pastures and haylands on soil properties is needed to help ensure the sustainability of the soil resource.

In summary, nutrient management is a key component of overall pasture and hayland management. Nutrient management in U.S. pastures and haylands has critical implications for producers, the environment, and society at large. Overall, the production aspects of Code 590 are supported by U.S. research, and in most cases are scientifically sound. However, many aspects of nutrient management were identified that need further scientific support to ensure the future sustainability of our pasture, hayland, water, air, and soil resources. These are pointed out in the text and summarized in Table 5.1.

> "A knowledge gap…is the lack of data on…cost effectiveness of various BMPs available for retarding nutrient loss from pastures and haylands."

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