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CHAPTER 9

Nutrient Cycling in Forage Production Systems

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The Systems Approach to Nutrient Cycles

In most forage production systems, the nutrients needed for plant growth are provided by microbially mediated breakdown and release of plant-available mineral nutrients from dead plant tissues, livestock excreta, soil organic matter, and geochemically bound mineral forms. Even in fertilized forage systems, determining appropriate fertilizer application rates requires a “systems” approach on the part of the manager (e.g., Di and Cameron, 2000; Rotz et al., 2002). Fertilizer additions are simply one input in the system of inputs, outputs, pools, and fluxes that characterize nutrient cycling in a particular ecosystem.

In a systems approach, the size of the system is determined by the observer, and it is often management driven. It could be a particular field (Ball and Ryden, 1984; Trott et al., 2004), an entire farm (Watson and Atkinson, 1999; Rotz et al., 2002), a watershed (Howarth et al., 1996; Jordan et al., 1997), or, as is the case for global biogeochemical cycles, the entire earth (Vitousek et al., 1997; Smil, 2000). Whereas harvestable forage and livestock have traditionally been the outputs driving management decisions in forage systems, outputs of nutrients such as NO_3^- leaching, nitrous oxide (NO_x) gaseous emissions, and P runoff are becoming increasingly important (Sharpley et al., 1994; Vitousek et al., 1997; Stout et al., 2000).

Central to nutrient cycling in any ecosystem is the concept of mass balance. Nutrient inputs must balance nutrient outputs. Societal concerns over nutrient pollu-

tion in the environment and economic pressures to have profitable forage systems are forcing scientists and managers to document nutrient budgets more completely and precisely (Nord and Lanyon, 2003). The dynamics of water and carbon in forage systems can be analyzed with the same systems approach outlined here, but they are beyond the scope of this chapter (see Wedin [2004] for a review of grassland carbon budgets; see Wever et al. [2002] for a study of grassland water budgets).

A nutrient cycle or budget involves a network of pools (amounts) of a particular element, joined by fluxes (transfers) connecting those pools (Chapin et al., 2002). Although most elements have either a large atmospheric (e.g., C and N) or geologic (e.g., P and K) pool, the flux or transfer rate of elements from these pools into organic forms is usually low. The microbially mediated fixation of atmospheric N into organic forms by legumes is an obvious and important exception to that generalization.

Most discussions of nutrient cycling in forage systems emphasize the following pools: (1) soil organic matter, which, in more complex analyses, may be considered as multiple pools or fractions; (2) living plant biomass, including above- and belowground tissues; (3) plant residues (dead, relatively undecomposed plant tissues); (4) living animal biomass, the most obvious being the grazing animal but the most abundant being above- and belowground invertebrates and microbial populations; and (5) a small but critical pool of plant-available mineral forms of elements necessary for plant growth.

This last pool, the concentration of soil NO_3^- and NH_4^+ in the case of N, deserves special attention. This pool is often measured as an index of site fertility or nutrient availability, but, technically speaking, a pool or concentration is not a measure of nutrient availability, which is a flux or rate. Although the concentration of mineral soil N in a grassland may be very low on average, this tells us little about the rate at which N is being made available for plant uptake, which could be high in a fertile soil and low in an infertile soil (Hart et al., 1994; Robertson et al., 1999).

Simply put, pools have units of mass (kg ha^{-1} , g m^{-2} , mg kg^{-1} , etc.), whereas fluxes have units of mass transferred per unit time ($\text{kg ha}^{-1} \text{yr}^{-1}$, $\text{g m}^{-2} \text{d}^{-1}$, etc.). In a systems approach, residence times are the ratios of pools to fluxes and have units of time, because the units of mass cancel. Pools with short residence times are dynamic and are expected to change rapidly as management or environmental fluctuations affect the system. For example, consider a hypothetical grassland in which the only source of mineral N for plant uptake is net N mineralization, the flux from soil organic N to soil mineral N, and in which the soil organic matter pool of N contains 5000 kg N ha^{-1} , the soil mineral N pool contains 5 kg N ha^{-1} , and the annual rate of net N mineralization is 50 $\text{kg ha}^{-1} \text{yr}^{-1}$. In this case, the residence time of N in soil organic matter is 100 yr, whereas the residence time of mineral soil N is 0.1 yr or 36.5 days. The turnover rate is the inverse of the residence time, so in this example, the mineral soil N pool “turns over” 10 times, whereas only 1% of the soil organic N pool turns over per year.

Calculations of residence times assume a steady state or equilibrium. Although never completely valid, it is a useful starting point in analyzing system behavior (Chapin et al., 2002). In a steady state, pool sizes and flux rates are constant, and fluxes into and out of each pool must balance. This includes net fluxes into and out of the total system.

A system dominated by internal recycling of nutrients with relatively small inputs (e.g., fertilizer or N fixation) and outputs (e.g., leaching or animal and forage offtake) is considered relatively closed. As management intensity increases in forage systems, nutrient cycles inevitably become more open. Because nutrients such as N and P behave differently, one element in a system may have a relatively open nutrient cycle, whereas another element's cycle is relatively closed. For example, grasslands receiving animal manures may be managed to minimize N losses, yet still have significant P losses.

Why Does Nitrogen Frequently Limit Forage Production?

Nitrogen limitation is widespread, and by far N is the dominant nutrient constraint on primary production in most forage systems, although the reasons for this pattern

are not clear (Vitousek and Howarth, 1991). All terrestrial ecosystems have access to a near infinite pool of N in the atmosphere, which contains 78% N_2 gas. Many genera of bacteria are able to break the triple bonds of N_2 and reduce (“fix”) it to NH_4^+ . These bacteria include both symbiotic N fixers such as *Rhizobium* (associated with legumes) and *Frankia* (associated with woody species including *Alnus* and *Ceanothus*), and free-living N fixers such as *Azotobacter* and *Nostoc* (Paul and Clark, 1996). Despite the abundant source of N and a pathway for its incorporation into the ecological cycle, most natural and managed ecosystems are N limited.

Hypotheses for widespread N limitation involve the mass balance of inputs and outputs of N from terrestrial ecosystems. Until the advent of fossil fuel combustion, atmospheric inputs of N to ecosystems were generally small to negligible ($1\text{--}5 \text{ kg N ha}^{-1} \text{yr}^{-1}$). Sources of NO_3^- and NH_4^+ deposition included fixation in the atmosphere by lightning and volatilization from oceanic sources in coastal regions (Vitousek et al., 1997). Biological N fixation, in contrast, can potentially add $>200 \text{ kg N ha}^{-1} \text{yr}^{-1}$ to ecosystem N cycles (Fig. 9.1).

Biological N fixation has three general constraints. First, N fixation is expensive energetically. Thus, legumes fixing N divert energy from growth, giving them a disadvantage in competition for light with non-N fixers. N fixation is generally restricted to open, high-light environments such as deserts, grasslands, and savannas (Chapin et al., 2002). Leguminous trees in dense forests are rarely nodulated and probably contribute little to forest N cycles.

Second, biological N fixation requires significant amounts of other elements, including P, iron, sulfur, and molybdenum. In highly weathered and low-pH soils, these elements, although present, may be immobilized in a variety of geochemical forms. Increased grassland productivity in many regions may ultimately be limited by non-N nutrient constraints on legumes, especially P. Moore (1970) concluded that N is almost universally deficient in humid tropical and subtropical grasslands. However, “for the successful establishment of tropical grass and legume mixtures, every encouragement must be given to the legumes” (Moore, 1970). In tropical grasslands, which are often affected by low P and micronutrient availability, P and molybdenum fertilizer additions are critical to the establishment and maintenance of legumes and subsequent improvements in the N budget.

The third general constraint on N fixation is herbivory. Plant productivity in most temperate terrestrial ecosystems is N limited, and, as a consequence, the protein concentration of available forage is low. Legumes, which generally have high leaf N concentrations, are often targeted by both generalist herbivores, such as large ruminants, and specialist herbivores, such as many invertebrates. Reducing herbivory has led to increased legume abundance

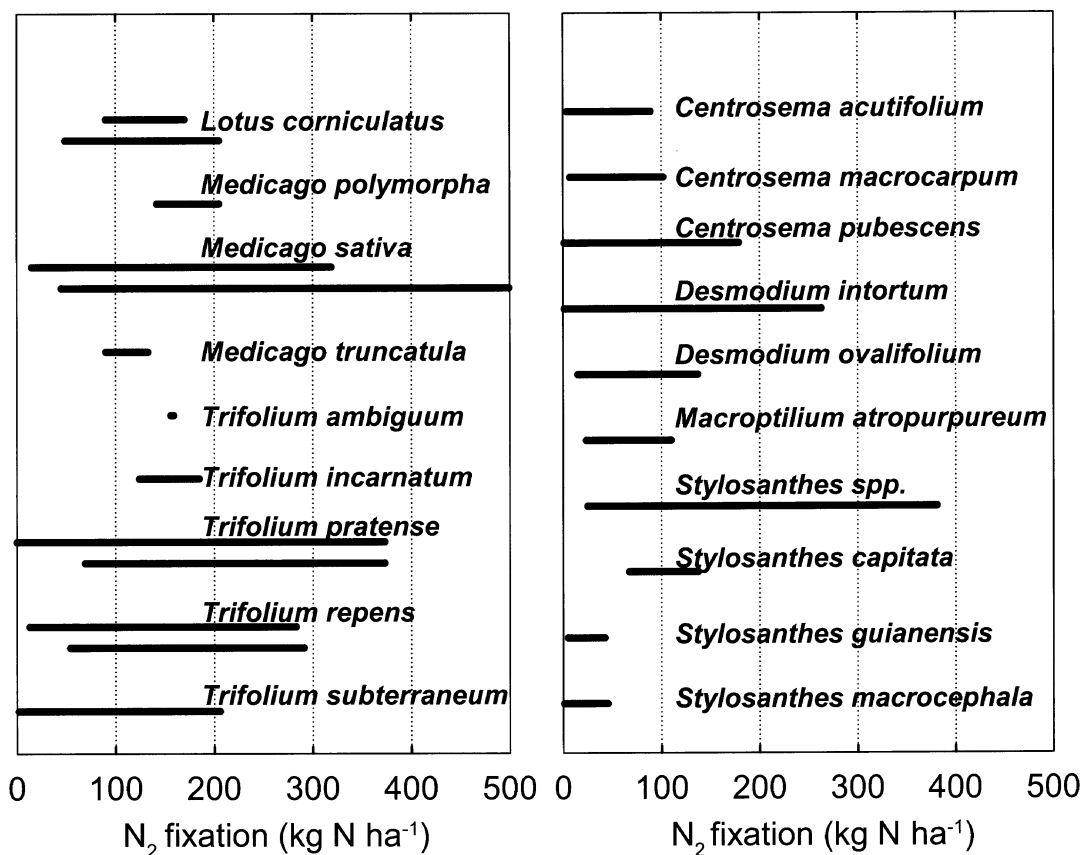


FIG. 9.1. Ranges of reported symbiotic N_2 fixation by temperate (*left panel*) and tropical (*right panel*) forage legumes (Russelle, in press). For temperate legumes N_2 fixation in mixtures with non-legumes is shown by the upper line of a pair; that in pure stands is shown by the lower line.

and greater N fixation in a variety of ecosystems. In areas with a long evolutionary history of grazing, such as Africa, legumes have often countered the threat of herbivory with physical (e.g., thorns) or chemical (e.g., alkaloid) defenses (see Chaps. 44 and 45).

Nitrogen loss from ecosystems may be as important as constraints on N inputs in explaining the chronic N limitation found in many temperate, terrestrial ecosystems. Because the N cycle is prone to both gaseous losses (NH_3 volatilization, nitrification, denitrification, combustion losses during fire) and leaching losses (NO_3^- and, to a lesser degree, dissolved organic N), it is inherently leakier than the cycles of P, K, Ca, and various micronutrients (Chapin et al., 2002). The availability of P or Fe may decrease over time in a particular ecosystem because those elements are chemically immobilized by reactions with soil and subsoil minerals, but, unless erosion or surface runoff occurs, those elements are rarely exported from the local system. In contrast, N losses inevitably increase

when ecosystems are disturbed (e.g., tillage, grazing, or cutting) and plant uptake from the soil mineral N (NO_3^- and NH_4^+) pool is disrupted (Vitousek and Howarth, 1991).

Nitrogen in the Plant–Soil System

In the long-term (centuries to millennia), net inputs and outputs of N play a large role in determining a particular ecosystem's fertility. In the short term, however, the supply rate of plant-available mineral soil N in an unfertilized ecosystem is regulated by soil biological activities. A diverse community of soil invertebrates, bacteria, and fungi is responsible for physically and chemically breaking down large organic molecules into smaller organic molecules, CO_2 , and various mineral nutrients (Swift et al., 1979). The list of new techniques for assessing the functional, taxonomic, and genetic diversity of soil communities is growing rapidly, but will not be discussed here (Sinsabaugh et al., 1999).

By far the largest pool of N (excluding the atmosphere) in grassland and forage systems is soil organic matter. The key flux from that pool is net N mineralization, defined as the microbially mediated release of NH_4^+ and NO_3^- from soil organic matter and plant residues. Various net N mineralization assays provide key insights into soil fertility and the degree to which N may limit plant productivity (Hart et al., 1994; Robertson et al., 1999). Mineralization, taken together with biological N fixation, N returned by grazing animals, and fertilizer or atmospheric N inputs, make up an ecosystem's N supply rate.

Treating net N mineralization as a single process or flux ignores much of the complexity of soil N dynamics. In the transition from organic matter to mineral N, organic substrates must first be broken down into soluble compounds that compose the dissolved organic N (DON) pool. Recent research has focused on DON for several reasons (Chapin et al., 2002; Jones et al., 2004). First, organic compounds must be broken down to DON before they can be absorbed and mineralized by microbes. Second, leaching losses of DON, although rarely measured, may be a significant component of the N budget in some ecosystems (Perakis and Hedin, 2002). Third, direct uptake of DON by plant roots or associated mycorrhizae has been documented in numerous ecosystems. Most of the reported cases involve uptake of neutrally charged amino acids such as glycine in cold, wet, and/or acidic environments such as tundra and conifer forests, where up to 65% of plant N uptake has been attributed to DON (Chapin et al., 2002). Because the direct uptake of DON short-circuits the role of N mineralization and the importance of NH_4^+ and NO_3^- availability, researchers are reconceptualizing N cycling where DON uptake has been documented. The importance of DON uptake in the N cycle of managed forage systems is still unsettled (Nasholm et al., 2000).

Microbial decomposers use DON as an energy source, respiring CO_2 and releasing NH_4^+ as a by-product. In aerobic soils, much of this NH_4^+ is subsequently nitrified by bacteria that oxidize NH_4^+ as an energy source. This is the key step in N mineralization; the total amount of mineral N released is called gross mineralization. Much of this NH_4^+ and NO_3^- may be reabsorbed or immobilized by the microbial community, however, in order to meet nutritional needs. If the C:N ratio of decomposing organic matter is high, N is limiting for microbes relative to labile organic C (their energy source), and there is little if any net release, or net mineralization, of NH_4^+ into the soil.

A C:N ratio of 25–35 is generally accepted as a critical ratio for net N mineralization from decomposing plant residues. This is somewhat higher than the C:N ratio of microbial biomass (generally about 10), but also reflects microbial growth efficiency (the proportion of consumed C incorporated into growth versus that respired) (Chapin

et al., 2002). At C:N ratios less than the critical level, the sink for NO_3^- and NH_4^+ provided by microbial immobilization disappears, and net mineralization increases sharply. The presence of this critical ratio or breakpoint in N cycling (the shift from immobilization to net mineralization) means that soil N availability and ecosystem N losses may respond nonlinearly to gradual changes in fertilization, herbivory, or other processes in forage systems (Wedin and Tilman, 1996).

Because of the strong role of plant tissue chemistry in regulating the N cycle, it is not valid to consider soil N availability as an abiotic or soil property in isolation from the characteristics of past and present vegetation. The C:N ratios of plant residues affect both the rate of decomposition and the balance between N immobilization by microbes and net N mineralization (Hobbie, 1992). In addition, the C chemistry of plant tissues strongly affects how it decomposes and contributes to formation of soil organic matter.

Lignin in aboveground tissues and suberin in roots are energetically expensive to break down for microbes and slow to decompose. Much of the polyphenolic ring structure of lignin is not broken down during decomposition, but instead is transformed and incorporated into large-molecular-weight amorphous compounds known as soil humus (Swift et al., 1979; Melillo et al., 1989). During this transformation, considerable N is tied up in the transformed C rings. Thus, although the C:N ratio of humus is quite low (10–20), the energetic costs for microbes using humus-bound N are high and its contribution to net mineralization is often low. As humus binds with clay or is protected in soil aggregates, its availability for decomposition and mineralization decreases further.

A number of decomposition studies suggest that approximately 20% of decomposing plant residues become stabilized as soil organic matter (Melillo et al., 1989). Using a simple model of N immobilization and soil organic matter formation, Knops et al. (2002) suggested that no net mineralization occurs in decomposing plant residues if they initially contain less than 0.75% N. All of the plant N becomes incorporated into soil organic matter. Although 0.75% N is low for aboveground plant tissues in managed cool-season pastures, it is typical for aboveground senesced tissues of unfertilized C_4 grasses. It is also a typical N concentration for roots in unfertilized stands of both cool-season and warm-season grasses. The low rates of net N mineralization observed in many grasslands and their ability to build soil organic matter rich in N are related, especially considering that roots make up over one-half of net primary production in most grasslands.

The N in soil organic matter in grasslands generally ranges from 5000 to over 20,000 kg N ha^{-1} . Net N mineralization rates generally range from 20 to 80 $\text{kg N ha}^{-1} \text{yr}^{-1}$, so the residence time of N in soil organic matter

would be centuries in most grasslands (Parton et al., 1987). Thus, soil organic matter does not appear to be a dynamic pool. However, numerous studies have shown that net N mineralization in grassland soils is dynamic, responding within months to fire, grazing, or changes in plant species composition. This conflict illustrates the point that soil organic matter does not behave as a single pool when considering N, C, or other elemental cycles.

Numerous methods have been published for partitioning soil organic matter into chemical, physical, or functional fractions or pools. Many grassland studies follow the Century model (Parton et al., 1987), which partitions soil organic matter into three fractions. The “active” fraction contains low-molecular-weight fractions of recently added plant residues and live microbial biomass. It makes up 2%–8% of total soil organic matter and has a residence time of 1–5 yr. The “slow” pool makes up 40%–60% of soil organic matter and has a residence time of 20–50 yr. The “passive” pool makes up 30%–50% of soil organic matter and has a residence time of over 1000 yr. The slow and passive pools are strongly affected by soil texture and climate. These two pools comprise the vast majority of soil organic matter, yet they contribute less than 30% of the net N mineralization from grassland soils (Schimel et al., 1994). Various methods of soil organic matter fractionation all indicate that a small, highly active soil organic matter fraction (e.g., Century’s “active” fraction) dominates soil biological activity, including N cycling (Collins et al., 1997).

Referring to tropical grasslands and savannas, Huntley and Walker (1982) said, “N has been shown to be of great significance . . . but despite many thousands of N measurements, in all its forms, an understanding of the N cycle still eludes us.” Subsequent N cycling research in grassland/forage systems has emphasized the strong linkages between vegetation and the small active fraction of soil organic matter. In unmanaged humid and subhumid grasslands, this plant–soil interaction reinforces low soil N availability (Wedin, 1995). The low tissue N concentrations of senesced grass leaves and roots lead to microbial N immobilization, reducing net N mineralization, which, in turn, reduces both forage production and forage quality. Low soil moisture in semiarid and arid grasslands constrains both soil microbes and plants, and the role of plant–soil interactions in regulating N cycling is less clear (Burke et al., 1998). To address the natural tendency toward N limitation in grasslands, forage production in humid regions has relied on increasing N inputs to forage systems (N fixation by legumes, animal wastes, inorganic N fertilizer) and managing the plant–soil–grazer (livestock) system to enhance N cycling.

Legumes and N₂ Fixation

Dinitrogen fixation by legumes depends on many factors, including host species and genotype, rhizobial strain and

population size, developmental stage of the host, inorganic N (mainly NO₃⁻) supply, yield of the host, nutrient and toxic element supply, and abiotic growing conditions (Russelle, in press).

There is considerable uncertainty about how much N₂ a particular legume will fix. In general terms, N₂ fixation by forage legumes usually ranges from 50 to 200 kg N ha⁻¹ yr⁻¹ (Fig. 9.1). Estimates of N₂ fixation in white clover–perennial ryegrass mixtures, which make up most of the published measurements on a forage legume, range from 0 to more than 300 kg N ha⁻¹ yr⁻¹ (Russelle, in press). Dinitrogen fixation in pastures tends to be less than in mown forages (Fig. 9.1) because of feedback through excreta.

Constraints to N₂ Fixation

Three conditions are necessary for large amounts of symbiotic N₂ fixation in mixed forage stands (Boller and Nösberger, 1987): (1) high forage yield, (2) high proportion of legume in the mixture (>50%), and (3) high reliance of the legume on N₂ fixation (>70% of plant N). Maintenance of sufficient legume populations has been difficult in many pastures due to selective grazing, inadequate soil fertility, and stand declines caused by pest pressures. Legume production may vary from one year to the next, in part because of oscillations in soil N availability (Loiseau et al., 2001).

Pathways of N Transfer

Oscillations in legume population contribute to transfer of fixed N to nonlegumes. In addition, N is transferred from legumes due to (1) exudation and leakage of N from roots and nodules; (2) senescence and degradation of nodules or roots; (3) direct transfer from legume roots to nonlegume roots through connections made by arbuscular mycorrhizal fungal hyphae; (4) NH₃ loss from legume herbage and reabsorption by grass herbage; (5) movement of N from legume herbage to the soil by leaching or decomposition of surface litter; and (6) redeposition of consumed N by livestock.

Of these, the two most important transfer mechanisms appear to be the decomposition of plant residues, both below and above ground, and the return of N through deposition of livestock excreta. Ledgard (1991), for instance, found N transfer below ground from white clover to perennial ryegrass in a pasture (70 kg N ha⁻¹ yr⁻¹) was similar to that transferred through excreta (60 kg N ha⁻¹ yr⁻¹). Nearly half of the annual N₂ fixed by clover (270 kg N ha⁻¹) was transferred to the grass under these conditions.

What proportion of a mixed stand must be comprised of legumes to provide sufficient N to the nonlegume? In grazed white clover/perennial ryegrass, Sheehy (1989) estimated 41 kg N ha⁻¹ yr⁻¹ was needed to sustain the system, and this may be achieved with clover contents of

about 10% on an area basis. In Brazil, the legume *Calopogonium mucunoides* should make up 13%–23% of the forage dry mass for the sustainability of a mixture with *Brachiaria* (Cadisch et al., 1994). The required proportion of legume in a stand varies with how the forage is used, which depends on livestock species, stocking rate, management, and forage palatability.

Palatable legumes are grazed selectively and need to comprise 20%–30% of the pasture herbage dry matter when pasture utilization (consumption by livestock) is between 10% and 40%. However, with higher utilization rates (40%–70%), legumes must comprise up to 45% of total dry matter (Thomas, 1992). Decreasing the palatability of legumes by planting species or genotypes with higher tannin concentrations, for example, may provide a partial solution to the problem of maintaining legume populations at desirable levels. Factors affecting palatability are discussed in Chapters 45 and 46.

Transfer of Fixed N in Mixtures

It is unclear how much fixed N is transferred from legumes to nonlegumes growing in mixtures because a wide range of estimates has been reported. This is likely due to the large number of interacting conditions that affect N_2 fixation. Transfer of fixed N is positively related to the proportion of legume N derived from the atmosphere; therefore, more N is fixed and transferred under low-N fertility conditions. More N transfer occurs with a higher proportion of legumes in the stand (Brophy et al., 1987). This is due both to greater competition for soil N by the nonlegume and a larger “pool” of fixed N being added to the system.

Transfer of N increases with stand age in perennial forage mixtures, presumably because of increased reliance of the legume on N_2 fixation and the cumulative decomposition of above- and belowground tissue (Jorgensen et al., 1999). Maximum N transfer from alfalfa to meadow bromegrass was $55 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (Walley et al., 1996) and from white clover to perennial ryegrass was $43 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (McNeill and Wood, 1990).

Nitrogen in the Plant-Soil-Grazer System

Cattle, sheep, and other large herbivores affect plant growth rates, plant species abundance, and plant elemental composition by removing herbage, trampling vegetation, compacting soil, and excreting waste. All these effects alter the rates of N transformations, the fate of N, and, ultimately, the N balance of pastures.

Growing ruminants use 5%–10% of the feed N they consume, and lactating dairy cows use 15%–30% for milk production (Haynes and Williams, 1993); the remainder is excreted. Fecal N is mostly insoluble in water and comprises microbial cells (50%–65%), undigested plant residues (15%–25%), and products of livestock metabolism (Haynes and Williams, 1993). Urinary N is

largely soluble and in the form of urea (60%–90%) and other metabolic products such as hippuric acid, creatine/creatinine, and allantoin. Consequently, fecal N contributes mainly to medium- to long-term N-cycling processes, whereas urinary N is subject to rapid cycling or loss.

Nitrogen use efficiency (NUE) by the animal is low, and more N is excreted in urine when the diet is high in degradable protein and low in available energy. Conversely, proper supplementation of pastures with digestible energy improves NUE and reduces N excretion. On the other hand, diet composition causes little change in fecal N output. Urinary N output by sheep was lower on perennial ryegrass/white clover swards (54 g N d^{-1}) than on perennial ryegrass fertilized with $420 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (82 g N d^{-1}), but there was no change in fecal N output (Parsons et al., 1991).

Patchiness of Nitrogen Distribution in Pastures

Concentrated excreta patches generally affect only 14%–22% of the land area of a pasture annually, assuming the patches do not overlap (Haynes and Williams, 1993). Thus, soil sampling must be more intensive than in mechanically harvested forages to produce accurate maps of nutrient distribution (Bramley and White, 1991). A more feasible approach is to determine plant nutrient needs through spectral reflectance of herbage (Taylor et al., 1998). Optimum fertilization with N requires site-specific application, but most farmers in North America have not adopted this practice with forages. More research is needed on this topic because benefits of site-specific N applications in pastures have not been consistent (Taylor et al., 1998; Cuttle et al., 2001).

More excreta are “deposited” in areas where livestock spend time, such as shelter from sun and wind, near field gates, or near watering stations (Bogaert et al., 2000). Moving the water supply or using moveable shade structures improves nutrient distribution in the pasture, as does short-term, high stocking rate grazing systems (Peterson and Gerrish, 1996).

Nitrogen Losses in Pastures

In urine spots, the combination of high soil pH from urea hydrolysis, high NH_4^+ concentration, and high osmotic strength increases NH_3 volatilization and slows nitrification. Gaseous NH_3 losses increase with soil temperature (Lockyer and Whitehead, 1990) and lower soil moisture, making it the primary pathway of N loss in grazed semi-arid grasslands. Under subhumid and humid conditions, between 15% and 25% of urinary N is lost as NH_3 volatilization (Haynes and Williams, 1993). Gaseous N loss by denitrification can be significant in wet soil conditions (28% of urine N in one study; Fraser et al., 1994) but generally accounts for less than 5% of urine N (Luo et al., 1999).

Nitrate leaching loss may be larger when forages are grazed than when mechanically harvested, but this depends on the amount and timing of excess soil water, the general level of N fertility, and crop growth. The amount of available N in a urine spot (equivalent of up to 250 kg N/ha for sheep and 1000 kg N/ha for dairy cattle; Steele, 1987) greatly exceeds the N needs of neighboring plants. High NO_3^- leaching losses occur when precipitation or irrigation occurs during periods of high NO_3^- concentrations (usually 10–30 days after urination; Ball and Ryden, 1984).

Intensive grassland management in humid climates has been implicated in NO_3^- contamination of well water (Burden, 1982). Because N is redeposited by livestock, the probability of NO_3^- leaching losses is higher with higher N fertilizer or manure deposition rates under grazing than under mowing. In New Zealand, for example, critical N application rates were 200–300 kg N ha⁻¹ yr⁻¹ lower for grazed than mown forages to maintain concentrations of leachate NO_3^- below the drinking water standard (Di and Cameron, 2000). Leaching losses are also large in the humid eastern United States on shallow soils, especially with high rates of N fertilization (Stout et al., 2000). In the subhumid region of the Midwest, however, where deeper soils and lower rainfall are typical, NO_3^- leaching losses in forage systems are small, with low to moderate N addition rates (Russelle, 1996).

Excellent management of legume–grass mixtures can yield moderate to high animal production levels with modest N losses (Ledgard et al., 2001). As indicated above, it often is difficult to maintain sufficient legume populations in mixed stands under grazing. The solution to this site-specific problem requires integrated knowledge of plant characteristics, soil conditions, weather, livestock management, pest pressure, and fertilizer and lime management.

Phosphorus Cycling in Forage Systems

After N, P is the nutrient receiving most attention in forage systems. Although plant tissue concentrations of P are much lower than N, P can limit plant productivity under some circumstances. Like N, concern over runoff and leaching of P from agricultural landscapes has also increased dramatically in recent decades. However, the P cycle has important differences from the N cycle that must be considered whether the goal is optimizing P supply for plant and animal production, minimizing P losses to the environment, or, as is increasingly the case, both.

The various transformations that regulate soil N availability (i.e., N mineralization) are almost entirely microbially driven (Chapin et al., 2002). Abiotic soil factors such as low pH affect N availability through their effects on microbes and plants. In contrast, phosphate ions (PO_4^{3-} , the main form of available P in soils) easily form

chemical bonds with various minerals (Smil, 2000). The resulting precipitates are generally unavailable to plants and are known as occluded P. The chemical reactions that PO_4^{3-} undergoes depend on the concentrations of other minerals and pH. At low pH, PO_4^{3-} binds with oxides of Fe, Al, and Mn to form insoluble precipitates. As rock weathers (a process that occurs over millennia), the abundance of Fe, Al, and Mn oxides increases. Thus, highly weathered, ancient soils, such as those found throughout the tropics, have a high potential to chemically immobilize available P (Walker and Syers, 1976). At high pH, PO_4^{3-} binds with Ca to form various calcium phosphates that also precipitate and are relatively unavailable for plant uptake. Thus, P availability is highest at soil pH values around 6.5 and is less available at both higher and lower values.

The rapid geochemical immobilization of PO_4^{3-} in most soils also explains why leaching of PO_4^{3-} into groundwater is rare (Smil, 2000). When P inputs to the soil are high, for instance, with repeated additions of animal wastes to forage systems, the geochemical potential of upper soil horizons to rapidly immobilize or precipitate PO_4^{3-} may be exceeded. Soil solution concentrations of PO_4^{3-} may increase near the surface under these circumstances. In regions of high precipitation, PO_4^{3-} and P associated with dissolved organic matter may leach into lower soil horizons, but P is usually immobilized at that point. This contrasts sharply with NO_3^- , which readily moves with percolating water to great depths and frequently enters groundwater. Like N, high concentrations of soluble and particulate P near the soil surface are vulnerable to loss through runoff and associated soil erosion (Sharpley et al., 1994).

In contrast to N_2 gas for N, there is no atmospheric or gaseous pool of P to replenish terrestrial and aquatic ecosystems. Rather, the ultimate input source for P cycling in natural ecosystems is rock weathering, a process that is very slow compared with N_2 fixation by legumes and other N-fixing organisms (Walker and Syers, 1976). P is abundant in many of the minerals, such as apatite, that form rock, but the solubility of these minerals is low. Because P has no atmospheric pool and the solubility and transport of PO_4^{3-} in soil solutions is low, the linkages between terrestrial P and aquatic P cycles are weak. Simply put, natural terrestrial ecosystems do not leak P to nearby freshwater ecosystems the way they leak N. In addition, in aquatic ecosystems P is limited by the lack of a biotic mechanism for P inputs equivalent to N-fixing cyanobacteria in the plankton. Thus, freshwater ecosystems are often highly responsive and vulnerable to human-caused P loading (Chapin et al., 2002).

Because of concerns over eutrophication of aquatic systems, P management is becoming increasingly important in forage and livestock management (Sharpley et al., 1994; Rotz et al., 2002). With the development of total

maximum daily loads (TMDLs) for surface-water bodies, states have begun to institute limitations on P application to agricultural and residential land. Some states limit or prohibit P addition to fields that have high soil-test P levels, whereas others base application rates on a P risk index that assesses the likelihood of P loss from a field. A P risk index typically includes many factors known to affect runoff, such as slope, soil cover, distance to surface water, etc. Regardless of the approach, producers who manage manure are being affected by public concerns and regulations regarding P runoff.

Public concerns about P runoff are supported by widespread increases in soil-test P levels (Sharpley et al., 1994). In Wisconsin, for example, most soils tested recently contain excessive P. Such buildup can be attributed to repeated applications of livestock and poultry waste, overapplication of fertilizer P, and large amounts of imported P in livestock rations that end up in waste. Because of the relatively high P content of some animal wastes (e.g., poultry litter), soil-test P levels may continue to increase even when manure application rates based on N content are matched to crop N requirements (Rotz et al., 2002). Where soil-test P levels are high, it may take many years to “crop down” fields by harvesting forages. The P removal rate in animal products from pastures is only 10%–35% of that for harvested forages (Gillingham, 1987). Thus, hay sales will send more P off farm than meat or milk. The best long-term solution to P accumulation is to reduce the net import of P to the farm. This generally can be achieved only by reducing input of off-farm P sources (feed, fertilizer, manure, etc.) and increasing export of P in animal and plant products.

Although well-managed perennial pastures provide better soil protection than most annual cropping systems, P losses from damaged vegetation, thatch, and dung can be environmentally important. Loss rates for P of several kilograms per hectare per year have been measured in snowmelt runoff from hay fields and pastures in cold regions. Surface applications of manure, either as nonincorporated broadcast manure from storage or as dung pats from grazing stock, are a rich reservoir of water-soluble or biologically available P. As with N, P distribution on a farm is generally heterogeneous because of long-term management decisions (e.g., fields nearest the manure source receive the most manure) and animal behavior (more dung is deposited in areas where livestock rest than in other areas). Knowledge of soil-test P levels within and among fields and paddocks on the farm is necessary to make appropriate decisions about where manure should be applied by either process.

The Challenge of “Balancing” Nutrient Budgets

Lanyon (1995) published a provocative paper entitled “Does nitrogen cycle?: Changes in the spatial dynamics of nitrogen with industrial nitrogen fixation.” The simple

nutrient cycle diagram found in many ecology or agronomy texts (e.g., N flowing from soil to plant to animal and back to soil within an idealized field) rarely exists in modern agricultural landscapes. Many, if not most, forage systems have relatively small losses of N and P to the atmosphere, groundwater, or surface water when compared with arable land at the field level. Yet, forage systems are an integral component of modern agriculture, which has dramatically changed local, regional, and global nutrient cycles over the last century. Nutrient outputs (forage, grain, livestock, milk) from one field become intentional or unintentional nutrient inputs to landscapes dozens or hundreds of kilometers away.

This spatial uncoupling of nutrient cycles is combined with unprecedented increases in the magnitude of global nutrient cycles. Vitousek et al. (1997) concluded that human activities (primarily industrial N fertilizer production, inadvertent N fixation during fossil fuel combustion, and agricultural management of legumes) have more than doubled the preindustrial global rate at which atmospheric N₂ is transferred to biologically active pools (i.e., fixed). Although the source of P inputs differ (e.g., mining), changes in the global P cycle are of similar magnitude (Smil, 2000).

The potential risk of environmental damage from farming systems may be estimated from nutrient budgets. Assuming conservation of mass, the difference between inputs and outputs indicates the mass of a nutrient that is unaccounted for (Meisinger and Randall, 1991). If one assumes steady-state conditions, mass that is not accounted for is presumed to be a net nutrient loss from the system. The simplest approach at the whole-farm level is to measure the difference between purchased inputs and marketed outputs of a given nutrient and to assume steady-state conditions (e.g., no change in the size of nutrient pools in the soil).

This crude approach, however, is unlikely to be valid for most situations, because management systems (tillage, residue removal, crop rotations, fertilizer management, etc.) vary and interact at time scales shorter than those required for equilibrium of the soil pools. In addition, there can be transfers within the farm, such as those that occur with sediment runoff and deposition that disrupt equilibrium within the farm. The simple balance approach also fails to partition net nutrient losses into specific fluxes, which is critical in determining the broader environmental impacts of local management decisions. For example, while both NH₃ volatilization and N₂O emissions are N losses to the atmosphere, the former has a short residence time in the atmosphere and relatively local negative impacts, whereas the latter is long-lived in the atmosphere and is a potent greenhouse gas (Vitousek et al., 1997).

Given the large spatial and temporal heterogeneity in nutrient fluxes, many have used simulation models to estimate flows. For example, Rotz et al. (2002) projected

that long-term whole-farm P balance could be achieved for dairy farms in the northeastern United States by feeding the minimum dietary P and by maximizing the production and use of forages. Reducing animal N intake or supplementing a grazing herd with metabolizable energy also reduces environmental risk of N loss (Soder and Rotz, 2001). Models have also been used to estimate watershed or regional results (e.g., Cassell et al., 1998), and these can lead to crucial insights. For example, Nord and Lanyon (2003) found that changing the production strategy (e.g., heavy reliance on purchased feeds) on one farm can have larger effects on watershed nutrient balances than changing farm operations (e.g., field-specific manure application rates) on a number of farms.

As more parameters are used in a model (i.e., symbiotic N_2 fixation, net N mineralization, NO_3^- leaching, or gaseous losses), more can be inferred about likely nutrient transfers and other pathways of loss, but the number of estimated and uncertain parameters also increases (Watson and Atkinson, 1999). The nature and magnitude of these uncertainties are important, especially when nutrient budgets are used as policy instruments (Oenema et al., 2003). In western France, for example, NO_3^- leaching from grazed pastures was low for stocking rates less than 550 grazing d $ha^{-1} yr^{-1}$ but increased quickly above that stocking rate (Simon et al., 1997). As farm-scale budgets are aggregated, it is possible to derive general conclusions relevant to watershed and regional spatial scales.

It is difficult to measure nonpoint nutrient losses at large scales, although some pathways are more amenable than others to measurement. Phosphorus loss to a stream (Sharpley et al., 1994), N_2O emission (Weinhold et al., 1995), NH_3 volatilization (Marshall et al., 1998), and NO_3^- loss through tile drains (Watson et al., 2000) have been measured on field scales. Nutrient losses to streams or groundwater are measurable at the watershed scale (Jordan et al., 1997). Many of these approaches, however, are expensive, difficult to replicate, or restricted to a limited suite of sites. Nevertheless, significant advances in remote sensing of land cover and land use, the computational power of geographic information systems, and the instrumentation available for environmental monitoring offer potential. Perhaps most of all, however, the conceptual integration of traditionally separate disciplines, such as soil science, hydrology, agronomy, atmospheric science, and ecology, provide hope that our ability to understand, predict, and manage nutrient cycles will continue to progress rapidly.

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