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Plant–soil–animal interactions and nutrient cycling in grazed grasslands

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1 Introduction

Grasslands are an abundant and important global ecosystem, occupying more than 40% of Earth's ice-free terrestrial surface (Hewins et al., 2018). Grasslands grazed by livestock occur on ~40 million km^2 (Lambin and Meyfroidt, 2011). Interactions among plants, soils, and animals are inherent to grazed grasslands (Sollenberger and Wallau, 2020) and affect a wide range of ecosystem functions. Nutrient cycling is one of the most consequential (Dubeux and Sollenberger, 2020).

In extensively managed grasslands where fertilizer inputs are low, senescent plant litter is an important nutrient pool and plays a significant role in

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nutrient cycling (Dubeux et al., 2007). Forage intake by herbivores, especially at greater stocking rates, reduces the proportion of nutrients cycling in plant litter while increasing that in excreta (Thomas, 1992). Eighty percent or more of most nutrients consumed by herbivores are excreted (Haynes and Williams, 1993), and nutrients in excreta are more labile and cycle at a greater rate than those in plant litter (Bardgett and Wardle, 2003).

Plant recovery of excreted nutrients can be negatively affected by heterogeneous distribution of excreta in the grassland (Dubeux et al., 2009) and by leaching, volatilization, and immobilization (Haynes and Williams, 1993). Although elemental cycles in perennial grasslands are generally more strongly coupled than in annual cropping systems (Soussana and Lemaire, 2014), animal consumption, digestion, and metabolism can result in decoupling of these cycles, leading to potentially negative impacts on air and water quality (Dungait et al., 2012). Decoupling occurs because dung is enriched in C and P and urine in N and K. These nutrients can be lost, recycled, or stored in plants or soils depending on how grazing animals change nutrient distribution in the landscape (Vertès et al., 2019). Decoupling can result in accelerated rates of pasture degradation or limited pasture growth due to nutrient scarcity (Boddey et al., 2004). Thus, understanding nutrient pools and cycles and the effect of management practices on nutrient cycling efficiency is a requisite for improving pasture productivity, reducing off-farm inputs, and minimizing negative consequences to the environment (Dubeux and Sollenberger, 2020).

The focus of this chapter is nutrient cycling in grazed pasture in temperate climates. A limited number of references from subtropical environments are included to provide additional depth and breadth. Within the grazed grassland context, our objectives are to

- describe the most important nutrient pools and the nutrient fluxes among pools;
- discuss the roles of management practices, soil fauna, and the soil microbial community in nutrient cycling;
- provide example nutrient budgets for grazed swards;
- present a case study which quantifies nutrient pools and fluxes in an N-fertilized grass monoculture and a grass-legume mixed pasture; and
- identify current knowledge gaps and future research needs.

These are discussed in more detail in this chapter.

2 Nutrient pools and fluxes in grazed pastures

Others have described in detail the nutrient pools and fluxes in grazed pasture (Dubeux et al., 2007). Here, we briefly overview N, P, and K pools, providing greater emphasis on nutrient fluxes among pools.

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The largest N pool is the atmosphere, but because of the high stability of N_2, it is only available to plants through biological or industrial N fixation. The atmospheric N pool is ~16 000 times greater than the sum of the soil and biotic N pools (Russelle, 1996). Thus, N is the most limiting nutrient in many grasslands (Russelle, 1992). Second in size to the atmospheric N pool is soil, with the size of this pool being site-specific as it is determined by the quantity of organic matter, microbial biomass, adsorbed NH_4^+, and plant-available inorganic N (Stevenson and Cole, 1999). Soil mesofauna (invertebrates, 0.1–2.0 mm in size, e.g. nematodes and mites) and macrofauna (e.g. earthworms, snails, beetle larvae) are important components of the soil N pool and represent far greater amounts of N than the sum of aboveground and belowground plant N (Dubeux et al., 2007). Plant litter is a relatively small N pool, but it is important because litter and soil microbiota constitute the link between N in living plant tissue and that available for subsequent plant uptake (Thomas and Asakawa, 1993).

The soil is the greatest reservoir of K. Because K is not found primarily as a component of organic compounds, its chemistry in soils is based on cation exchange reactions. In sandy soil environments, the absence of organic matter or clay decreases effective cation exchange capacity and makes K vulnerable to leaching (Dubeux et al., 2007). Animals represent a relatively small K pool, but they play an oversized role in K cycling because of the relatively high K concentration in ingested herbage and the large proportion of ingested K that is excreted.

Compared with N and K, both plant and animal tissue P concentrations are relatively small, resulting in soil being the largest and most important P pool in grasslands (Haynes and Williams, 1993). The availability of soil P depends not only on turnover of organic P but also on the chemistry of inorganic P (Rao et al., 1999). The potential retention of inorganic P in unavailable oxide forms, a function of soil pH and chemical composition, is a constraint to efficient P cycling.

The proportion of K, P, and N in live herbage, aboveground plant litter (belowground plant litter was not measured), live root-rhizome biomass, and soil (to a 20-cm depth) nutrient pools is shown for rotationally stocked bermudagrass (*Cynodon* spp.) pastures (Table 1; Liu et al., 2011c; 2017). Expressed as a proportion of the sum of nutrients in the four pools, the soil N and P pools contained >80% of these nutrients, regardless of grazing or N fertilizer management, while for K, the proportions in soil and live herbage were nearly the same and always in a range of 31–49% (Table 1). Across all nutrients, the plant litter pool was the smallest. Increasing N fertilization increased the proportion of K and N in plant pools and decreased the proportion in soil (Liu et al., 2017). As grazing stubble height increased, the size of the plant nutrient pools increased, mainly as a function of greater herbage mass (Liu et al., 2011c).

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Table 1 Ranges in potassium, phosphorus, and nitrogen in each of four nutrient pools as a proportion of the total amount of that nutrient present across all measured pools of rotationally stocked Tifton 85 bermudagrass pastures

	N fertilization	Polynomial contrast: N[a]	Stubble height	Polynomial contrast: height[b]
Nutrient/Pool	% in pool		% in pool	
Potassium				
Live herbage	31–42	L*↑[c]	35–42	L**↑[c]
Plant litter	2–4	L*↑	3–4	L*↑
Root-rhizome	18–22	Q*	16–20	NS
Soil to 20 cm	36–49	L**↓	36–46	L**↓
Phosphorus				
Live herbage	7–10	NS	6–8	L↑
Plant litter	2	NS	2	NS
Root-rhizome	6–9	NS	6–7	NS
Soil to 20 cm	79–85	NS	83–86	L↓
Nitrogen				
Live herbage	2–4	L**↑	3–4	Q**
Plant litter	1–2	L**↑	1–2	L**↓,Q**
Root-rhizome	3–4	L*↑,Q*	4	NS
Soil to 20 cm	90–93	L**↓,Q	90–92	Q**

* and ** indicate the significance at the $P \leq 0.05$ and 0.01 levels, respectively. L or Q not followed by an asterisk, $P \leq 0.10$; NS = not significant, $P > 0.10$. ↑ indicates the linear response was increasing as the N level or stubble height level increased. ↓ indicates the linear response was decreasing as the N level or stubble height level decreased.
[a]Polynomial contrast for the effect of N level (50, 150, and 250 kg ha^{-1} $year^{-1}$) within nutrient and pool.
[b]Polynomial contrast for the effect of post-grazing stubble height (8, 16, and 24 cm) within nutrient and pool.
[c]L = linear and Q = quadratic effects of N level or stubble height.
Ranges for each nutrient in each pool bound the responses observed across different levels of pasture N fertilizer rate and post-grazing stubble height. Source: Data adapted from Liu et al. (2011c, 2017).

Nutrient pool size is a snapshot in time as nutrients cycle among pools and undergo biochemical processes that alter their chemical structure and biological availability (Dubeux and Sollenberger, 2020). Grazing animals affect grassland nutrient cycling by modifying spatial distribution of nutrients (Mathews et al., 1994a) and altering nutrient lability (Vertès et al., 2019). Within the soil–plant–animal continuum, nutrient cycling is a critical supporting ecosystem service because grassland productivity is often nutrient limited (Sollenberger et al., 2019).

Although soil is generally the largest nutrient pool in grazed pastures, mineralization rates of soil nutrients (20–50 g kg^{-1} of that nutrient in soil $year^{-1}$) are much lower than for nutrients in livestock excreta and plant residues

(650–850 g kg^{-1} $year^{-1}$; Brady and Weil, 2002), limiting the quantity made available from the soil for plant growth (Dubeux et al., 2007). As a result, plant litter and animal excreta play an important role in nutrient cycling in grazed grasslands (Boddey et al., 2004).

Aboveground deposition and decomposition of plant litter occur continuously in grasslands and influence plant nutrient supply. Litter chemical composition affects the balance between nutrient mineralization and immobilization processes that are especially important for N, P, and S (Myers et al., 1994). Increasing N fertilization and stocking rate increase litter quality (i.e. greater N concentration, lesser C:N ratio) and rate of litter turnover and nutrient release (Dubeux et al., 2006a). Species composition of grasslands also affects litter dynamics, with the presence of legumes increasing litter quality, decomposition rate, and net nutrient mineralization compared with grasslands devoid of legumes (Kohmann et al., 2018).

Most mineral nutrients consumed by herbivores are excreted in dung and urine (Rotz et al., 2005). Nutrients in urine are either in inorganic forms and immediately plant-available or are mineralized in a few days and rapidly available for plant uptake (Mathews et al., 1996). Nutrients in dung vary in form and rate of availability, and the release of nutrients from dung is generally slower than from urine but more rapid than from plant litter (Bardgett and Wardle, 2003). Because the quantity of nutrients contained in excreta patches greatly exceeds the immediate plant requirement, particularly in areas of the animal congregation, significant nutrient losses to the environment can occur (Cai et al., 2017; Mitchel et al., 2021).

3 Excreta deposition and nutrient return in grazed pastures

This theme explores deposition patterns, excreta effects on plant responses and soil characteristics, nutrient losses from excreta, and the use of nitrification inhibitors to minimize loss of excreta N. In terms of excreta nutrient composition, we can generalize that P, Ca, Mg, and several micronutrients (e.g. Fe, Cu, Mn, and Zn) are excreted primarily in the dung, while K and to a lesser extent Na are excreted primarily in the urine. The proportion of N and S excreted in dung versus urine is a function of diet composition (Mathews et al., 1996), with their proportion in urine increasing as dietary N and S concentrations increase (Rotz et al., 2005). Braz et al. (2002) reported that 93% of the N ingested by animals grazing a low-protein grass pasture was excreted in dung, while 80% of N was excreted in urine when forage N concentration was high (Whitehead, 2000).

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3.1 Patterns of excreta deposition

Forage consumption by grazing herbivores and subsequent excreta deposition redistributes and concentrates nutrients in the landscape, with effects on sward composition and productivity and on the environment (Auerswald et al., 2010). Excreta patch distribution is generally not homogeneous, and there are positive correlations between the time spent in a location by herbivores and excreta patch density (Draganova et al., 2016). Understanding patterns of excreta deposition and what affects them is important for enhancing our understanding of nutrient cycling within a pasture.

Grassland topography affects excreta deposition patterns. On hilly topography of southern Bavaria, Germany, aggregation of excreta occurred on the lowlands and crests due to increased time spent grazing and resting, respectively (Auerswald et al., 2010). In New Zealand hill country, annual excreta N load on low slopes was two- and four-fold that of medium and high slopes due to livestock behavior, specifically the choice of low slopes as primary areas for congregating (Hoogendorn et al., 2016). In a sloping pasture in southern Germany, nutrients accumulated in a flat crest zone and were depleted in the steeper areas (Schnyder et al., 2010). These studies indicate that slope and/or landscape position affect where cattle congregate and the quantity of excreta loading, with implications for soil nutrient concentrations, nutrient cycling, and potential nutrient loss.

Structural features of pastures affect patterns of excreta deposition. Herd concentration areas such as laneways, water troughs, and shade have greater nutrient loading from excreta than areas where such structural features are not present (De Rosa et al., 2020; Mitchell et al., 2021). Excreta deposition in locations where animal treading occurs accentuates nutrient losses due to compaction, and these areas increase spatial concentration of soil nitrate (NO_3^-) and nitrous oxide (N_2O) fluxes (Mitchell et al., 2021). In Japan, the spatial distribution of cow dung was well-predicted considering green herbage mass and distance from a water trough (Yoshitoshi et al., 2020). In warm climates or seasons characterized by warm temperatures, well-distributed shade caused dung distribution to be more homogeneous; the absence of trees resulted in the concentration of excreta around gates, water troughs, and mineral feeders (Carpinelli et al., 2020).

Shade affected excreta deposition patterns under high-temperature conditions in Hawaii, USA (Mathews et al., 1999). Authors reported that N, P, and K accumulation was greater around shade than waterers, but in paddocks without shade, substantial amounts of P and K accumulated near the water points. In a high-temperature environment in Florida, USA, soil N, P, and K accumulated in zones closest to shade and water, suggesting that positioning of structural features plays a major role in excreta deposition and soil

nutrient redistribution in warm environments (Mathews et al., 1994a; 1994b). Likewise, soil P, K, and Mg concentrations in tall fescue [*Lolium arundinaceum* (Schreb.) S.J. Darbyshire] pastures were ~2–8, 3–15, and 1–1.5 times greater, respectively, near shade and water than at greater distances (Schomberg et al., 2000).

Fence line position may affect the pattern of excreta deposition. In an isolated pasture, excreta density decreased with proximity to a fence line (Auerswald et al., 2010), but deposition along fence lines increased if animals occupied adjoining pastures (Dubeux et al., 2014). In strip-grazed dairy paddocks in Australia, soil K was greater near the gate end than at the back of the paddock (Aarons et al., 2015). These studies support that structural features and arrangement of pastures are important determinants of excreta deposition patterns with consequent effects on soil nutrient concentrations and that shade has an especially important role in deposition patterns in warm environments or seasons.

3.2 Nutrient return from excreta and plant response

Nutrient input to pastures from excreta affects forage production and composition. Nutrients in urine are immediately available for plant uptake, but rate of nutrient availability from dung varies (Mathews et al., 1996). Most K in dung is water soluble and readily available, while N, Ca, Mg, and S release is slower and negatively affected by lignin and polyphenols that reduce mineralization rates (Mathews et al., 1996). Nutrient returns (kg ha^{-1} $year^{-1}$) in dung for year-round grazing systems ranged from 5 kg to 9 kg P, 10 kg to 26 kg K, 10 kg to 16 kg Ca, 9 kg to 13 kg Mg, and 23 kg to 40 kg N (Garcia et al., 2021). Total annual N excretion in dung plus urine ranged from 69 kg ha^{-1} to 89 kg ha^{-1} (Garcia et al., 2021), demonstrating the importance of nutrient return in excreta to subsequent plant response.

Urine deposition increased herbage accumulation by 15% and herbage N and K concentrations by 19% and 17%, respectively, over a no-excreta control in perennial grasslands in Germany (Scheile et al., 2018). The effect on herbage accumulation was measurable 7 months after urine deposition. Forage accumulation in perennial ryegrass (*Lolium perenne* L.)-white clover (*Trifolium repens* L.) swards in New Zealand increased linearly with increasing urine-N application rates (Di and Cameron, 2007). Forage accumulation in urine patches increased from 3040 kg ha^{-1} to 4820 kg ha^{-1} as deposition frequency increased in bahiagrass *(Paspalum notatum* Flugge) pastures in Florida, USA (White-Leech et al., 2013b). Forage accumulation and nutritive value increased to distances of 15 cm and 30 cm beyond urine deposits, respectively, with increases in forage accumulation observed for ≥84 days following single or multiple urine applications (White-Leech et al., 2013a).

Forage responses to dung deposition have been less consistent. In the northern Czech Republic, herbage N, P, and K concentrations were greater at dung patches in intensively grazed swards (Pavlu et al., 2019), but herbage accumulation at dung patches was not different than the control in Solling Uplands, Germany (Scheile et al., 2018). Bahiagrass forage accumulation decreased linearly from 3040 kg ha^{-1} to 2800 kg ha^{-1} with increasing dung application frequency, apparently due to grass smothering associated with crusting of the exposed surface of dung pads and minimal dung beetle activity (White-Leech et al., 2013b). In contrast, perennial ryegrass herbage accumulation increased for 112 days after the application of dung from grazing dairy cattle in temperate Australia (Aarons et al., 2009). Overall, the literature suggests that urine deposition will almost always be accompanied by increasing forage accumulation and nutritive value, but plant responses to dung are less predictable and depend significantly on the presence of soil fauna like dung beetles and on weather conditions, especially rainfall, following deposition.

3.3 Excreta effects on soil characteristics

Excreta deposition generally changes soil nutrient status sufficiently to affect fertilizer recommendations (Carran and Theobold, 2000). Grazed New Zealand pastures with natural excreta deposition for 23 years were compared with those protected from excreta deposition and grazing (Carran and Theobold, 2000). Where excreta was deposited, total soil N was 20% greater in the top 15-cm soil layer and more N was mineralized in the 7.5–15 cm layer than in excluded areas. Exchangeable K was much greater in areas accessed by cattle than where they were excluded.

In the northern Czech Republic, the presence of dung in grazed, mixed-species perennial pastures did not affect soil nutrient concentrations, likely because nutrients not utilized by plants were leached, volatilized, or transformed into unavailable forms (Pavlu et al., 2019). Conversely, in temperate dairy pastures in Australia, extractable P and K increased in the 0–5 cm soil layer under dung, total organic P was unaffected, and microbial soil P increased (Aarons et al., 2004a; 2004b; Aarons et al., 2009). In Switzerland, cattle urine application stimulated nitrification and denitrification enzyme activities in soil and increased NH_4^+ and NO_3^- concentrations in species-rich temperate pastures (Hartmann et al., 2013).

Pasture species composition can affect nutrient cycling via animal excreta. Dung from cattle grazing a smooth bromegrass (*Bromus inermis* Leyss.)-legume mixture had 5–12% greater N concentration than that of cattle grazing a bromegrass monoculture receiving 90 kg N ha^{-1} $year^{-1}$ (Schick et al., 2019). Seven days after dung deposition, the soil below dung patches in the mixed-species pasture had 20–42% greater inorganic N than the soil under dung

patches in the N-fertilized grass monoculture. Greater diet nutritive value in the grass-legume mixture increased dung N concentration, resulting in faster dung decomposition and more rapid nutrient movement into the soil.

3.4 Nutrient losses from excreta

Nutrients exit grazed grasslands in animal products or through losses including volatilization, leaching, and runoff (Dubeux and Sollenberger, 2020). Increasing intensity of grassland management leads to greater amounts of reactive N and P in the soil, increasing the likelihood of losses to the environment (Soussana and Lemaire, 2014). Because of its importance to productivity and the ease at which it can be lost from the system, N is a primary nutrient of concern. Pathways of N loss vary with climate, with gaseous losses predominating in arid environments and NO_3^- leaching predominating in high-rainfall environments (Russelle, 1992).

Deposition of either dung or urine increases N_2O emissions by enhancing N mineralization, nitrification, denitrification, or nitrifier denitrification. Nitrification is the microbe-mediated aerobic process resulting in oxidation of NH_4^+ to NO_2^- and then to NO_3^-, with the release of N_2O as a byproduct (Cai et al., 2017). Increased NH_4^+ availability after excreta deposition enhances soil nitrification by stimulating its first step (Hartmann et al., 2013). Denitrification is the reduction of NO_3^- to NO_2^-, then to NO, N_2O and N_2, in association with the increased presence of ammonia-oxidizing bacteria and denitrifying bacteria (Cai et al., 2017). Nitrifier denitrification is a process mediated by ammonia-oxidizing bacteria, where NO_2^- is reduced to N_2, but N_2O is produced as an intermediate or final product (Wrage et al., 2004; Wrage-Mönnig et al., 2018). Denitrification activity is expected to increase after dung deposition due to increasing C and decreasing O_2, and after urine deposition because of increasing soil moisture, NO_3^-, and labile C (Wrage-Mönnig et al., 2018).

Temporal and spatial fluctuations in N_2O emissions occur. Increased fluxes immediately after grazing may be due to increased N deposition from animal excreta, soil compaction, and reduced plant N uptake (De Rosa et al., 2020). Spatial differences in N_2O fluxes and soil NO_3^- concentrations occur because of greater animal traffic in some areas of the pasture and resultant greater excreta N deposition. As an example of the latter, N_2O fluxes in the proximity of pasture gates were 11 times greater than the field average (De Rosa et al., 2020). Similarly, herd congregation areas had 3–14 times greater N_2O emissions compared with remaining pasture areas (Mitchell et al., 2021). Authors concluded that 3% of farm area was responsible for 28% of soil N_2O emissions (Mitchell et al., 2021). Thus, spatial management of N loading may reduce N_2O emissions and N losses from grazing systems.

Urine patches are hotspots of N_2O emissions from grazed pasture. There is variation in emissions among patches, and in Sweden most was accounted for

by cumulative precipitation 20 days after application (R^2 = 0.60) and average soil water-filled pore space 30 days after application (R^2 = 0.45) (Barczyk et al., 2023). In smooth bromegrass pastures in Nebraska, USA, patches that received a single urination event lost on average six times more N as N_2O than areas that received no urine (Snell et al., 2014). The N_2O emissions were positively associated with measurements of soil water-filled pore space and were much greater in a year with average precipitation versus drought.

Nitrate–N leaching losses from urine are generally a function of the amount of N deposited and live plant cover. On perennial ryegrass-white clover pastures in New Zealand, NO_3^-–N leaching losses increased with increasing amounts of urine N (Di and Cameron, 2007). Similar results were found on orchardgrass (*Dactylis glomerata* L.) swards in Pennsylvania, USA (Stout, 2003). Rates of 0 kg, 300 kg, and 700 kg of urine N ha^{-1} were applied to lysimeters with perennial ryegrass present or absent (Rayner et al., 2020). Presence of perennial ryegrass reduced leachate from 28 kg N ha^{-1} (plants absent) to 2 kg N ha^{-1} for the no-urine control and from 375 kg N ha^{-1} to 93 kg N ha^{-1} for the 700 kg N ha^{-1} treatment.

Nitrogen in dung is primarily in an organic form requiring mineralization for release, with less leaching occurring than from urine (Wachendorf et al., 2008). Conversely, K in both dung and urine is primarily in a soluble form. While leaching of K from grasslands is usually low (Kayser and Isselstein, 2005), it can occur in sandy soil environments if high levels of available soil K are present, for example, at urine patches (Dubeux et al., 2007). In New Zealand, grazing dairy cows were estimated to be directly or indirectly responsible for 74–92% of all K losses in pastures over a 30-year period (Williams et al., 1990).

4 Mitigating nitrous oxide emissions: nitrification inhibitors

Because livestock urine patches are an important source of N_2O emissions in grazed grasslands, nitrification inhibitors have been investigated as a N_2O mitigation strategy (Adhikari et al., 2021). Mitigation of N_2O emissions from urine patches must target the production of NO and NO_2^- during nitrification by inhibiting NH_3 oxidation (Clough et al., 2020). Research into nitrification inhibitors has evaluated chemical additives and plant species with observed biological nitrification inhibition. The effects of nitrification inhibitors and selected management interventions on N_2O emissions and other categories of N losses are described in the following text and summarized in Table 2.

In a review of 196 datasets, the chemical nitrification inhibitors dicyandiamide, 3,4-dimethylpyrazole phosphate, and nitrapyrin reduced N_2O emissions from urine patches by 44%, 28%, and 28%, respectively (Adhikari et al. 2021). A meta-analysis found the three inhibitors reduced N_2O emission

Table 2 Selected management interventions and their effects on N losses in grazed pasture

Management intervention	Treatments	Responses	References
Biological nitrification inhibitors (BNIs)	Urine applied to pasture grass with either high or low BNI potential	High BNI pasture grass ↓ N_2O emissions from urine spots 60% compared with low BNI grass	Byrnes et al., 2017
BNI	Urine applied from cattle with diet containing fodder beet or with no fodder beet	Beet diet urine ↓ ammonia-oxidizing bacteria population 36%, soil NO_3^-–N concentration 31%, and NO_3^-–N leaching losses from urine 64% versus pasture diet	Talbot et al., 2020
BNI	Standard urine applied to pastures varying in % plantain in herbage mass	Total urine N_2O emissions ↓ linearly as proportion of plantain in the sward increased ($r^2 > 0.96$)	Simon et al., 2019
Chemical nitrification inhibitor (CNI)	0 kg, 300 kg, 700 kg, and 1000 kg urine N ha^{-1} with or without dicyandiamide	CNI ↓ NO_3^-–N leaching from urine by 63%; ↑ N offtake in forage by 32%	Di & Cameron, 2007
CNI	Various meta-analysis	CNI ↓ N_2O emissions from excreta 28–44%;↑ forage accumulation (15%) and N uptake (13%)	Adhikari et al., 2021
CNI	Incubated urea + manure with or without nitrapyrin	CNI ↓ N_2O emissions from excreta by 64–71%	Tao et al., 2021
CNI	Meta-analysis (N sources: dung, urine, slurry, fertilizer)	CNI ↓ N_2O emissions from various excreta sources by 52–66%	Soares et al., 2023
Inoculation with arbuscular mycorrhizal fungi (AMF)	Grass microcosms fertilized with NO_3^- or NH_4^+ and inoculated with AMF or a non-mycorrhizal control	Presence of AMF in pasture soils ↓ N_2O fluxes by 62% and ↓ NH_4^+ leaching by 75% (pasture wide, not only excreta); plant N content ↑ 13% due to AMF	Bender et al., 2015
Legumes versus N fertilizer	Legume–grass mixture with 44% legume and no N fertilizer versus mixture with 4% legume + fertilizer	Substituting legume for N fertilizer ↓ N_2O emissions 39–54% (pasture wide, not only from excreta)	Fuchs et al., 2018
Legumes versus N fertilizer	Legume–grass mixture (~30% legume) versus grass with N (224 kg N ha^{-1} $year^{-1}$)	Legume integration ↓ NO_3^- leaching (pasture wide, not only from excreta) by 51% versus N-fertilized grass monoculture	Garcia et al., 2021; Trumpp, 2024
Minimize excreta deposition in livestock traffic areas	Excreta deposition in herd concentration areas (HCA), pasture, and riparian zones	Excreta deposits in HCA ↑ N_2O emissions 9× compared with those on pasture; 3% of farm area accounts for 28% of greenhouse gas budget	Mitchell et al., 2021
Plant cover	Presence/absence of perennial ryegrass and application of 0 kg, 300 kg, or 700 kg N ha^{-1} as bovine urine	Presence of perennial ryegrass ↓ NO_3^-N leaching losses from urine by 75%	Rayner et al., 2020

factors for urine, dung, and slurry by 57% (Soares et al., 2023). Mechanistically, dicyandiamide reduced the ammonia-oxidizing bacteria population in soil at cattle urine patches, but there was no effect on the denitrifying bacteria population (Cai et al., 2017). In a calcareous soil in China, nitrapyrin reduced total N_2O emissions from a urea plus cattle manure mixture by 65–71% compared with the no-inhibitor control, and the reduction was associated with lesser ammonia-oxidizing bacteria gene abundance (Tao et al., 2021). Application of dicyandiamide to a perennial ryegrass-white clover mixture growing in lysimeters in New Zealand reduced NO_3^-–N leaching losses by 63% and increased N harvested in forage by 32% (Di and Cameron, 2007). Across multiple studies, the application of dicyandiamide increased herbage accumulation and N uptake by 13% and 15%, respectively (Adhikari et al., 2021).

Small amounts of dicyandiamide residues in milk products have been reported from its use on grazed pasture, and 3,4-dimethylpyrazole phosphate and nitrapyrin can enter the food chain via grazing livestock (Adhikari et al., 2021). Despite these observations, no negative impacts on non-targeted organisms have been documented from the use of chemical nitrification inhibitors at recommended rates (Adhikari et al., 2021). Continued research is needed to confirm these findings.

Biological nitrification inhibitors (BNIs) may be released as root exudates of plants and can decrease nitrifier activity in agricultural soils, improving N recovery from fertilizer and excreta and enhancing N-use efficiency (Zhang et al., 2022). Two *Urochloa* spp. grasses were compared, one which had reported BNI activity (cv. Tully) and one that did not (cv. Mulato) (Byrnes et al., 2017). Following urine application, soil under Tully showed suppression of nitrification and denitrification and reduced presence of ammonia-oxidizing archaea relative to soils under Mulato. Cumulative N_2O fluxes from urine application were greater on pastures under Mulato than for Tully. Inhibitor effects also occurred following the application of urine from animals fed fodder beet (*Beta vulgaris* L.) (Talbot et al., 2020) and when urine from cattle consuming non-fodder beet diets was applied to fields where fodder beet was growing (Yao et al., 2018). Secondary plant metabolites in urine following beet consumption or from root exudates of live beets may inhibit bacteria growth and nitrification, leading to lower N_2O emissions. Biological nitrification inhibition remains under-researched as a mitigation option in pastures (Clough et al., 2020). Readers are referred to Zhang et al. (2022) for a consideration of mechanisms of BNI.

Diet species composition can affect urine volume and urine N concentration. Chicory (*Cichorus intybus* L.) and plantain (*Plantago lanceolata* L.) have high water and mineral concentrations and may modulate urination behavior and urinary N excretion (Mangwe et al., 2019). Even with similar N intake, increased

urination events for animals fed chicory diluted urine N concentration compared with animals consuming white clover, decreasing N loading at individual urine patches. Similarly, urine N-loading rates of animals fed plantain decreased with increasing proportions of plantain in the diet, lowering N_2O emissions (Simon et al., 2019).

5 Litter deposition and decomposition in grazed pastures

In grazed perennial grasslands, deposition and decomposition of aboveground plant litter continuously influence plant nutrient supply compared with periodic pulses of litter deposition in annual cropping systems (Eilittä et al., 2003). Litter composition affects the net balance between nutrient mineralization and immobilization, influencing nutrient availability of N, P, and S (Myers et al., 1994).

5.1 Litter deposition

Pasture herbage mass, grazing management, N fertilization, and species composition affect litter mass and deposition rate. Smooth bromegrass litter deposition was 48% greater in N-fertilized versus unfertilized rotationally stocked pastures, and the greatest litter mass and litter deposition rate coincided with periods of peak herbage mass (Guretzky et al., 2014). Bermudagrass litter mass and litter deposition rate increased as target grazing height and residual herbage mass increased (Liu et al., 2011a).

Greater N fertilizer rates combined with greater stocking rates of bahiagrass pastures caused a larger nutrient contribution from litter than in extensively managed swards (Dubeux et al., 2006b). Comparing grass–legume mixtures with N-fertilized grass monocultures, litter mass was greater for heavily fertilized grasses than for the mixture (Silva et al., 2021). In contrast, grass and grass–legume litter deposition was similar when the grass received a relatively low N rate (Kohmann et al., 2018). Overall, management practices that increase pasture herbage mass are likely to increase litter deposition rates.

5.2 Litter decomposition

Decomposition of plant litter depends in part on initial N, soluble sugar, and recalcitrant compound concentrations and whether it has already partially senesced (Rumpel et al., 2015). When added to soil, grass litter of different stages of maturity decomposes at different rates (Sanaullah et al., 2010). Litter decomposition can also vary significantly for grasses receiving different N rates, managed under different grazing treatments, or associated with various proportions of legumes.

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Greater N fertilization and stocking rate of bahiagrass pastures increased litter turnover and nutrient release compared with lesser management intensity (Dubeux et al., 2006a). Litter composition for the high and low intensities, respectively, were 22.9 versus 14.1 g N kg^{-1}, 1.3 versus 0.8 g P kg^{-1}, a C:N ratio of 22 versus 40, and a lignin:N ratio of 4.4 versus 5.8 (Dubeux et al., 2006b). In bermudagrass pastures, grazing intensity did not affect litter decomposition, but increasing N fertilization increased the decomposition rate (Liu et al., 2011b). Smooth bromegrass litter N concentration was greater when N fertilizer was applied compared with a no-N control (18.9 g N kg^{-1} and 16.5 g N kg^{-1}, respectively), and N release through decomposition was 46 g N kg^{-1} and 27 kg N ha^{-1} for the same treatments, respectively (Guretsky et al., 2014).

A comparison of grass–legume mixtures versus a grass monoculture receiving 112 kg N ha^{-1} showed similar amounts of N released through plant litter decomposition (Jaramillo et al., 2021a). When litter N returns were compared for legume–grass mixtures and grass monocultures that were unfertilized or fertilized at 50–60 kg N ha^{-1}, N return from mixture litter was greater than that from litter of unfertilized grass pastures and as great or greater than litter from the N-fertilized pastures (Kohmann et al., 2018; 2019). These data suggest the amount of N fertilizer applied to a grass monoculture affects comparisons of litter dynamics with grass–legume mixtures. This was supported further by a comparison of year-round forage systems, where those including legumes contributed important quantities of N via aboveground litter decomposition, but amounts were less than for grass systems receiving large N fertilizer inputs (290 kg N ha^{-1}; Silva et al., 2021).

6 Efficiency of nutrient cycling in grazed pastures

Grassland species composition and grazing management are important factors driving nutrient cycling efficiency. Species diversity or introduction of legumes influence nutrient budgets and soil nutrient availability through plant physiological attributes as well as intra- or interspecific interactions (Rumpel et al., 2015). Different rooting depths and root architecture among species in a diverse sward may reduce N losses. The presence of legumes, due to biological N fixation (BNF), can influence nutrient cycling. This N can be made available to associated non-fixing species through excreta of grazing animals, decomposition of legume litter, sloughing of legume roots and nodules in response to defoliation, and rhizodeposition pathways including decomposition and decay of nodules and root cells and exudation of soluble N compounds by legume roots (Fustec et al., 2010). Associating legumes with non-fixing species in pastures leads to greater efficiencies in N use than legumes grown as a component of sequential crop rotations. This is due to the niche separation effect

which stimulates the BNF of legumes and increases the herbage accumulation rate of the non-fixing species (Corre-Hellou et al., 2006; Fustec et al., 2010).

Replacing fertilizer N with BNF from legumes was found to reduce N_2O emissions by 39–54% (Fuchs et al., 2018; Table 2), with N_2O emissions from clover residue decomposition and rhizodeposition less than N_2O emissions from N fertilization. A summary of whole-system and life-cycle assessment analyses for dairy farms indicated that at similar total N inputs, clover-grass pasture systems can be more efficient than N-fertilized grass systems per kilogram of milk produced from both energy-use and greenhouse gas emissions perspectives (Ledgard et al., 2009).

Livestock grazing modifies ecosystem properties and can elicit cascading effects, with ultimate impacts on soil nutrient cycling and primary production (Sun et al., 2017). The selection of an optimum stocking rate is important because it affects soil function significantly, even more so than stocking method (Read et al., 2016). Pasture decline is often caused in part by a lack of maintenance fertilization, but it can be hastened by greater than optimal stocking rates. Overgrazing results in a large proportion of nutrients cycling in dung and urine which are disproportionately deposited in high-traffic areas where plant cover and ability to take up nutrients are minimal (Boddey et al., 2004). In the Hongyuan Alpine Meadow of China, heavy grazing also decreased root exudation rate and soil N mineralization rate, while moderate grazing effects on these parameters were not different from ungrazed areas (Sun et al., 2017). During the recovery period after grazing, moderately grazed swards had greater root exudation rate, soil inorganic N concentration, and net soil N mineralization rate than heavily grazed pastures. On the Tibetan Plateau in China, heavy grazing reduced gross N mineralization rates by 19%, suppressing microbial N mining and overall soil N availability (Sun et al., 2018). Similar to the recommendations of Boddey et al. (2004), they suggested that overgrazing is likely to amplify plant N limitation, hampering productivity and pasture recovery, but a reduction in grazing pressure can be a sustainable way to maintain soil fertility, C sequestration, efficient nutrient recycling, and overall ecosystem stability.

Enhancing uniformity of excreta deposition is an important goal of grazing management with potential benefits to nutrient-use efficiency, but achieving this goal is challenging (Dubeux et al., 2009). After 3 years of cattle grazing bahiagrass at three intensities, soil nutrient concentrations were greatest close to shade and water regardless of grazing intensity (Dubeux et al., 2009). In a study comparing rotational and continuous stocking of bahiagrass during summer, soil nutrient concentration was generally not affected by stocking method, but soil nutrients accumulated in pasture zones closest to shade and water (Dubeux et al., 2014). Similarly, in a 2-year study with bermudagrass in a warm climate, stocking method had less effect on soil nutrient distribution than

position of shade and water points (Mathews et al., 1994a). Of these, shade was a greater attractant than water points (Mathews et al., 1999). Thus, the position of shade and water may have a greater effect on the efficiency of nutrient cycling than stocking method in warm environments.

Grazing exclusion or duration of grazing during a season affect nutrient pools and cycling. In a subalpine pasture, longer periods of grazing during the warm season increased soil total and extractable N, almost doubled microbial biomass C, increased extracellular enzyme activity, and enhanced nutrient cycling mobilization (Gavrichkova et al., 2022). Grazing exclusion in the Tibetan Plateau enhanced net CO_2 uptake by 17% during the growing season, but it aggravated mineral nutrient limitations to plant growth by decreasing soil NO_3^-- N concentration by 50% and foliar concentrations of N by 10%, Ca by 9%, S by 10%, Fe by 39%, and Mn by 20% (Liu et al., 2020). Likewise in karst alpine grasslands in China, rotational or continuous stocking reduced soil organic C after 17 years compared with grazing exclusion, but grazing promoted soil total N (Yang et al., 2023). While exclusion provided improvements in plant recovery and ecosystem C stock, grazing promoted greater ecosystem nutrient stocks.

7 Role of soil fauna and microbial communities in pasture-nutrient cycling

Invertebrates exert a major influence on soil N availability, particularly in soils of low N fertility (Schon et al., 2011). They increase plant uptake of N from surface-applied plant litter, and in compacted soils, tunneling organisms such as earthworms improve soil structure and N availability (Schon et al., 2011).

Much of the available research assessing the role of invertebrates in pasture-nutrient cycling focuses on dung beetles. They exhibit tunneling behavior and process herbivore dung, burying it deeply in the soil profile (Ma et al., 2023). Dung beetles have been referred to as ecosystem engineers, altering physical and chemical properties of the soil in ways that affect availability of resources to other organisms (Doube, 2018). Dung beetles also aerate soil, reduce greenhouse gas emissions, aid in control of flies and nematodes that affect livestock, and support secondary seed dispersal (Noriega et al., 2023). Conservation of dung beetle populations is also valued because their activity can reduce the survival of gastrointestinal parasites on pastures (Sands and Walls, 2017).

Dung decomposition and nutrient cycling is advantaged by a functionally diverse dung beetle community (Noriega et al., 2023). In South Dakota, USA, a diverse dung beetle community was credited with more effective dung degradation in grasslands, and early beetle colonization following dung deposition was key to maximizing success (Pecenka and Lundgren, 2018). The introduction of exotic dung beetles across southeastern Australia, where

native dung beetles are not efficient in processing livestock dung, reduced the quantity of dung on the soil surface (Ma et al., 2023). Arthropods that feed on dung accelerate the cycling of nutrients in grazed pasture (Tixier et al., 2016) and can increase herbage accumulation. For example, dung beetle presence increased soil N content and herbage accumulation of pearl millet [*Pennisetum glaucum* (L.) R. Br.] (Garcia et al., 2023). Likewise, herbage accumulation of oat (*Avena sativa* L.)-vetch (*Vicia benghalensis* L.) mixture in temperate Australia improved with dung beetle activity (Ma et al., 2023). To date, over 60 species of exotic dung beetles from both temperate and tropical environments have been introduced to Australian grasslands, with important positive impacts (Ma et al., 2023).

Increased use of N and P fertilizers on pastures may decrease belowground soil faunal diversity (Donnison et al., 2000). In New Zealand, greater quantity and quality of inputs, including P fertilization, disadvantaged fungal-based communities but increased the number of bacterial-feeding nematodes (Parfitt et al., 2010). Similarly, soils with greater N status may cause a shift from fungal-to-bacterial populations, increasing the rate of nutrient cycling and the risk of N leakage compared with soils containing fungal-based food webs, which dominate low-fertility pastures (van der Heijden et al., 2008).

Long-term differences in grazing management can influence microbial communities (Pan et al., 2018). Restriction from grazing in the semiarid grasslands of Inner Mongolia increased microbial biomass but did not affect microbial diversity (Liu et al., 2016). In the same study, ammonia-oxidizing bacteria and ammonia-oxidizing archaea had greater growth rates in soils of grazed versus grazing-restricted swards, but defoliation management effects were limited to the top 20 cm of soil. In Inner Mongolia, China, grazing intensity affected the distribution of nitrifying communities by impacting soil characteristics including bulk density and NH_4^+-N (Pan et al., 2018). For grasslands in Poland, the microbial community structure was sensitive to different systems of defoliation management (Musial et al., 2020), with actinobacteria growth increasing in mowed versus grazed swards, creating favorable conditions for greater release of inorganic N from its organic soil pool. Thus, microbial communities are sensitive to a range of pasture management practices, with implications for nutrient cycling in grassland ecosystems (Liu et al., 2016).

There are limited data on the contributions of arbuscular mycorrhizal fungi to nutrient cycling, but it is generally considered they enhance plant nutrition and growth. In microcosms containing Italian ryegrass [*Lolium* multiflorum (Lam.)], mycorrhizae reduced reactive and unreactive P leaching by 31%, enhanced plant P content by 15%, and soil P mobilization by 18%, reduced N_2O fluxes by 62% and NH_4^+ leaching by 75%, and increased plant N concentration by 13% (Bender et al., 2015; Table 2). Soil nutrient status affects root cell membrane integrity, quantity of root exudates, and mycorrhizal activity. When no N and

K were applied to three perennial pasture grasses, plant root exudations of citrate and malate increased leading to increased mycorrhizal colonization relative to grasses receiving N and K fertilizer (Tshewang et al., 2020). The literature suggests that increasing arbuscular mycorrhizae colonization may promote sustainable nutrient cycling, but there is a need to conduct process-based studies to better understand the mechanisms involved and how they are affected by management (Bender et al., 2015).

8 The role of nutrient cycling in the sustainability of grazed grasslands

The sustainability of grazed grasslands is threatened by a range of factors including overgrazing, soil erosion, depletion of soil organic matter, inadequate nutrient supply, soil compaction, and nutrient losses to the environment (Boddey et al., 2004; Franzluebbers et al., 2020). Efficient nutrient cycling greatly enhances sustainability of grazed grasslands (Sollenberger et al., 2019), but achieving this goal is particularly challenging because the presence of animals affects nutrient availability and spatial distribution (Dubeux et al., 2006b; Vertès et al., 2019). Concurrently, the size of nutrient pools and magnitude of nutrient fluxes vary widely in grazed grasslands and can be associated with divergent outcomes ranging from grassland degradation, when nutrient inputs and availability are inadequate, to significant nutrient losses to the environment when inputs are greater than uptake and storage capacity.

Several important strategies have been suggested for enhancing the efficiency of nutrient cycling and advancing the cause of grazed grassland sustainability. Avoiding overstocking is critical because greater stocking rates increase the proportion of nutrients cycling in excreta (Thomas, 1992), often increasing gaseous or leaching losses from the system (Dungait et al., 2012). Overstocking also reduces plant cover and above- and belowground biomass, triggering cascading effects including soil erosion, nutrient runoff and leaching, and decreasing soil organic matter which reduces nutrient storage (Sollenberger et al., 2012). The use of perennials is an important sustainability practice because they maintain year-round soil cover and root biomass, minimize runoff and leaching losses, and reduce a need for soil tillage (Picasso et al., 2022). Increased species richness has been associated with greater N retention and soil nitrate concentrations (Cong et al., 2014), increased resource-use efficiency, and greater resilience when grasslands encounter disturbance (Tracy et al., 2018). Increased use of legumes allows a reduction in synthetic fertilizer use in some cases and increases the amount of N cycling (Kohmann et al., 2018; 2019) and N cycling efficiency (Garcia et al., 2021). As a result, year-round grass–legume systems were able to sustain similar animal productivity as N-fertilized grasses despite receiving only 15% as much N via fertilizer (Jaramillo et al., 2021b).

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Considering grassland sustainability broadly, optimization of agricultural practices is essential to close biogeochemical cycles and minimize the negative impacts of nutrient cycle decoupling on the environment (Vertès et al., 2019). Ecologically oriented practices (e.g. integrated crop–livestock systems, grassland species diversification, and greater landscape complexity) can increase productivity and ecosystem services through diversification and greater nutrient-use efficiency (Franzluebbers et al., 2020). Conservation nutrient management (e.g. minimize soil disturbance, maximize soil cover and use of perennial plant species, and integrate legumes) can improve soil health, increase soil biological activity, and provide essential nutrients to forage crops such that nutrient input via fertilizers is considered a secondary choice (Franzluebbers et al., 2020).

9 Use of nutrient budgets to guide management of grazing systems: the case of nitrogen and phosphorus

A nutrient budget, also called nutrient mass balance, is the difference between inputs and outputs of nutrients of interest within specific spatial and time scales (Soberon et al., 2015). Results expressed on a per area basis indicate environmental impact, while on a per unit of product (meat, milk) basis indicate production efficiency (Soberon et al., 2015). Farm-to-gate nutrient budgets have been successfully applied in dairy operations to evaluate environmental risks and improve economic feasibility and nutrient-use efficiency (Soberon et al., 2013; Soberon et al., 2015; Spears et al., 2003a). Evaluated over time, they quantify the effect of changes in management practices and regulations in reducing environmental risks (Aarts et al., 2000; Kohmann et al., 2021; Oenema and Roest, 1998). Variability in nutrient budgets among farms with different management practices can also be used as an indicative for improvement in nutrient-use efficiency (Akert et al., 2020).

Most nutrient budgets are estimated at farm-to-gate scales. Positive nutrient budgets imply nutrient-use inefficiency and soil nutrient buildup with potential losses to water bodies or the atmosphere, increasing environmental risks (Carpenter et al., 1998; Soberon et al., 2015). However, losses of nutrients and associated environmental risks are subject to site-specific management factors such as rate, timing, type, and placement of nutrients, as well as weather conditions (Gourley et al., 2012a; Soberon et al., 2015). Van Leeuwen et al. (2019) demonstrated that field-level nutrient balances are more useful than farm-level when there is considerable within-farm variation in soil type or management practices, particularly because of variations in leaching associated with soil type. Moreover, high losses of nutrients at a field level can be underestimated when only farm-scale nutrient budgets are used.

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Van Leeuwen et al. (2019) indicate that hot spots and hot moments should be considered when there are large environmental risks associated with nutrient losses. Supporting this conclusion, Gourley et al. (2015) noted that areas from which dairy cows were restricted had lesser extractable nutrient concentration, but areas, where significant urine and dung were deposited, had greater environmental risk. These differences are not captured in farm-to-gate nutrient budgets (Gourley et al., 2012a). Negative budgets can indicate limitations for forage production and potential for pasture degradation (Kohmann et al., 2021; Soberon et al., 2015), but the risk of pasture degradation must be considered in the context of soil nutrient analysis. For example, negative nutrient budgets in areas with a history of soil nutrient loading might not result in degradation (Akert et al., 2020; Mu et al., 2016). Thus, the interpretation of farm-to-gate nutrient budgets is subject to factors such as soil analysis, soil nutrient saturation levels, climate, and management practices.

9.1 Nitrogen budgets

Various comparisons were made of farm-to-gate N budgets of two dairy farm systems in Ireland and four systems in Switzerland with a significant grazing component (Table 3). For systems that did not account for atmospheric deposition and had low pasture legume proportion with negligible BNF, chemical fertilizer and purchased feed made up a large proportion of N inputs (~97%; Mihailescu et al., 2014; Mu et al., 2016). When BNF and atmospheric deposition were considered, they made up 38–64% of total N inputs, suggesting the adoption of legumes can alleviate reliance on N for chemical fertilizer. Milk exports were the main N output (45–82%). Nitrogen balance was positive and highly variable across the systems (75–217 kg N ha^{-1} $year^{-1}$), and N-use efficiency was relatively low (22–57%). This suggests there is considerable unaccounted N and potential to improve both N balance estimates and N-use efficiency. It is important to note that these budgets did not consider N losses through leaching, runoff, or volatilization. Although this is a small selection of cases, it indicates improving estimates of N losses increases the total amount of N accounted for in dairy grazing systems and informs management decisions.

Management of animal excreta plays an important role in farm-to-gate N budgets. In grazing dairies, 75–90% of total excreted N is deposited directly on pasture while this proportion is far lower in confined systems (0–35%; Gourley et al., 2012a). Adjusting N and P fertilizer application to account for nutrients recycled through animal urine and dung in grazing systems is essential to avoid over-application that negates the benefits of nutrient cycling (van Leeuwen et al., 2019). Requiring manure application during spring instead of fall and adaptation of chemical N fertilizer recommendations to account for manure

Table 3 Average farm-level N inputs, outputs, balance, and efficiency (outputs divided by inputs, expressed in percentage) of various dairy systems with significant grazing in Ireland and Switzerland

Country	Ireland[a]	Ireland[b]	Switzerland[c]	Switzerland[d]	Switzerland[e]	Switzerland[f]
Number of farms	21	19	8	8	1	1
Years	3	1	3	3	3	3
			N inputs (kg ha^{-1})			
Concentrate fed	27	40.1	2.9	23.3	6.6	16.3
Roughage (hay, silage, wheat straw, alfalfa)	NA[g]	12.4	16.8	19.6	41.4	11.6
Forage	12.4			Accounted for under roughage		
Bedding material	4			Accounted for under roughage		
Chemical fertilizer	186	199	24.5	27.3	52	15.5
Organic fertilizer	0	0	17.6	33.2	21.4	17.9
Livestock	3.8	NA	6	5.5	2.4	2.3
N fixation	NA	NA	41.5	42.8	86.8	88.9
Atmospheric deposition	NA	8.1	22.5	23.8	25	25
Total inputs	**233**	**260**	**132**	**176**	**236**	**178**
			N outputs (kg ha^{-1})			
Milk	40	46.7	30.3	45.5	60	57.4
Livestock	12.8	11.4	10.7	9.8	10.5	11.4
Plant material	0	0	12.4	37.5	0	20.4
Manure	NA	NA	0	7.4	2.3	1.1
Total outputs	**53**	**58**	**53**	**100**	**73**	**90**
N balance (kg ha^{-1})	***180***	***217***	***78***	***75***	***163***	***87***
N-use efficiency (%)	***23***	***22***	***41***	***57***	***31***	***51***
Source	Mihailescu et al. (2014)	Mu et al. (2016)	Akert et al. (2020)	Akert et al. (2020)	Akert et al. (2020)	Akert et al. (2020)

Forage and grazing management for the farms in each column are described as: [a]pasture grazed and ensiled; [b]grass-based, wheat straw, grass silage, feed produced on-farm; [c]grazing for 230 days year^{-1}, minimum concentrate, commercial farms; [d]grazing for 202 days year^{-1}, hay and fresh herbage (192 days year^{-1}), minimum concentrate, commercial farms; [e]full-time grazing (24 h day^{-1}), minimum concentrate, experimental farm; [f]part-time grazing (6–8 h day^{-1}), hay and fresh herbage, minimum concentrate, experimental farm. [g]NA: not applicable, not considered, or negligible.

inputs were successful in reducing farm-level N balance in Ireland (Mihailescu et al., 2014). One of the limitations of farm-to-gate N budgets is they are often not sensitive enough to elucidate the risk of N losses from areas with concentrated N deposition.

Feeding strategies can also alter farm-to-gate N budgets, but results vary. Mu et al. (2016) reported the Irish dairy farms that rely on grazing have ~20% lower N inputs ha^{-1} and 10% lower N balance ha^{-1} than systems in the Netherlands that have a greater reliance on concentrate. In Switzerland, however, greater N inputs via concentrate increased N outputs due to greater milk yields per cow, resulting in lesser N balance and greater N-use efficiency in farms using part-time compared with full-time grazing (Akert et al., 2020).

9.2 Phosphorus budgets

Farm-to-gate P budgets for dairy systems are highly variable as shown in the examples from Germany, Northern Ireland, Ireland, and Switzerland (Table 4). They ranged from −10.6 to 17.3 kg P ha^{-1}, and P-use efficiency ranged from 268% to 37% (Table 4). Phosphorus inputs were mostly from animal feed (33–94%) or fertilizer (23–55%), with their proportions inversely related. Atmospheric P deposition (input) and runoff and leaching (output) were considered only in one study (Ohm et al., 2015). Most exports were from animal products (milk and livestock; ~85%) except for the case study in Germany where a large proportion of manure was exported outside the farm instead of re-distributed on grasslands (Ohm et al., 2015). Different from the N budgets, P budgets accounted for the majority of P and presented near zero budgets and ~100% P-use efficiencies.

Negative P budgets indicate the P exported is not being replenished at the farm level. However, the impact on forage production will depend on soil P status (Ohm et al., 2015, 2017). For example, in a dairy where soils were highly impacted and P was elevated, P mining by crops and grasslands can reduce risk of P losses to the environment in the short term, but over a longer time span, P inputs would likely be necessary (Ohm et al., 2015; 2017). Livestock operations that produce crops on-farm often have greater P-use efficiency, which has been suggested as the best management strategy to reduce P balance in dairy operations (Spears et al., 2003a). However, this is possible only if P fertilizer and manure applications are managed to match and not exceed crop requirements. For example, P-rate recommendations for forages in Florida (USA) are species-specific and consider both agronomic performance and environmental safety with a distinction between grazed and harvested systems (Silveira et al., 2011). This approach reflects the abundance of grazing systems in the region and the elevated risk of P pollution due to coarse-textured soils with low P-holding capacity (Obour et al., 2011; Silveira et al., 2011). Tailoring

Table 4 Average farm-level P inputs, outputs, balance, and efficiency (outputs divided by inputs, expressed in percentage) of various dairy systems with significant grazing in several countries

Country	Germany[a]	Northern Ireland[b]	Ireland[c]	Switzerland[d]	Switzerland[e]	Switzerland[f]	Switzerland[g]
Number of farms	1	83	19	8	8	1	1
Years	3	6	1	3	3	3	3
				P inputs (kg ha^{-1})			
Minerals	0.6	NA[h]	NA		In concentrate		
Concentrate fed	0.5	In concentrate	7.4	1.1	5.7	0.9	7.5
Roughage (hay, silage, wheat straw, alfalfa)	6.0	19.7	2.4	2.7	3.1	6.3	2.6
Beeding material	0.2	NA			In roughage		
Chemical fertilizer	NA	7.5	3.2	1.4	2.3	0	0
Organic fertilizer	NA	0	0	5	7.2	3.6	3.2
Seed	0.2	NA	NA	NA	NA	NA	NA
Livestock	NA	NA	NA	1.4	1.3	0.6	0.5
Atmospheric deposition	0.2	NA	NA	NA	NA	NA	NA
Total inputs	**6.6**	**27.2**	**13**	**11.6**	**19.6**	**11.4**	**6.3**
				P outputs (kg ha^{-1})			
Milk	3.9	7.6	8.3	5.9	8.1	10	10.6
Livestock	1.4	2.0	3.3	2.6	2.4	2.5	2.7
Plant material	NA	0.3	0	2.2	6.4	0	3.5
Manure	11.6	NA	NA	0	1.3	0.1	0.1

(Continued)

Table 4 (Continued)

Country	Germany[a]	Northern Ireland[b]	Ireland[c]	Switzerland[d]	Switzerland[e]	Switzerland[f]	Switzerland[g]
Runoff, leaching	0.3	NA	NA	NA	NA	NA	NA
Total outputs	**17.2**	**9.9**	**11.6**	**10.7**	**18.2**	**12.6**	**16.9**
P balance (kg ha^{-1})	***−10.6***	***17.3***	***1.4***	***0.9***	***1.4***	***−1.2***	***−10.6***
P-use efficiency (%)	***260***	***37***	***89***	***92***	***93***	***111***	***268***
Source	Ohm et al. (2015)	Adenuga et al. (2018)	Mu et al. (2016)	Akert et al. (2020)	Akert et al. (2020)	Akert et al. (2020)	Akert et al. (2020)

Forage and grazing management for the farms in each column are described as [a]grazing management: young stock grazing full time; lactating animals grazed 7 h day^{-1}; [b]>3/4 of total grazing livestock is dairy; [c]wheat straw, grass silage, feed produced on-farm; [d]grazing for 230 days year^{-1}, minimum concentrate, commercial farms; [e]grazing for 202 days year^{-1}, hay and fresh herbage (192 days year^{-1}), minimum concentrate, commercial farms; [f]full-time grazing (24 h day^{-1}), minimum concentrate, experimental farm; [g]part-time time grazing (6–8 h day^{-1}), hay and fresh herbage, minimum concentrate, experimental farm. [h]NA, not applicable, not considered, or negligible.

P-rate recommendations in this manner has been successful in reducing farm-to-gate P budgets to near neutrality, with the greatest benefit in operations that are self-sufficient in feed production (Kohmann et al., 2021). When on-farm feed production is not possible, exporting manure from operations that import feed rather than producing feed on the farm reduces the P balance, but this may not be advantageous from a watershed perspective (Spears et al., 2003a).

Farm-to-gate P budget results must be interpreted while considering within-farm animal management practices. In Australia, forage-based dairy farms typically use rotational stocking, but areas where animals are held and fed supplements have high stocking density, leading to nutrient buildup from animal excreta (Gourley et al., 2015). Calving, feeding, and holding areas in Australian dairy farms had 10 times greater Olsen P and 15 times greater Colwell K compared with grazed pastures. Similarly, in Wisconsin (USA), soil Bray-1 P and K were much greater in areas used for exercise and feeding compared with crop fields (Powell et al., 2005). Management practices focused on reducing erosion and runoff in areas with high soil P concentration can be an efficient way to reduce environmental risks (Carpenter et al., 1998).

10 Case study: nitrogen cycling in nitrogen-fertilized grass and grass–legume pastures

Few studies of N cycling in pastures have measured pools and fluxes across multiple seasons and years. Such measurements are challenging because N is highly transient and affected by biotic and abiotic factors (Galloway et al., 2004). Nitrogen is a critical element for plant and animal growth, but it is also implicated in NO_3^- leaching to ground water (Markov, 2012; Santos et al., 2023) and N_2O emissions (Crews and Peoples, 2004; Jensen et al., 2012). Therefore, quantifying N flow among different pools of the air–soil–plant–animal interface is important.

This case study derives from a series of experiments conducted in the mild and humid temperate region of northwestern Florida, USA (30.77°N latitude, 85.23°W longitude). Details of the case study were derived from Jaramillo et al. (2021a, 2021b), Garcia et al. (2021), and Garcia et al. (2023), and the N cycle for the two grazing systems is shown in Figure 1.

The study compares two well-established, long-term grazing systems, each having cool- and warm-season forage components that allow for nearly year-round grazing. One system included N-fertilized bahiagrass monoculture pastures in the summer, overseeded in the fall with an N-fertilized mixture of oat and Italian ryegrass for winter grazing (Grass + N). Total annual N fertilization in this system was 224 kg ha^{-1} (urea and polymer-coated urea), split equally in the cool and warm seasons. The second system (Grass + Legume) included a bahiagrass-rhizoma peanut (*Arachis glabrata* Benth) mixed pasture during the

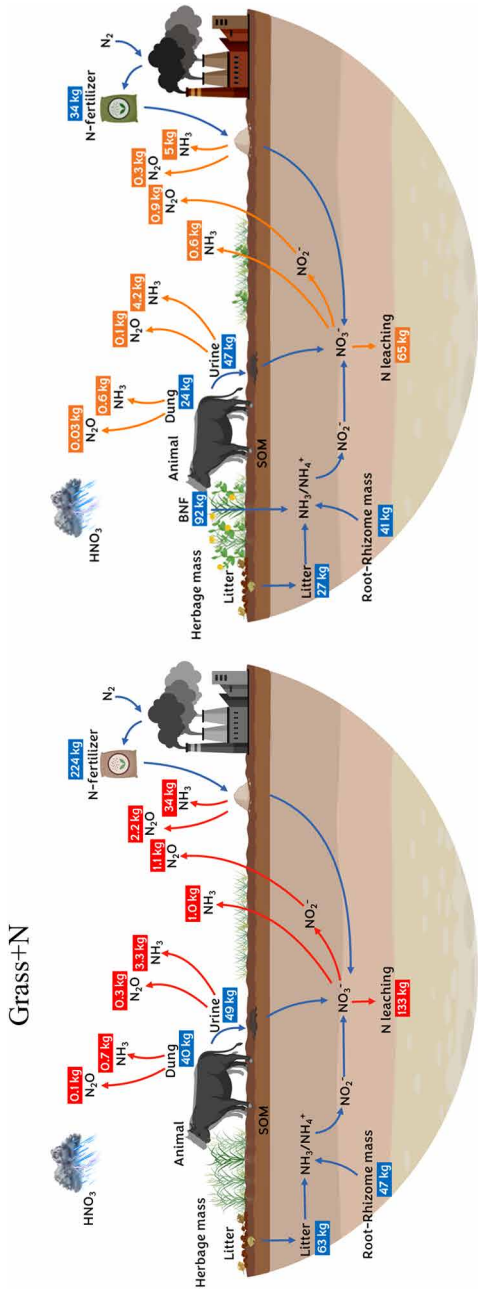

Figure 1 Nitrogen cycle in contrasting grazing systems. Multiple transformations take place in the N cycle. External N inputs (blue arrows) are industrial fertilizers (urea is usually used because it is rapidly converted to inorganic forms in the soil) and BNF (symbiosis between soil microorganisms and plant roots). Nitrogen outputs (red or orange arrows) are losses due to product harvesting (forage or animal), N in leachate and runoff, NH_3 volatilization, and N_2O emissions. Values are kg ha^{-1} $year^{-1}$.

warm season, overseeded in the fall with a mixture of oat and ryegrass plus a blend of crimson (*Trifolium incarnatum* L.), red (*T. pratense* L.), and ball clovers (*T. nigrescens* Viv.). During the cool season, Grass + Legume received 34 kg N ha^{-1} $year^{-1}$. Grass + Legume had 92 kg BNF ha^{-1} $year^{-1}$, so total N inputs were 224 and 126 kg ha^{-1} for Grass + N and Grass + Legume, respectively (Pereira Neto et al., 2024).

Pastures in both systems were continuously stocked at variable stocking rates to maintain constant forage allowance (Sollenberger et al., 2005). Nitrogen returning to pastures in dung was greater for Grass + N than Grass + Legume because of a greater dung N concentration and stocking rate in Grass + N (Garcia et al., 2021; Jaramillo et al., 2021b), while N excreted from urine was similar between systems (Table 5) (Garcia et al., 2021). The Grass + N system returned a greater amount of N in aboveground litter compared with Grass + Legume (Table 5) (Jaramillo et al., 2021a). This can be partially explained by greater herbage accumulation in Grass + N, which is often associated with greater litter deposition rates (Guretzky et al., 2014; Jaramillo et al., 2021a, 2021b).

The belowground mass in Grass + N and Grass + Legume systems returned a similar quantity of N (Table 5), and 43% of the warm-season N in Grass + Legume was from rhizoma peanut. This belowground pool acts as a nutrient source when BNF is limited, supporting aboveground growth (Dubeux et al., 2017). Belowground plant mass and microbial biomass contribute to the N cycle and are important in forming soil organic matter, the second largest N reservoir after the atmosphere (Bai and Cotrufo, 2022).

Nitrogen losses occur from fertilizers, the soil, and animal excreta via different pathways (Williams & Haynes, 1990), including ammonia (NH_3) volatilization, N_2O emission, and NO_3^- leaching. Greenhouse gas emissions from fertilizer were estimated using the IPCC emission factor (IPCC, 2019).

Ammonia loss, which is a main cause of low N-use efficiency from fertilizers, was greater for Grass + N than Grass + Legume (Table 6; Bouwman et al., 2002). After a fertilization event, there is an increase in soil N abundance

Table 5 Nitrogen fluxes from animal sources (dung and urine) and plant sources (aboveground and belowground litter) for Grass + N and Grass + Legume systems

Nitrogen fluxes, kg N ha^{-1} $year^{-1}$		System	
		Grass + N	Grass + Legume
Animal	Dung	40	24
	Urine	49	47
Plant	Aboveground litter	63	27
	Belowground litter	47	41

Source: Garcia et al. (2021); Jaramillo et al. (2021a; 2021b).

Table 6 Total N losses as NH_3 and N_2O volatilization from fertilizers, soil, dung, and urine, and losses as NO_3^- from leaching in Grass + N and Grass + Legume systems

			System	
Nitrogen losses, kg N ha^{-1} $year^{-1}$			Grass + N	Grass + Legume
N-fertilizer		NH_3	34	5
		N_2O	2.2	0.3
Soil		NH_3	1.0	0.6
		N_2O	1.1	0.9
Animal	**Dung**	NH_3	0.7	0.6
		N_2O	0.1	0.03
	Urine	NH_3	3.3	4.2
		N_2O	0.3	0.1
Nitrate leaching			133	65

Source: Aapted from Garcia et al. (2021); Trumpp (2024).

as well as an increase in pH, resulting in NH_3 volatilization (Robertson and Vitousek, 2009). The type of fertilizers applied and weather conditions play an important role in these emissions (Robertson and Vitousek, 2009). Similarly, N_2O emissions are predicted to be greater for Grass + N than Grass + Legume (Table 6).

Excreta also contributed to N losses. Urine N losses were greater in the Grass + Legume system, particularly in the form of N_2O, but NH_3 volatilization and N_2O emissions from dung were similar between systems (Table 6). Nitrogen emissions from excreta are dependent on quantity and composition. Animals grazing in Grass + Legume have a greater percentage of N excreted in the urine (Garcia et al., 2021), and the high concentration of soluble N in urine is more susceptible to losses, whereas N in dung is in an organic form and losses are typically smaller (Williams and Haynes, 1990). Lastly, N lost as NO_3^- by leaching was greater for Grass + N (133 kg N ha^{-1} $year^{-1}$) in comparison with Grass + Legume (65 kg N ha^{-1} $year^{-1}$) (Table 6), which is associated with the high soluble-N inputs in the form of fertilizer.

Finally, animal performance demonstrates that similar average daily gain and gain per area are possible between the two systems (Jaramillo et al., 2021b) (Table 7). Based on the results of this case study, the inclusion of legumes in grazing systems allowed an 85% reduction in N-fertilizer inputs without compromising animal productivity when compared with the N-fertilized grass monoculture. As observed in Table 7, the Grass + Legume gain per N input was 1.9-fold greater than in the Grass + N system. Thus, the integration of legumes is an alternative for sustainable intensification of livestock systems.

Table 7 Animal performance during the cool and warm seasons, average daily gain, gain per area, stocking rate, and gain per kilogram of N input for Grass + N and Grass + Legume systems

	System	
Item	Grass + N	Grass + Legume
Average daily gain, kg ha^{-1} day^{-1}	0.58	0.69
Gain per area, kg ha^{-1}	620	669
[a]Stocking rate, AU ha^{-1}	4.9[a]	3.7[b]
[b]Gain per kg of N input	2.8	5.3

[a]Means within a row followed by different letters are different ($P < 0.05$).
[b]Gain per kg of N input refers to the kg of animal produced per kilogram of N.
Source: Adapted from Jaramillo et al. (2021b).

11 Conclusion and future trends

Nutrient cycling is a critical supporting ecosystem service provided by grazed grasslands. The presence of animals and their consumption, digestion, and metabolism of forage affect nutrient pools and the rate of nutrient cycling, redistribution of nutrients from excreta deposition, and the risk of nutrient losses to the environment via leaching, runoff, and gaseous losses. Choice of grazing intensity (stocking rate or sward height), stocking method, species composition (including presence of legumes), arrangement of structural features of grazed pastures (shade, watering stations, etc.), and the use of nitrification inhibitors are potentially important tools to address these challenges and enhance the efficiency of nutrient cycling.

Nutrient cycling research will continue to be an important area of scientific inquiry going forward because important knowledge gaps remain for increasing the efficiency of nutrient cycling in grazed pastures. These gaps include (i) the impact of management practices on changes in diversity and abundance of soil fauna and microbial communities, (ii) the effect of soil fauna and microbial community changes on nutrient cycling, and (iii) the efficacy and practical value of chemical and biological nitrification inhibitors in reducing N losses and increasing N-use efficiency. Future studies might be categorized within broad themes of grassland sustainability, nutrient-use efficiency, and the role of soil fauna and microbial communities.

Within grassland sustainability, research is needed to address the role of legumes as an alternative to fossil-fuel-intensive inorganic N. Legumes can be an important component of a strategy to avoid inadequate N inputs leading to degradation of grazed grasslands (Boddey et al., 2004). Ideally, this can be achieved while reducing costs by reducing N-fertilizer inputs, sustaining animal performance, and increasing delivery of critical ecosystem services (Jaramillo et al., 2021b; Sollenberger and Dubeux, 2022). Determining the optimal proportion of legume, or indeed the optimal species composition of

a species-rich grassland, is critical information to achieve particular ecosystem, economic, or animal productivity goals.

Within the theme of nutrient-use efficiency, research is needed to address the efficacy, economic viability, and potential unintended consequences associated with the use of chemical and biological nitrification inhibitors. Data cited in this chapter show increases in N-use efficiency and reduction in leaching and gaseous emission losses associated with the use of nitrification inhibitors. Additional research is needed to establish optimal rates for chemical inhibitors and acceptable maximum residue levels in soil, plant, and animal products (Adhikari et al., 2021). Biological nitrification inhibition remains under-researched as a mitigation option in grazed pastures (Clough et al., 2020).

Understanding the role of soil fauna and soil microbial communities in nutrient cycling is only beginning to emerge, and the effects of management practices on their diversity and abundance are not well defined. Particularly in soils of low N fertility, invertebrates increase plant uptake of nutrients from surface-applied plant litter, and in compacted soils, tunneling organisms such as earthworms and dung beetles improve soil structure and nutrient availability (Schon et al., 2011). Much available research assessing the role of invertebrates in pasture nutrient cycling focuses on the activities of dung beetles, and studies are needed across a broader range of soil fauna. Microbial communities are sensitive to a range of pasture management practices, and the resultant changes in microbial communities have implications for nutrient cycling in grassland ecosystems that are not well understood (Liu et al., 2016).

12 Where to look for further information

The following articles provide useful additional information for subjects described in this chapter:

- Adhikari, K. P., Chibuike, G., Saggar, S., et al. (2021), Management and implications of using nitrification inhibitors to reduce nitrous oxide emissions from urine patches on grazed pasture soils – A review, *Sci. Total Environ*. 791: 148099.
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