Nitrogen Dynamics In Integrated Crop-Livestock Systems

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By nature, nitrogen (N) cycling is complex and prone to leaks. Nitrogen exists as gaseous, ionic, and solid forms, and N cycling is driven and regulated by biological, chemical, and physical processes. Nitrogen is a required element for all life forms – their size and/or reproductive success are related to N availability. Industrial N fertilizer production, N fixation during fossil fuel combustion, and legume use in agriculture have more than doubled the pre-industrial global rate at which atmospheric N_2 is transformed to biologically active pools (Vitousek et al., 1997).

In many undisturbed natural ecosystems, N cycling is relatively tight, with little loss to water or the atmosphere per unit of dry matter production. Losses often are greater in managed ecosystems, because humans increase system productivity, remove products, disrupt vegetation, and alter herbivore:land ratio.

Agriculture has been utterly transformed by the availability of manufactured fertilizers that are inexpensive and easy to transport and handle. These fertilizers severed the need for livestock and poultry manure in crop production (Oomen et al., 1998). Improved transport systems have allowed farmers to utilize distant markets, encouraging farm specialization and independence. Results can be seen in livestock density data. Increases of about 20% in the number of cows ha^{-1} of farmed land occurred between 1992 and 2002 on mid-sized USA dairy farms (Powell et al., 2007b). Western US dairy farms have grown quickly because of higher efficiency of investment in cows, rather than land, but also the availability of inexpensive labor. In turn, they rely more on contracts for removing manure than do more integrated farms in other areas.

Specialization and growth in productivity have lead to disrupted and inefficient N cycling. For example, in the mid 1990s the N surplus on Dutch dairy farms was 35 to 40 kg N 1000 kg^{-1} milk, or 350 to 400 kg N ha^{-1} (Van Keulen et al., 1996; Hilhorst et al., 2001). Nitrogen excess is typical in many large livestock farms in industrialized countries. The impacts are exacerbated by repeated manure applications on nearby fields (Sims et al., 2005) and by higher dairy cow population density in counties with less suitable cropland (Isik, 2004). Many considerations affect a farmer's decision about where to apply manure, including land tenure, presence of neighbors, and travel distance. Furthermore, considerable amounts of manure are not collected and managed, which may lead to concentrated sources of water contamination (Harter et al., 2001; Powell et al., 2005; Russelle et al., 2007c).

Local management of N has local, regional, and global impact (Carton and Jarvis, 2001), including nitrate (NO_3)-contaminated drinking water, eutrophied terrestrial ecosystems, lakes, streams, and estuaries, acidified surface soils, degraded atmospheric clarity and increased particulate load, increased global warming, and altered composition and damaged health of natural plant communities. These impacts have resulted in increased regulation, especially of large concentrated animal feeding operations (CAFOs) in the USA and animal feeding operations (AFOs) in the European Union (EU).

In the USA, the Environmental Protection Agency established nutrient management plan (NMP) requirements for CAFOs that require an active site-specific NMP, known application rates and nutrient contents of manures, soil testing, nonmanured buffers around surface water or connections to surface water, equipment maintenance, and on-site record keeping (Koelsch, 2005). Those rules "do not explicitly encourage or discourage manure transfer to off-farm users" (p.154), nor do they require or encourage a nutrient balance approach. Due tocross-border impacts, nutrient balance in Dutch agriculture moved from being a "good idea" to the "law" in 1998 (Kuipers et al., 1999).

Re-integrating crop and livestock farming, either within farms or among farms, can improve both N utilization and recycling, and can help reverse soil quality degradation, both from soil organic matter (SOM) losses (Larson et al., 1983) and excessive nutrient accumulation on livestock farms. There are tremendous opportunities to help agriculture transform into integrated systems that are economically and ecologically beneficial, and that are better able to meet both private and public goals (Allen et al., 2007; Franzluebbers, 2007; Russelle et al., 2007b; Sulc and Tracy, 2007). This paper summarizes key sources and products of N cycling, examples of within-farm and among-farm N cycling, and ways to achieve high N use efficiency (NUE) in integrated crop-livestock systems.

NITROGEN SOURCES

There are several sources of N on integrated crop-livestock systems: fertilizer, symbiotic and nonsymbiotic N_2 fixation, imported manure, SOM, purchased feed, and atmospheric deposition. Minor sources include run-on of water and sediment from adjacent properties through erosion or flooding, irrigation water, and shallow groundwater.

Fertilizer N

Fertilizer N has become the major source of N in most industrialized agricultural systems because of ease of shipping, handling, and on-farm management, predictable N supply (assuming that application methods, rate, and timing are appropriate), and relatively low cost. Unfortunately, as fertilizer availability increased in the past half century, farmers and their advisors began to ignore other N sources. Millions of hectares have received too much N, some of which has escaped the fields to cause environmental damage. Fertilizer management for arable and grassland crops has been fully covered in other publications, and recommendations vary with crops, soils, climate, and local laws.

Dinitrogen fixation

Annual symbiotic N_2 fixation was the primary source of biologically available N on Earth until recently (Vitousek et al., 1997). Annual and perennial legumes can fix hundreds of kg N ha^{-1} (Wedin and Russelle, 2007), but legumes also help limit overall N inputs because fixation is inversely proportional to N availability from other sources (Allos and Bartholemew, 1959; Russelle and Buzicky, 1988; Russelle et al., 2007c). Although we have large-scale estimates (e.g., Russelle and Birr, 2004), no reliable techniques exist for local, field-scale estimates of symbiotic N_2 fixation. Some of the fixed N moves off the farm as forage, feed, or food, but soil N accumulation reduces fertilizer need for subsequent crops (Lory et al., 1995). The fertilizer N equivalent (i.e., the reduction in fertilizer N requirement) from legumes in crop rotations and pastures is relatively well known (Thomas, 1992; Cadish et al., 1994; Mohr et al., 1999; Kelling and Schmitt, 2003).

Nonsymbiotic N_2 fixation is less well characterized and often is considered to account for inputs of less than 10 kg N ha^{-1} annually in temperate-region agriculture. In subtropical and tropical regions, however, there is increasing evidence that N_2 fixation in the rhizosphere and within nonlegumes can be substantial (de Oliviera et al., 2006).

Soil organic matter

Nitrogen mineralization from SOM provides N to crops in proportion to the amount of organic matter present in the soil (Wang et al., 2004), chemical characteristics of recently added plant residues, soil temperature and moisture conditions, and length of the period during which N mineralization can occur (Cabrera et al., 2005). Thus, under conditions typical of the Upper Midwest USA on fertile Mollisols, N uptake by maize (*Zea mays* L.) in the absence of manure and fertilizer N is often 80 to 150 kg N ha^{-1} (Carpenter-Boggs et al., 2000; Sawyer et al., 2006). In contrast, N supply from SOM on more weathered soils in the southeastern USA may be only 30 to 60 kg N ha^{-1} (Franzluebbers, 2007). Greater N mineralization must be matched with timely crop uptake and lower fertilizer N application to limit N loss. Soil under grazed perennial forages accumulates SOM faster than under nonharvested or mechanical removal (Franzluebbers and Steudemann, 2003).

Manure

Livestock and poultry manure, a mixture of organic and inorganic forms of N, has been a mainstay of agriculture for 8 to 10 millennia (Smith, 1995). Livestock feces contain primarily organic forms of N that mineralize at different rates. In dairy cattle manure, for example, nondigested plant residues are more recalcitrant than N from endogenous sources, such as intestinal flora and sloughed cells from the animal's digestive tract (Haynes and Williams, 1993; Whitehead, 1995; Powell et al., 2006b). The main N-containing compounds in urine, on the other hand, are urea, NH_4, creatinine, creatine, and hippuric acid (Whitehead, 1995). Urinary excretion protects the animal against NH_3 toxicity. As dairy cows consume more N, total N excretion increases linearly, but urinary N excretion increases exponentially (Kebreab et al., 2001).

Imported manure can serve as an important nutrient source on farms, and can increase the yield and quality of forages (Cherney et al., 2002; Macoon et al., 2002; VanWieringen et al., 2005). Manure additions can result in crop yields that exceed those obtained by fertilizer N, P, and K additions, perhaps because of improved soil conditions, micronutrient supply, or other effects. Like SOM, however, N mineralization from manure may produce excessive amounts of NO_3, which can accumulate to concentrations that are toxic to livestock (Schmidt et al., 2003) or be lost from the field.

Purchased feed

Imported feed concentrates and forage comprise a large N addition, because most dairy farmers choose to purchase high-protein concentrates, but grow the roughages for the ration. For example, purchased feed equaled 4 to 181 kg N ha^{-1}, representing between 2 and 32% of total farm N imports in New Zealand and the Netherlands, respectively (Ledgard et al., 1998). Purchased feed was 61% of total N import on a farm in New York, USA, that had 320 cows, 290 heifers, and 604 ha of cropland (Klausner et al., 1998), but purchased feed comprised 98% of total N input on large western USA dairy farms with no cropland (Spears et al., 2003).

Atmospheric deposition

It is often assumed than annual N input from atmospheric sources is small. However, dry deposition in the EU can be substantial (50 to 100 kg N ha^{-1}) downwind of livestock and poultry facilities (Ferm, 1998). Complete accounting for significant N sources requires that both wet and dry deposition be assessed if the land of interest is near significant NH_3 sources.

NITROGEN EXPORTS IN PRODUCTS

Milk and meat are the primary animal products from livestock farms, although excess crops and manure can be important exports. Because NUE decreases as trophic level rises, production of milk and meat protein has lower NUE than production of plant protein (Bleken et al., 2005). However, dairy cow herds exhibit a wide range of NUE for milk production (15 to 35%), whereas efficiency of feed N conversion to meat averages only 5 to 10% (Haynes and Williams, 1993; Whitehead, 1995).

Substantially more nutrients are removed by mechanical harvest than by grazing. Therefore, cutting-only management systems respond to higher N rates than grazing on sandy soils in Northern Germany (Rotz et al., 2005). Achieving optimum NUE in grazed forage requires less external N than in harvested forage because much of the forage N is returned by the livestock (Whitehead, 1995). This recycling improves N supply to the subsequent crop when perennial forages are rotated (Assmann et al., 2003).

NITROGEN LOSSES

Although exported products represent an economic pathway of N removal from the farm, other pathways can have negative outcomes. When N imports exceed N exports in marketable products, the risk of environmentally harmful N losses increases (Rotz et al., 2005). In a survey of 125 German farms, over one-half had annual N balances exceeding 30 to 50 kg N ha^{-1}, the established limits for soils with high or low estimated rainwater percolation, respectively (Eckert et al., 2000). Crop productivity was unrelated to the N balance, demonstrating the all-important role of management by the farmer. Similarly, no differences in herd size, cropland area, livestock density, or efficiency of manure collection were found between Wisconsin dairy farmers who applied N at recommended rates and those who applied excessive N (Powell et al., 2007a). Applications of available N ranged from 106 to 222 kg N ha^{-1} (Powell et al., 2007a) and was comprised of legume N credits (22 to 42%), manure N (20 to 40%), and fertilizer (32 to 48%). Excess N escapes the farm via several pathways.

Ammonia volatilization

Ammonia volatilization losses may be inevitable in animal agriculture because NH_3 is formed rapidly in manure. German legislation reportedly assumed default values of 28% loss of total N excretion for animal slurry and 40% for farmyard manure (Eckert et al., 2000). Ammonia losses can be reduced by separating urine and feces with special slatted barn floors (Rotz et al., 2006), covering manure storage basins, acidifying the manure, or trapping it in the bedding (Misselbrook and Powell, 2005; Janni et al., 2007; Russelle et al., 2007a). Misselbrook and Powell (2005) found that the physical characteristics of bedding material were more important than chemical characteristics. Least NH_3 was released from sand bedding, which had little absorbance but small and tortuous diffusion paths. Few measurements of NH_3 losses from dairy barns are available (e.g., Zhang et al., 2005), but estimates suggest these can represent nearly 40% of the total

N loss (Misselbrook et al., 2000). In Wisconsin, USA, apparent NH_3 loss from the barn floor was only 18% of total N excretion by dairy cows during winter, but 50% during summer (Moriera and Satter, 2006). Frequency of floor scraping did not affect NH_3 loss.

As the amount of urine in manure increases, so does the probability of NH_3 volatilization. Manure pH modification is too expensive and urease inhibitors are not yet reliable under field conditions, so diet modification appears to be the only effective abatement strategy in outside beef feedlots and dairy cow facilities (Parker et al., 2005). Creative, economically viable solutions for reducing NH_3 loss are needed, especially given the regulatory attention NH_3 has received in the EU and USA.

Runoff

Runoff of manure N from pastures or manured cropland can occur as dissolved ions or organic forms and as N attached to sediment or organic matter. Such runoff has contaminated surface waters (Carpenter et al., 1998), but runoff risk can be significantly reduced by full or partial incorporation of manure (Ross et al., 1979; Jokela and Côte, 1994; Bittman, et al., 2005). Use of a mechanical aerator before manure slurry application reduced N losses in natural runoff events by 40 to 80%, although concentrations in the first event after slurry application were high (van Vliet et al., 2006). A large percentage of dairy farmers in the north central and northeastern USA apply manure daily or weekly year-round because they lack storage facilities (Russelle, 1997; Dou et al., 2001; Turnquist et al., 2006). Incorporation is not possible on frozen soil, increasing the risk of manure runoff after rainfall or during snowmelt. Environmental risk is lessened by leaving nonmanured buffer areas near water, reducing or eliminating applications on steeper slopes, and enhancing soil surface roughness by tillage before manure application.

Leaching

Both NO_3 and dissolved organic N (DON) can be lost by leaching, and DON can comprise a substantial proportion of that loss (Siemens and Kaupenjohann, 2002). There have been few reports of DON loss, but concern over the human health impacts of NO_3 in drinking water has lead to considerable research and regulation of NO_3 leaching. Nitrate leaching was larger with manure than fertilizer N at low N rates (Thomsen et al., 1997), but losses can be similar when optimal practices are followed (Stopes et al., 2002). As demonstrated on sandy soils, leaching losses are generally directly related to the whole-farm N surplus (Wachendorf et al., 2004). Nitrate leaching was low in pastures at less than 550 grazing d ha^{-1} yr^{-1}, but increased quickly at higher stocking rates (Simon et al., 1997).

Subsurface tile drains cause large N losses in humid climates (van Es et al., 2006) and subhumid climates (Randall et al., 2000), and this drainage water can degrade fresh waters and continental shelf areas (Goolsby et al., 1999). Although losses can be decreased with lower rates of fertilizer N or manure, losses may continue to be unacceptably high under annual cropping (Jaynes et al., 2001; Tan et al., 2002). Deeply rooted perennial crops, like alfalfa (*Medicago sativa* L.) and C3 grasses, minimize N leaching losses during growth (Randall et al., 1997; Russelle et al., 2001; Crush et al., 2005). Leaching losses often rise dramatically after the perennial hay or pasture crop is terminated, but proper fertilizer, crop, and soil management can reduce the risk of NO_3 leaching (Mohr et al., 1999; Eriksen, 2001; Huggins et al., 2001; Sawchik, 2001; Angus et al., 2006).

Denitrification

Conversion of inorganic N to N_2 and N_2O occurs primarily through reduction of NO_3, but N_2O also can be produced during nitrification of NH_4 (Wrage et al., 2001). Emission of N_2 from agriculture is rarely measured and usually is considered only as an economic loss. Nitrous oxide, on the other hand, is a potent greenhouse gas and contributes to ozone depletion.

Solid manure and grazing release about 20 times more N_2O than from liquid and slurry manures (Mosier et al., 1998), although increasing hippuric acid concentration in urine can halve N_2O production (Kool et al., 2006). Lowering dietary N content will reduce both NH_3 and N_2O production, whereas manure injection greatly increases N_2O emission (Brink et al., 2001). Covering slurry with straw during storage caused a large increase in N_2O production, anaerobic digestion did not change NH_3 loss but decreased N_2O loss by 60%, and slurry aeration increased N_2O emission by about 125% and NH_3 volatilization by about 85% (Amon et al., 2006). Large-scale measurements in Ireland indicated that about 3% of the N from both fertilizer and manure slurry was lost as N_2O in intensively grazed grassland (Scanlon and Kiely, 2003). As governments agree to reduce greenhouse gas emissions, farmers need better methods of manipulating manure composition and application to minimize both NH_3 and N_2O losses.

HIGH EFFICIENCY N RECYCLING

Are we stuck with the incompatibility between good economic returns in livestock and crop farming and improved environmental outcomes? In the absence of significant modification of mainstream agricultural practices including manure redistribution, the answer appears to be “Yes” (Granstedt, 1995; Oomen et al., 1998; Bleken et al., 2005; Boody et al., 2005; Sims et al., 2005; Wolf et al., 2005).

Several advantages may accrue if a mixed crop-livestock farming mentality is adopted either within or among farms (Oomen et al., 1998): i) external inputs will be reduced (with the exception of higher transport costs for manure) and utilized more efficiently; ii) manipulation of animal manure composition will improve crop NUE; iii) short-term grasslands added to the crop rotation will reduce inorganic N accumulation and improve fodder yields; and iv) incorporation of legumes will help balance N inputs. According to GIS-based simulation model results, applying alternative farming practices (increased use of perennial forages, integration of livestock with arable cropping, and reduced fertilizer use) could reduce NO_3 leaching by two-thirds in Iowa, USA (Burkart et al., 2005), compared to the current maize-soybean [*Glycine max* (L.) Merr.] rotation.

Improved N balance in integrated crop-livestock systems will require reduced import of feed and fertilizer, better storage, distribution, and use of manure, and perhaps reduced local concentrations of livestock. The active use of a reliable comprehensive nutrient management plan is fundamental to success. Simulation modeling can help elucidate pathways that are difficult to measure (Hutson et al., 1998; Rotz et al., 2005).

Herd NUE

On most farms, improved herd NUE is required to improve overall NUE (Kohn et al., 1997; Spears et al., 2003). For example, Kohn et al. (1997) estimated that a 50% improvement of herd NUE would yield a similar improvement in whole-farm NUE and reduce losses per unit product by 36 to 40%, whereas doubling manure N availability to crops would improve whole-farm NUE by only 13% and reduce N losses by an equal amount. Spears et al. (2003) suggested that whole-dairy farm herd NUE would be improved by removing heifers, but heifers would need to be raised elsewhere, so the NUE of the entire dairy system would not necessarily change.

Herd NUE can be enhanced by a number of tactics, including improved ration formulation for different parts of the lactation cycle, milking three times daily, or simply feeding less crude protein (Dou et al., 1998; Jonker et al., 2002; Powell et al., 2006a). In a survey of 372 commercial dairy herds in the eastern USA, Jonker et al. (2002) found that feed NUE increased by 0.05 percentage units for each gram of N removed from the daily diet, reaching the extraordinary NUE of about 35% when herds were fed 100 g less crude protein per day than the recommended 499 g d^{-1} for these herds. Improved NUE by the herd has the crucial effect of enhancing N retention in manure and recycling in soil (Powell et al., 2006b). Reducing manure urinary N content also reduces manure pH, which helps reduce NH_3 volatilization (Paul et al., 1998).

High dietary protein also results in greater risk of both NH_3 and N_2O emission during manure storage (Külling et al., 2001) and after application to soil (Velthof et al., 2005). Other feed characteristics affect herd NUE. Misselbrook et al. (2005) found that condensed tannin shifted N excretion from urine to feces, because of reduced ruminal degradation. This, in turn, reduced NH_3 losses from the barn and from manured soil.

Manure transport

Manure redistribution is the most direct, but also the most problematic, way to achieve improved N cycling. The relationship between concentration of manure production and availability of cropland affects the distance over which manure must be hauled. For example, in the Chesapeake Bay watershed (on the eastern seaboard of the USA), average manure hauling distance from livestock farms was estimated to be 37 km if all crop farmers were willing to accept manure, but 120 km if only 20% were willing (Ribaudo et al., 2003). The economic distance for manure hauling depends on its form and the means of transport, but usually is considered to be 25 km or less in the USA. A majority of Wisconsin, USA, dairy farmers spread all manure on fields within a 5-minute driving distance from the manure storage, resulting in manure application to only 23 to 44% of the available cropland (Saam et al., 2005).

Transport distance is more important for manure application than for other operations, because each field requires multiple trips. On integrated crop-dairy farms in Wisconsin, farmers made 15 to 32 trips per field, even though average field sizes were only 2.0 to 4.8 ha (Powell et al., 2007a). Rented land now comprises over one-half of all farmed land in the USA, and these authors found that less manure was applied to nearby rented land than to nearby owned land, presumably because farmers did not want to invest the manure resource on land they might be unable to use in the future.

Crop NUE

Achieving proper manure distribution will not necessarily improve N cycling, however, because the synchrony between mineralization of organic N in manure (or accumulated soil organic matter) and crop N uptake can be nearly as large a problem as manure transport (McNeill et al., 2005). Nitrogen mineralization is affected by soil characteristics (e.g., texture and buffering capacity), manure placement and mixing in soil, weather, and prior organic additions.

Lack of synchrony between plant demand and N mineralization can exacerbate losses as N supply becomes more dominated by organic sources. Soil organic matter concentration increases with repeated manure additions, which brings an increased risk of NO_3 accumulation in soil during periods when annual crops are not present or are not actively absorbing N (Myers et al., 1997). Schröder et al. (2005) estimated that the fertilizer N value of dairy slurry would rise from 55-60% in the first year to 80% after 6 to 8 yr of application, due to residual organic N mineralization. This estimated N fertilizer value ranged from 40 to 70% in another study with a variety of manures (Gutser et al., 2005). Manure rate recommendations need to take this residual effect into account.

As SOM accumulates in integrated crop-livestock farming, so will the need to increase cropping intensity to absorb inorganic N not taken up by the principal crops, through multiple cropping, adding catch or cover crops, substituting more deeply-rooted crops, or growing perennials in the rotation (Wivstad et al., 2005). Cover crops are not reliable in areas with adverse growing conditions (Strock et al., 2004), but can be effective in capturing N and are more economically justifiable in mixed farming systems because they provide a source of feed (Franzluebbers, 2007).

Within-farm integration – a Dutch example

By the early 1990s, it was recognized that the largest imbalance of N in the EU occurred in the Netherlands. Livestock feed imports into the country exceeded fertilizer N applications in 1995 (Olsthoorn and Fong, 1998), resulting in an average national surplus of 295 kg N ha^{-1} on dairy farms (Wolf et al., 2005). In 1995, an N and P accounting system called MINAS was introduced, which included: i) gradual lowering of allowed N application rates (on dry sandy soil to 60 kg N ha^{-1} for arable crops and 140 kg N ha^{-1} for grassland; on other soils to 100 and 180 kg N ha^{-1}, respectively), ii) manure rates limited to 170 kg N ha^{-1} for arable crops and 250 kg N ha^{-1} for grassland, and iii) levies applied per kg of excess surplus N (Wolf et al., 2005).

The De Marke experimental farm has shown that N losses can be reduced while maintaining milk production (Hilhorst et al., 2001; Rotz et al., 2005). Whole-farm surpluses of only 10 kg N 1000 kg^{-1} milk were achieved primarily by improving herd and crop NUE (Rotz et al., 2005), leading to a decline in ground water NO_3 concentrations from 50 to 10 mg NO_3 L^{-1} within only 3 yr (Kuipers et al., 1999). Analysis of other farms made clear that significant overall N losses could be curtailed only by adoption of several of the De Marke technologies (Bleken et al., 2005; Rotz et al., 2005). Enclosing the manure storage and injecting manure reduced NH_3 volatilization, which allowed lower fertilizer N purchases, but did not reduce NO_3 leaching or denitrification.

Verhoeven et al. (2003) described an approach by farmer cooperatives in the Netherlands to voluntarily reduce environmental problems in a region-specific way. The 93 farms used the MINAS nutrient bookkeeping system to monitor import, export, and surplus N, and included assessment of on-farm feed. As expected, farms varied widely in these characteristics, resulting in surpluses of 162 - 560 kg N ha^{-1}, herd NUE of 8 - 24% (product N/dietary N), crop NUE of 33 - 78% (crop N/fertilizer + manure N), and farm-level NUE of 10 - 28% (exported N/imported N).

After only three years on 37 farms with reliable data, N surplus was reduced from an average of 345 to 247 kg N ha^{-1}, with no change in milk production (Verhoeven et al., 2003). There was no change in average milk NUE (18%), but crop NUE rose from 46 to 53%. These results were achieved by altering the fertilization and management of grass silage (to reduce N concentration and increase fiber by delaying harvest), managing feed concentrates to reduce dietary N, which reduced manure N and increased C:N ratio, and reducing application rates of purchased fertilizer N. These authors emphasized the importance of iterative fine-tuning and "re-balancing" of the dairy farm N cycle.

In a nationwide analysis on predicted long-term N losses, Wolf et al. (2005) evaluated two approaches, MINAS and a more restrictive low-emission approach. They concluded that the only significant effects were due to decreasing N applications to maize, that there was no practical difference in outcomes of N loss to surface and ground water between the two approaches, and that on dry sandy soils, ground water quality would be improved, but would not meet current drinking water guidelines under any cropping pattern. This latter finding highlights the risk in utilizing sandy soils for agriculture, whether or not manure is involved.

Among-farm integration

More common than within-farm integration is among-farm integration in Maine, because the enterprises of potato (*Solanum tuberosum* L.) production and dairy farming differ dramatically. Coupling these two farm types resulted in higher profits for both within only two years, because of increased herd sizes and higher yields of marketable potatoes (Hoshide et al., 2006). These collaborations typically began slowly, relied on basic trust between the farmers, and have required that the farms were separated by no more than about 25 km (Files and Smith, 2001). There are no reports of the NUE achieved in these integrated systems.

The effect of manure storage and treatment on N losses and subsequent N supply to crops and soil varies with manure composition and type (Kirchmann and Lundvall, 1998). For example, only 2% of the total N in swine dung was lost by volatilization when stored anaerobically and rapidly incorporated in the soil, but nearly 24% was lost during composting. A key to establishing trust between manure supplier and recipient is the provision of reliable manure analyses for N and other nutrients.

Other hurdles to establishing contracts with manure recipients include odors, concerns about weed seed and pathogen transmission, and logistical problems in moving and applying manure in a timely manner when roads are trafficable with heavy equipment and when soil structure in the field will not be compromised by wheel traffic. Unlike some EU countries, there are no mechanisms in the USA for guaranteed analysis and no government or market incentives to localize nutrient exchange. A recent article in a farming magazine covered some of the considerations that both suppliers and recipients should consider (Johnson, 2007).

Closing comments

The vast majority of literature reports on agricultural N cycling are based on relatively short-term experiments. Nitrogen cycling experiments may produce misleading results with regard to N losses when conducted on soils that are not near SOM equilibrium levels (Ledgard, 2001). Based on simulation modeling, Piñeiro et al. (2006) estimated that 880 kg N ha^{-1} had been lost from the Río de la Plata grasslands by livestock grazing beginning with European settlement. This unsettling scenario is supported by Hoglund (1985), who measured declining soil C and N under pasture after stocking density increased. However, losses due to arable cropping usually are larger, and can be replaced by integrating perennial pastures with annual cropping, as demonstrated by the 4-decade-long rotation trial at La Estanzuela, Uruguay (Sawchik, 2001). Sustainable solutions will depend on local climate, soils, and cropping alternatives.

Some means of achieving lower N losses (i.e., reduced fertilizer and feed purchases, improved herd NUE, use of free stall barns, bottom loading manure storage, and direct injection of manure into soil) can improve farm profitability (Rotz et al., 2005; Rotz and Oenema, 2006), but many (cover cropping, low emission barns, covered manure storage, better manure distribution and incorporation) increase net costs (Rotz et al., 2005; Rotz et al., 2006). Furthermore, farm profitability declines rapidly if N availability to the herd or crops is too restricted. Government incentives may be required to counterbalance these economic hurdles.

Ultimately, integrated crop-livestock farming should strive to achieve the characteristics of "globally-oriented agriculture," which include high productivity and efficiency through the biological mechanisms that regulate how these systems function (Oomen et al., 1998). Sustainable solutions to specialization of agriculture could be achieved by rearranging crop and livestock enterprises to reduce transport distances and take advantage of synergies, such as improved crop rotations for disease control, improved soil tilth with manure application, direct nutrient exchange among crop and livestock farms, etc. (Oomen et al., 1998). Beyond our lack of an adequate knowledge base, it also is unlikely such a rearrangement of farming will occur in the absence of manure marketing systems (Ribaudo et al., 2003) and significant government regulations or incentives (Oomen et al., 1998; Nord and Lanyon, 2003; Sims et al., 2005; Boody et al., 2005).

REFERENCES

Allen, V.G., M.T. Baker, E. Segarra, and C.P. Brown. 2007. Integrated crop-livestock systems in dry climates. Agron. J. 99:346-360.

Allos, H.F., and W.V. Bartholemew. 1959. Replacement of symbiotic fixation with available N. Soil Sci. 87:61-67.

Amon, B., V. Kryvoruchko, T. Amon, and S. Zechmeister-Boltenstern. 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agric. Ecosyst. Environ. 112:153-162.

Angus, J.F., T.P. Bolger, J.A. Kirkegaard, and M.B. Peoples. 2006. Nitrogen mineralisation in relation to previous crops and pastures. Austr. J. Soil Res. 44:355-365.

Assmann, T.S., P. Ronzelli Júnior, A. Moraes, A.L. Assmann, H.S. Koehler, and I. Sandini. 2003. Rendimento de milho em área de integração lavoura-pecuária sob o sistema plantio direto, em presença e ausência de trevo branco, pastejo e nitrogênio. R. Bras. Ci. Solo 27:675-683.

Bittman, S., L.J.P. van Vliet, C.G. Kowalenko, S. McGinn, D.E. Hunt, and F. Bounaix. 2005. Surface-banding liquid manure over aeration slots: a new low-disturbance method for reducing ammonia emissions and improving yield of perennial grasses. Agron. J. 97:1304-1313.

Bleken, M.A., H. Steinshamn, and S. Hansen. 2005. High nitrogen costs of dairy production in Europe: Worsened by intensification. Ambio 34:598-606.

Boody, G., B. Vondracek, D.A. Andow, M. Krinke, J. Westra, J. Zimmerman, and P. Welle. 2005. Multifunctional agriculture in the United States. BioScience 55:27-38.

Brink, C., C. Kroeze, and Z. Klimont. 2001. Ammonia abatement and its impact on emissions of nitrous oxide and methane in Europe – Part I: Method. Atmos. Environ. 35:6299-6312.

Burkart, M., D. James, M. Liebman, and C. Herndl. 2005. Impacts of integrated crop-livestock systems on nitrogen dynamics and soil erosion in western Iowa watersheds. J. Geophys. Res. 110: G01009, doi:10.1029/2004JG000008.

Cabrera, M.L., D.E. Kissel, and M.F. Vigil. 2005. Nitrogen mineralization from organic residues. J. Environ. Qual. 34:75-79.

Cadisch, G., R.M. Schunke, and K.E. Giller. 1994. Nitrogen cycling in a pure grass and a grass-legume mixture on a red latosol in Brazil. Trop. Grassl. 28:43-52.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol. Applic. 8:559-568.

Carpenter-Boggs, L., J.L. Pikul, M.F. Vigil, and W.E. Riedell. 2000. Soil nitrogen mineralization influenced by crop rotation and nitrogen fertilization. Soil Sci. Soc. Am. J. 64:2038-2045.

Carton, O.T., and S.C. Jarvis. 2001. Nitrogen and phosphorus cycles in agriculture. p. 3-13. *In* P. De Clercq et al. (eds.) Nutrient management legislation in European countries. Wageningen Press, Netherlands.

Cherney, D.J.R., J.H. Cherney, and E.A. Mikhailova. 2002. Orchardgrass and tall fescue utilization of nitrogen from dairy manure and commercial fertilizer. Agron. J. 94:405-412.

Crush, J.R., J.E. Waller, and D.A. Care. 2005. Root distribution and nitrate interception in eleven temperate forage grasses. Grass Forage Sci. 60:385-392.

de Oliveira, A.L.M., E.D. Canuto, S. Urquiaga, V.M. Reis, and J.I. Baldani. 2006. Yield of micropropagated sugarcane varieties in different soil types following inoculation with diazotrophic bacteria. Plant Soil 284:23-32.

Dou, Z., D.T. Galligan, C.F. Ramberg, Jr., C. Meadows, and J.D. Ferguson. 2001. A survey of dairy farming in Pennsylvania: Nutrient management practices and implications. J. Dairy Sci. 84:966-973.

Dou, Z., L.E. Lanyon, J.D. Ferguson, R.A. Kohn, R.C. Bostrom, and W. Chalupa. 1998. An integrated approach to managing nitrogen on dairy farms: Evaluating farm performance using the Dairy Nitrogen Planner. Agron. J. 90:573-581.

Eckert, H., G. Breitschuh, and D.R. Sauerbeck. 2000. Criteria and standards for sustainable agriculture. J. Plant Nutr. Soil Sci. 163:337-351.

Eriksen, J. 2001. Nitrate leaching and growth of cereal crops following cultivation of contrasting temporary grasslands. J. Agric. Sci. Cambridge 136:271-281.

Ferm, M. 1998. Atmospheric ammonia and ammonium transport in Europe and critical loads: A review. Nutr. Cycling Agroecosyst. 51:5-17.

Files, A.C., and S.N. Smith. 2001. Agricultural integration: Systems in action. Maine Agric. Center Publ. 002. The Univ. of Maine, Orono. Online: www.mac.umaine.edu/publications/AgIntegration%20Systems.pdf

Franzluebbers, A.J. 2007. Integrated crop-livestock systems in the Southeastern USA. Agron. J. 99:361-372.

Franzluebbers, A.J., and J.A. Stuedemann. 2003. Bermudagrass management in the Southern Piedmont USA: VI. Soil-profile inorganic nitrogen. J. Environ. Qual. 32:1316-1322.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.P. Hooper, D.R. Keeney, and G.J. Stensland. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin – Topic 3 report for the integrated assessment on hypoxia in the Gulf of Mexico. Silver Spring, MD, NOAA Coastal Ocean Office Prog. Decision Anal. Ser. No. 17.

Granstedt, A. 1995. Studies of the flow, supply and losses of nitrogen and other plant nutrients in conventional and ecological agricultural systems in Sweden. p. 51-67. *In* L. Kristensen et al. (eds.) Nitrogen leaching in ecological agriculture. A.B. Academic Publ., UK.

Gutser, R., T. Ebertseder, A. Weber, M. Schraml, and U. Schmidhalter. 2005. Short-term and residual availability of nitrogen after long-term application of organic fertilizers on arable land. J. Plant Nutr. Soil Sci. 168:439-446.

Harter, T., M.C. Mathews, and R.D. Meyer. 2001. Effects of dairy manure nutrient management on shallow groundwater nitrate: A case study. ASAE Paper Number 01-2192. 30 July-1 Aug. 2001. 2001 ASAE Annual Int. Meeting, Sacramento, CA.

Haynes, R.J., and P.H. Williams. 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. Adv. Agron. 49:119-199.

Hilhorst, G., J. Oenema, and H. van Keulen. 2001. Nitrogen management on experimental dairy farm 'De Marke': Farming system, objectives and results. Neth. J. Agric. Sci. 49:135-151.

Hoglund, J.H. 1985. Grazing intensity and soil nitrogen accumulation. Proc. N.Z. Grassl. Assoc. 46:65-69.

Hoshide, A.K., T.J. Dalton, and S.N. Smith. 2006. Profitability of coupled potato and dairy farms in Maine. Renew. Agric. Food Syst. 21:261-272.

Huggins, D.R., G.W. Randall, and M.P. Russelle. 2001. Subsurface drain losses of water and nitrate following conversion of perennials to row crops. Agron. J. 93:477-486.

Hutson, J.L., R.E. Pitt, R.K. Koelsch, J.B. Houser, and R.J. Wagenet. 1998. Improving dairy farm sustainability II: Environmental losses and nutrient flows. J. Prod. Agric. 11:233-239.

Isik, M. 2004. Environmental regulation and the spatial structure of the U.S. dairy sector. Am. J. Agric. Econ. 86:949-962.

Janni, K.A., M.I. Endres, J.K. Reneau, and W.W. Schoper. 2007. Compost dairy barn layout and management recommendations. Appl. Eng. Agric. 23:97-102.

Jaynes, D.B., T.S. Colvin, D.L. Karlen, C.A. Cambardella, and D.W. Meek. 2001. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. J. Environ. Qual. 30:1305-1314.

Johnson, R. 2007. Ask right questions to make manure exchange work. Agri-View. 10 May 2007. Online: www.agriview.com/articles/2007/05/10/dairy_news/producer01.txt

Jokela, W.E., and D. Côte. 1994. Options for direct incorporation of manure. p. 210-215. *In* Liquid manure application systems. Design, management, and environmental impact. Northeast Reg. Agric. Eng. Serv., Ithaca, NY.

Jonker, J.S., R.A. Kohn, and J. High. 2002. Dairy herd management practices that impact nitrogen utilization efficiency. J. Dairy Sci. 85:1218-1226.

Kebreab, E., J. France, D.E. Beaver, and A.R. Castillo. 2001. Nitrogen pollution by dairy cows and its mitigation by dietary manipulation. Nutr. Cycl. Agroecosyst. 60:275-285.

Kelling, K.A., and M.A. Schmitt. 2003. Applying manure to alfalfa: Pros, cons and recommendations for three application strategies. North Central Reg. Res. Rep. 346. Online: www.soils.wisc.edu/extension/publications/Manure%20Alfalfa.pdf

Kirchmann, H., and A. Lundvall. 1998. Treatment of solid animal manures: Identification of low NH_3 emission practices. Nutr. Cycl. Agroecosyst. 51:65-71.

Klausner, S.D., D.G. Fox, C.N. Rasmussen, R.E. Pitt, T.P. Tylutki, P.E. Wright, L.E. Chase, and W.C. Stone. 1998. Improving dairy farm sustainability I: An approach to animal and crop nutrient management planning. J. Prod. Agric. 11:225-233.

Koelsch, R. 2005. Evaluating livestock system environmental performance with whole-farm nutrient balance. J. Environ. Qual. 34:149-155.

Kohn, R.A., Z. Dou, J.D. Ferguson, and R.C. Bostrom. 1997. A sensitivity analysis of nitrogen losses from dairy farms. J. Environ. Manag. 50:417-428.

Kool, D.M., E. Hoffland, E.W.J. Hummelink, and J.W. van Groenigen. 2006. Increased hippuric acid content of urine can reduce soil N_2O fluxes. Soil Biol. Biochem. 38:1021-1027.

Kuipers, A., F. Mandersloot, and R.L.G. Zom. 1999. An approach to nutrient management on dairy farms. J. Anim. Sci. 77 (Suppl. 2):84-89.

Külling, D.R., H. Menzi, T.F. Kröber, A. Neftel, F. Sutter, P. Lischer, and M. Kreuzer. 2001. Emissions of ammonia, nitrous oxide and methane from different types of dairy manure during storage as affected by dietary protein content. J. Agric. Sci., Cambridge 137:235-250.

Larson, W.E., F.J. Pierce, and R.H. Dowdy. 1983. The threat of soil erosion to long-term crop production. Science 219:458-465.

Ledgard, S.F. 2001. Nitrogen cycling in low input legume-based agriculture, with emphasis on legume/grass pasture. Plant Soil 228:43-59.

Ledgard, S.F., J.R. Crush, and J.W. Penno. 1998. Environmental impacts of different nitrogen inputs on dairy farms and implications for the Resource Management Act of New Zealand. Environ. Pollution 102 (S1):515-519.

Lory, J.A., G.W. Randall, and M.P. Russelle. 1995. Crop sequence effects on response of corn and soil inorganic nitrogen to fertilizer and manure nitrogen. Agron. J. 87:876-883.

Macoon, B., K.R. Woodard, L.E. Sollenberger, E.C. French, III, K.M. Portier, D.A. Graetz, G.M. Prine, and H.H. Van Horn, Jr. 2002. Dairy effluent effects on herbage yield and nutritive value of forage cropping systems. Agron. J. 94:1043-1049.

McNeill, A.M., J. Eriksen, L. Bergström, K.A. Smith, H. Marstorp, H. Kirchmann, and I. Nilsson. 2005. Nitrogen and sulfur management: Challenges for organic sources in temperate agricultural systems. Soil Use Manag. 21:82-93.

Misselbrook, T.H., and J.M. Powell. 2005. Influence of bedding material on ammonia emissions from cattle excreta. J. Dairy Sci. 88:4304-4312.

Misselbrook, T.H., J.M. Powell, G.A. Broderick, and J.H. Grabber. 2005. Dietary manipulation in dairy cattle: Laboratory experiments to assess the influence on ammonia emissions. J. Dairy Sci. 88:1765-1777.

Misselbrook, T.H., T.J. von-der Weerden, B.F. Pain, S.C. Jarvis, B.J. Chambers, K.A. Smith, V.R. Phillips, and T.G.M. Demmers. 2000. Ammonia emission factors for UK agriculture. Atmos. Environ. 34:871-880.

Mohr, R.M., M.H. Entz, H.H. Janzen, and W.J. Bullied. 1999. Plant-available nitrogen supply as affected by method and timing of alfalfa termination. Agron. J. 91:622-630.

Moriera, V.R., and L.D. Satter. 2006. Effect of scraping frequency in a freestall barn on volatile nitrogen loss from dairy manure. J. Dairy Sci. 89:2579-2587.

Mosier, A.R., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O. Van Cleemput. 1998. Closing the global N_2O budget: Nitrous oxide emissions through the agricultural nitrogen cycle. Nutr. Cycl. Agroecosyst. 52:225-248.

Myers, R.J.K., M. Van Noordwijk, and P. Vityakon. 1997. Synchrony of nutrient release and plant demand: Plant litter quality, soil environment and farmer management options. p. 215-229. *In* G. Cadisch and K. Giller (eds.) Driven by Nature: Plant litter quality and decomposition. CAB Int., Wallingford, UK.

Nord, E.A., and L.E. Lanyon. 2003. Managing material transfer and nutrient flow in an agricultural watershed. J. Environ. Qual. 32:562-570.

Olsthoorn, C.S.M., and N.P.K. Fong. 1998. The anthropogenic nitrogen cycle in the Netherlands. Nutr. Cycl. Agroecosyst52:269-276.

Oomen, G.J.M., E.A. Lantinga, E.A. Goewie, and K.W. Van der Hoek. 1998. Mixed farming systems as a way towards a more efficient use of nitrogen in European Union agriculture. Environ. Pollution. 102 (S1):697-704.

Parker, D.B., N.A. Cole, B.H. Baek, J.A. Koziel, M.B. Rhoades, Z. Perschbach-Buser, L.W. Greene, P. Sambana, and J.M. Sweeten. 2005. Abatement methods to reduce ammonia emissions from open-lot feedyards and dairies. *In* Symp. of the State of the Science on Animal Manure and Waste Management. 5-7 Jan. 2005, San Antonio, TX.

Paul, J.W., N.E. Dinn, T. Kannangara, and L.J. Fisher. 1998. Protein content in dairy cattle diets affects ammonia losses and fertilizer nitrogen value. J. Environ. Qual. 27:528-534.

Piñeiro, G., J.M. Paruelo, and M. Oesterheld. 2006. Potential long-term impacts of livestock introduction on carbon and nitrogen cycling in grasslands of Southern South America. Global Change Biol. 12:1267-1284.

Powell, J.M., D.B. Jackson-Smith, D.F. McCrory, H. Saam, and M. Mariola. 2006a. Validation of feed and manure management data collected on Wisconsin dairy farms. J. Dairy Sci. 89:2268-2278.

Powell, J.M., D.B. Jackson-Smith, D.F. McCrory, H. Saam, and M. Mariola. 2007a. Nutrient management behavior on Wisconsin dairy farms. Agron. J. 99:211-219.

Powell, J.M., D.F. McCrory, D.B. Jackson-Smith, and H. Saam. 2005. Manure collection and distribution on Wisconsin dairy farms. J. Environ. Qual. 34:2036-2044.

Powell, J.M., M.P. Russelle, and N.P. Martin. 2007b. Case Study: US dairy industry. *In* Livestock in a changing landscape: Drivers, consequences and responses. FAO. (in press).

Powell, J.M., M.A. Wattiaux, G.A. Broderick, V.R. Moreira, and M.D. Casler. 2006b.

Dairy diet impacts on fecal chemical properties and nitrogen cycling in soils. Soil Sci. Soc. Am. J. 70:786-794.

Randall, G.W., D.R. Huggins, M.P. Russelle, D.J. Fuchs, W.W. Nelson, and J.L. Anderson. 1997. Nitrate losses through subsurface tile drainage in Conservation Reserve Program, alfalfa, and row crop systems. J. Environ. Qual. 26:1240-1247.

Randall, G.W., T.K. Iragavarapu, and M.A. Schmitt. 2000. Nutrient losses in subsurface drainage water from dairy manure and urea applied for corn. J. Environ. Qual. 29:1244-1252.

Ribaudo, M., N. Gollehon, M. Aillery, J. Kaplan, R. Johansson, J. Agapoff, L. Christensen, V. Breneman, and M. Peters. 2003. Manure management for water quality: Costs of animal feeding operations of applying manure nutrients to land. USDA-Econ. Res. Serv. Agric. Econ. Rep. No. AER824. Online: www.ers.usda.gov/publications/aer824/

Ross, I.J., S. Sizemore, J.P. Bowden, and C. Haan. 1979. Quality of runoff from land receiving surface application and injection of liquid dairy manure. Trans ASAE 22:1058-1062.

Rotz, C.A., and J. Oenema. 2006. Predicting management effects on ammonia emissions from dairy and beef farms. Trans. ASABE 49:1139-1149.

Rotz, C.A., J. Oenema, and H. van Keulen. 2006. Whole farm management to reduce nutrient losses from dairy farms: A simulation study. Appl. Eng. Agric. 22:773-784.

Rotz, C.A., F. Taube, M.P. Russelle, J. Oenema, M.A. Sanderson, and M. Wachendorf. 2005. Whole-farm perspectives of nutrient flows in grassland agriculture. Crop Sci. 45:2139-2159.

Russelle, M.P. 1997. Nutrient management of humid, temperate region forages: Recommendations and practices. Proc. 18th Int. Grassl. Cong., Winnipeg, Manitoba, Canada. CDROM

Russelle, M.P., and A.S. Birr. 2004. Large-scale assessment of symbiotic dinitrogen fixation by crops: Soybean and alfalfa in the Mississippi River Basin. Agron. J. 96:1754-1760.

Russelle, M.P., and G.C. Buzicky. 1988. Legume response to fresh dairy cow excreta. p. 166-170. Proc. Am. Forage and Grassland Council.

Russelle, M.P., K. Blanchet, and L.A. Everett. 2007a. Characteristics and fertilizer value of compost dairy barn manure. p. 75-81. Proc. National Compost Dairy Barn Conf., 21-22 June 2007, Burnsville, MN. University of Minnesota, St. Paul.

Russelle, M.P., M.H. Entz, and A.J. Franzluebbers. 2007b. Reconsidering integrated crop-livestock systems in North America. Agron. J. 99:325-334.

Russelle, M.P., J.F.S. Lamb, B.R. Montgomery, D.W. Elsenheimer, B.S. Miller, and C.P. Vance. 2001. Alfalfa rapidly remediates excess inorganic N at a fertilizer spill site. J. Environ. Qual. 30:30-36.

Russelle, M.P., J.F.S. Lamb, N.B. Turyk, B.H. Shaw, and B. Pearson. 2007c. Managing nitrogen contaminated soils: Benefits of N_2-fixing alfalfa. Agron. J. 99:738-746.

Saam, H., J.M. Powell, D.B. Jackson-Smith, W.L. Bland, and J.L. Posner. 2005. Use of animal density to estimate nutrient recycling ability of Wisconsin farms. Agric. Syst. 84:343-357.

Sawchik, J. 2001. Dinámica del nitrógeno en la rotación cultivo-pastura bajo laboreao convencional y siembra directa. p. 211-224. *In* R. Díaz Rosello (ed.) Siembar directa en el Cono Sur. PROCISUR-IICA, Montivedeo. Online: www.inia.org.uy/siembra/sawchik2.pdf

Sawyer, J., E. Nafziger, G. Randall, L. Bundy, G. Rehm, and B. Joern. 2006. Concepts and rationale for regional nitrogen rate guidelines for corn. PM2015.

Scanlon, T.M., and G. Kiely. 2003. Ecosystem-scale measurements of nitrous oxide fluxes for intensely grazed, fertilized grassland. Geophys. Lett. 30:doi:10.1029/2003GL017454.

Schmidt, K., M.P. Russelle, C.C. Sheaffer, and C.E. Christians. 2003. Risks from high manure application before pasture restoration. Am. Forage and Grassland Conf. p. 233-237.

Schröder, J.J., A.G. Jansen, and G.J. Hilhorst. 2005. Long-term nitrogen supply from cattle slurry. Soil Use Manag. 21:196-204.

Siemens, J., and M. Kaupenjohann. 2002. Contribution of dissolved organic nitrogen to N leaching from four German agricultural soils. J. Plant Nutr. Soil Sci. 165:675-681.

Simon, J.C., F. Vertès, M.L. Decau, and L. le Corre. 1997. Les flux d'azote au pasturage. I. Bilans a l'expoitation et lessivage du nitrate sous prairies. [Nitrogen flows in grazed pastures. I. Balances on the farm scale and nitrate leaching under pastures.] Fourrages 151:249-262.

Sims, J. T., L. Bergström, B.T. Bowman, and O. Oenema. 2005. Nutrient management for intensive animal agriculture: Policies and practices for sustainability. Soil Use Manag. 21:141-151.

Smith, B.C. 1995. The emergence of agriculture. Scientific American Library, New York.

Spears, R.A., R.A. Kohn, and A.J. Young. 2003. Whole-farm nitrogen balance on Western dairy farms. J. Dairy Sci. 86:4178-4186.

Stopes, C., E.I. Lord, L. Phillips, and L. Woodward. 2002. Nitrate leaching from organic farms and conventional farms following best practice. Soil Use Manag. 18:256-263.

Strock, J.S., P.M. Porter, and M.P. Russelle. 2004. Cover cropping to reduce nitrate loss through subsurface drainage in the Northern US Corn Belt. J. Environ. Qual. 33:1010-1016.

Sulc, R.M., and B.F. Tracy. 2007. Integrated crop-livestock systems in the U.S. Corn Belt. Agron. J. 99:335-345.

Tan, C.S., C.F. Drury, W.D. Reynolds, P.H. Groenevelt, and H. Dadfar. 2002. Water and nitrate loss through tiles under a clay loam soil in Ontario after 42 years of consistent fertilization and crop rotation. Agric. Ecosyst. Environ. 93:121-130.

Thomas, R.J. 1992. The role of the legume in the nitrogen cycle of productive and sustainable pastures. Grass Forage Sci. 47:133-142.

Thomsen, I.K., J.F. Hansen, V. Khellerup, and B.T. Christensen. 1997. Effects of cropping system and rates of nitrogen in animal slurry and mineral fertilizer on nitrate leaching losses from a sandy loam. Soil Use Manag. 9:53-58.

Turnquist, A., J. Foltz, and C. Roth. 2006. Manure management on Wisconsin farms. PATS Res. Rep. No. 15. Univ. of Wisconsin-Madision.

van Es, H.M., J.M. Sogbedji, and R.R. Schindelbeck. 2006. Effect of manure application timing, crop, and soil type on nitrate leaching. J. Environ. Qual. 35:670-679.

Van Keulen, H., H.G. Van der Meer, and I.J.M. De Boer. 1996. Nutrient balances of livestock production systems in the Netherlands. p. 3-18. *In* A.F. Groen and J. Van Bruchem (eds.) Utilization of local feed resources in dairy cattle. EAAP Public. No. 84. Wageningen Press, Netherlands.

Van Vliet, L.J.P., S. Bittman, G. Derksen, and C.G. Kowalenko. 2006. Aerating grassland before manure application reduces runoff nutrient loads in a high rainfall environment. J. Environ. Qual. 35:903-911.

VanWieringen, L.M., J.H. Harrison, T. Nennich, D.L. Davidson, L. Morgan, S. Chen, M. Bueler, and F. Hoisington. 2005. Manure management effects on grass production, nutritive content, and soil nitrogen for a grass-based dairy farm. J. Environ. Qual. 34:164-173.

Velthof, G.L., J.A. Nelmans, O. Oenema, and P.J. Kuikman. 2005. Gaseous nitrogen and carbon losses from pig manure derived from different diets. J. Environ. Qual. 34:698-706.

Verhoeven, F.P.M., J.W. Reijs, and J.D. Van Der Ploeg. 2003. Re-balancing soil-plant-animal interactions: Towards reduction of nitrogen losses. Neth. J. Agric. Sci. 51:147-164.

Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and D.G. Tilman. 1997. Human alteration of the global nitrogen cycle: sources and consequences. Ecol. Applic. 7:737-750.

Wachendorf, M., M. Buchter, H. Trott, and F. Taube. 2004. Performance and environmental effects of forage production on sandy soils. II. Impact of defoliation system and nitrogen input on nitrate leaching losses. Grass Forage Sci. 59:56-68.

Wang, W.J., C.J. Smith, and D. Chen. 2004. Predicting soil nitrogen mineralization dynamics with a modified double exponential model. Soil Sci. Soc. Am. J. 68:1256-1265.

Wedin, D.A., and M.P. Russelle. 2007. Nutrient cycling in forage production systems. p. 137-148. *In* R.F. Barnes et al. (ed.) Forages; The science of grassland agriculture. Vol. II. Blackwell Publ., Ames, IA.

Whitehead, D.C. 1995. Grassland nitrogen. CAB International, Oxon, United Kingdom.

Wivstad, M., A.S. Dahlin, and C. Grant. 2005. Perspectives on nutrient management in arable farming systems. Soil Use Manag. 21:113-121.

Wolf, J., R. Rötter, and O. Oenema. 2005. Nutrient emission models in environmental policy evaluation at different scales – experience from the Netherlands. Agric. Ecosyst. Environ. 105:291-306.

Wrage, N., G.L. Velthof, M.L. Van Beusichem, and O. Oenema. 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol. Biochem. 33:1723-1732.

Zhang, G., J.S. Strom, B. Li, H.B. Rom, S. Morsing, P. Dahl, and C. Wang. 2005. Emission of ammonia and other contaminant gases from naturally ventilated dairy cattle buildings. Biosyst. Eng. 92:355-364.