Subsurface Application of Manures Slurries for Conservation Tillage and Pasture Soils and Their Impact on the Nitrogen Balance

Curtis J. Dell,* John J. Meisinger, and Douglas B. Beegle

Injection of cattle and swine slurries can provide soil incorporation in no-till and perennial forage production. Injection is expected to substantially reduce N loss due to ammonia (NH_3) volatilization, but a portion of that N conservation may be offset by greater denitrification and leaching losses. This paper reviews our current knowledge of the impacts of subsurface application of cattle and swine slurries on the N balance and outlines areas where a greater understanding is needed. Several publications have shown that liquid manure injection using disk openers, chisels, or tines can be expected to reduce NH_3 emissions by at least 40%, and often by 90% or more, relative to broadcast application. However, the limited number of studies that have also measured denitrification losses have shown that increased denitrification with subsurface application can offset as much as half of the N conserved by reducing NH_3 emissions. Because the greenhouse gas nitrous oxide (N_2O) is one product of denitrification, the possible increases in N_2O emission with injection require further consideration. Subsurface manure application generally does not appear to increase leaching potential when manure is applied at recommended rates. Plant utilization of conserved N was shown in only a portion of the published studies, indicating that further work is needed to better synchronize manure N availability and crop uptake. At this time in the United States, the economic and environmental benefits from reducing losses of N as NH_3 are expected to outweigh potential liability from increases in denitrification with subsurface manure application. To fully evaluate the trade-offs among manure application methods, a detailed environmental and agricultural economic assessment is needed to estimate the true costs of potential increases in N_2O emissions with manure injection.

J. Environ. Qual. 40:352–361 (2011)
doi:10.2134/jeq2010.0069
Posted online 4 Oct. 2010.
Received 16 Feb. 2010.
*Corresponding author (curtis.dell@ars.usda.gov).
© ASA, CSSA, SSSA
5585 Guilford Rd., Madison, WI 53711 USA

Incorporating manure with tillage shortly after application is known to greatly reduce nitrogen (N) losses due to ammonia (NH_3) volatilization (Sommer and Hutchings, 2001; Thompson and Meisinger, 2002; Webb and Misselbrook, 2004). However, manure commonly remains on the soil surface in no-till crop production or grass forage systems. Older manure injection equipment, often using chisels or tines, can provide the benefits of soil incorporation (Klausner and Guest, 1981; Schmitt et al., 1995; Sawyer et al., 1991; Sutton et al., 1982; Mattila, 1998) but often with more soil disturbance and burial of surface residues than is acceptable for soil erosion control (Hanna et al., 2000). Alternative "low disturbance" manure injection systems, such as shallow disk injectors, are now available that minimize soil disturbance and burial of surface residues and are compatible with no-till and forage systems.

The primary benefit of subsurface manure application, with respect to the N balance, is reduced NH_3 emissions. From an agronomic standpoint, lower NH_3 losses means the conservation of plant-available N. Reducing NH_3 emissions can also benefit air and water quality. A primary air quality concern that can be addressed by reducing NH_3 emissions is the formation of airborne particulate matter and associated effects on human health (Pinder et al., 2007). Decreased NH_3 emissions would also reduce the quantity of atmospheric N that is redeposited into natural ecosystems, such as forests, where it contributes to soil acidification and species shifts (Bobbink et al., 2010), and to water bodies where it contributes to eutrophication (Paerl, 1997). In the case of the Chesapeake Bay, it is estimated that about 6% of the N inputs to the watershed are derived from atmospheric deposition from agricultural sources (Chesapeake Bay Program, 2009).

Subsurface manure applications are theoretically susceptible to greater denitrification losses due to the close juxtaposition of the available C that drives microbial activity with nitrate (NO_3^-) resulting from nitrification of manure ammonium (NH_4^+) (Wulf et al., 2002b). Subsurface applications may also increase leaching losses due to possible channeling of preferential flow through the zone of high NO_3^- produced from nitrification and the mineralization of organic-N within the manure band (Shipitalo and Gibbs, 2000). These potentially higher losses to denitrification

C.J. Dell, USDA–ARS, Pasture Systems and Watershed Management Research Unit, University Park, PA; J.J. Meisinger, USDA–ARS, Environmental Management and By-Products Utilization Lab., Beltsville, MD; D.B. Beegle, Crop and Soil Sciences Dep., Pennsylvania State Univ., University Park, PA. Assigned to Associate Editor Phil Haygarth.

Abbreviations: DCD, dicyandiamide; DMPP, dimethylpyrazole phosphate; GHG, greenhouse gas; TN, total nitrogen.

and leaching could offset the reductions in NH_3 volatilization. Denitrification and nitrification are also of interest due to their contribution to nitrous oxide (N_2O) emissions, which is a potent greenhouse gas (GHG) that is roughly 300 times more effective than carbon dioxide (CO_2) at absorbing radiation in the atmosphere (IPCC, 2007). Therefore, it is important to develop a more complete understanding of the effects of subsurface manure applications on the fate of manure N. Traditional N balance calculations offer excellent opportunities for developing this holistic understanding.

The overall goal of this paper is to review existing research documenting the impacts of subsurface manure application methods on the total-N balance to evaluate the tradeoffs between N conservation through reduced NH_3 losses and potential increases in N losses from other pathways. However, no studies of surface vs. subsurface manure application include truly complete N balances. This situation results from the complex effects that manure has on the whole soil–crop N cycle, including NH_3 volatilization, denitrification, mineralization-immobilization, leaching, and crop N uptake. Indeed, estimating the N balance for a manured system is a formidable task. Accordingly, we have summarized the manure placement studies from the perspective of their effects on several N cycle processes and have outlined the study that has provided the most comprehensive N balance. Our focus is on the injection of liquid cattle and swine slurries because technology for their subsurface application is commercially available in the United States and data are available on the performance of those technologies. Although equipment for subsurface placement of poultry litters and other "dry" manures is emerging, actual data on their impacts on the fate of manure N are in the initial phases of field and laboratory research.

Ammonia Emission Reductions

There is very good evidence to show that substantial reductions in NH_3 losses can be expected with subsurface manure application compared with surface broadcasting. These reductions are achieved by limiting the manure surface area that is exposed to the air and by increasing immobilization of NH_4^+ because of greater contact of manure with soil. Although the extent of NH_3 reduction varies among studies, values from 40 to over 90% have commonly been reported with knife, chisel, or disk injectors when compared with unincorporated surface applications (Table 1). The greatest reductions in emissions have usually been observed with closed slot injectors, as contrasted with nonclosed injectors. A recent review of NH_3 emission abatement in the United Kingdom (Webb et al., 2009) reported an average of about 60% reduction in NH_3 emissions with both open and closed slot injection in grassland and an average 81% reduction with closed slot injection on arable land. Webb et al. (2009) reported no detectable decrease in emissions with open-slot injectors on arable land in the United Kingdom; however, only two studies were cited, and heavy rainfall occurred shortly after manure application. Therefore, the rainwater may have transferred manure NH_4^+ into the soil where it was not susceptible to volatile loss.

Many factors, such as applicator design, soil moisture, and manure dry matter content and application rate, can influence the emission of NH_3 by controlling the placement and distribution of soil-applied manure. In general, NH_3 emissions are expected to increase in proportion with the amount of soil surface area covered by manure. Hansen et al. (2003) found a linear relationship between the volume of the slot created by the applicator and the potential to reduce NH_3 emissions, indicating that sufficient slot volume is needed to ensure that manure does not remain on the soil surface. It follows that

Table 1. A review of ammonia emission reductions with manure injection (disk, knife, or chisel) relative to surface application without incorporation.

Reference	Country	Depth	Manure type	Time of application	Crop	NH_3 reduction†	NH_3 as % of TAN‡
		cm				%	
			Open slot injection				
Dosch and Gutser (1996)	Germany	NR§	beef	fall	SG¶	91	NR
Hansen et al. (2003)	Denmark	NR	beef	NR	grass	20–50	10–40
Lambert and Bork (2003)	Canada (AB)	10	swine	NR	grass	40–55	NR
Misselbrook et al. (1996)	England	6	beef	spring-fall	grass	37–80	21–35
Morken and Sakshaug (1998)	Norway	5–10	dairy	spring	grass	53–62	NR
Smith et al. (2000)	England	5	dairy/beef	winter	grass	53–81	NR
Smith et al. (2000)	England	5	dairy/beef	summer	grass	9–70	NR
Smith et al. (2000)	England	5	dairy/beef	fall-spring	SG	0–100	NR
Wulf et al. (2002a)	Germany	NR	dairy	NR	grass/SG	67	10
			Closed slot injection				
Dell et al. (unpublished)	US (PA)	5–6	dairy/swine	spring	maize	58 to ~100	0.5–35
Moseley et al. (1998)	England	5–6	swine	spring	SG	68–85	4–15
Rodhe et al. (2006)	Sweden	NA	beef	summer	grass	~100	~0
Thompson et al. (1987)	England	35	dairy	winter and summer	grass	94–97	1–2
Weslien et al. (1998)	Sweden	6	swine	spring	SG	95	1

† Reduction in NH_3 emission with injection relative to surface application of manure without tillage.

‡ Total ammoniacal-N in manure.

§ Not reported.

¶ Small grain.

application rates cannot exceed available slot volume if effective containment of applied manure is to be maintained. Open and closed slot applicators can perform similarly in grasslands (Rodhe et al., 2006; Webb et al., 2009), but open slot application may not always reduce NH_3 emissions in cropland soils (Webb et al., 2009). Soil moisture and manure dry matter content also appears to be an important factor influencing the ability of subsurface applicators to reduce NH_3 emissions. With wet soils, furrow closure can be poor and manure infiltration decreases (Sommer and Ersbøll, 1994), leading to some surface exposure of manure. Increasing dry matter content has also been shown to decrease the rate of manure infiltration into soil (Petersen et al., 2003)

Reductions in NH_3 emissions can also be achieved with surface banding and trailing shoe applications of slurries. For example, Misselbrook et al. (2002) found that mean NH_3 emissions from seven sites in England with trailing shoe and banded application were 57 and 26% lower than with broadcast application, respectively. Although these application methods merit consideration for greater use in the United States, there has been little adoption of these methods, and no NH_3 emission data have been reported for North America.

Aerators, vertical tillage implements, and other high-residue tillage tools have also been proposed for minimally disruptive incorporation of manures (Bittman et al., 2005) (see commercial advertisements such as: www.no-tillfarmer.com/pages/News-Smart-Till-20-Foot-Model-Released.phpinsert reference and http://www.yetterco.com/PressReleases/2007-08-21-vtaforjd726.html). However, these tools do not directly inject manure into soil but rather can enhance infiltration of broadcast or banded manures. Therefore, larger amounts of manure are likely to remain on the soil surface compared with injection, leading to highly variable reductions in NH_3 emissions. Little information is available on the impacts of these tools on NH_3 volatilization. Bittman et al. (2005) reported that banding dairy manure over aeration holes reduced NH_3 emissions by about 50% compared with broadcast application, but Gordon et al. (2000) and Myers (2010) saw no significant reduction with similar application methods.

Although the specific effect of subsurface applications can vary with type of injector, depth of injection, and soil moisture, the literature cited above shows that injection should reduce NH_3 loss by 40 to over 90% compared with unincorporated surface applications of liquid manure. Moreover, reduction in NH_3 emission with injection is nearly immediate, unlike incorporation by tillage, which requires a separate field operation. Fifty percent or more of the NH_3 emissions can be expected within 24 h of broadcast application (Thompson et al., 1987), so incorporation by a separate tillage operation (if possible in the system) must be done quickly after application to achieve a significant reduction.

Nitrogen Losses Due to Denitrification

Subsurface manure applications can lead to greater denitrification and subsequent N losses as dinitrogen gas (N_2) or N_2O. Large additions of readily metabolized carbon compounds may result in high rates of microbial activity within the injection slots, depleting oxygen and creating anaerobic conditions needed for denitrification (Flessa and Beese, 2000; Wulf et al., 2002b). The potential for losses due to denitrification may also increase if the N conserved by reducing NH_3 volatilization accumulates as NO_3^- before it is used by plants.

A limited amount of information has been published concerning the impacts of subsurface manure application methods on denitrification activity and N_2O emissions. Thompson et al. (1987) estimated that total N losses from denitrification were 76 and 293% greater when dairy slurry was injected (closed slot), compared with surface broadcast application, during winter and spring applications, respectively. Although reducing NH_3 emissions with injection conserved 50 to 75 kg N ha^{-1} in the Thompson et al. (1987) study, the conservation was partially offset by the additional N lost through denitrification. These authors also reported the addition of a nitrification inhibitor (nitrapyrin) to injected manure resulted in denitrification losses that were comparable to those from surface-applied manure in their winter experiment; however, nitrapyrin addition only reduced loss by about 20% in the spring experiments.

Dosch and Gutser (1996) found that slurry injection resulted in a 62% increase in denitrification losses ($N_2O + N_2$), despite reducing NH_3 emissions by 91% compared with surface application. In their case, the reduction in losses due to NH_3 volatilization was 20 kg N ha^{-1}, but there was an additional denitrification loss of 3 kg N ha^{-1}. Misselbrook et al. (1996) observed denitrification losses ranging from 0.3 to 8.1 kg N ha^{-1} with shallow injection of cattle slurry but saw little difference between injection and surface application. Neither Velthof et al. (1997) nor Vallejo et al. (2005) found a significant effect of manure injection in grassland on total denitrification losses.

Impacts on Leaching

Leaching N losses can be highly site specific because they are greatly influenced by precipitation, soil properties, topography, and land management. Therefore, only general trends in the impacts of manure injection on leaching are discussed. However, placement of concentrated bands of manure belowground could increase N leaching if manure bands intercept preferential flow pathways. There are only a few reports of N leaching in conjunction with subsurface manure application. Approximately fourfold greater leaching to a 60-cm depth was observed with shallow disk injection, compared with surface broadcast and tillage incorporation, in the first year of a 4-yr lysimeter study in Pennsylvania (Dell et al., unpublished). However, leaching losses were similar for injection, surface application, and tillage incorporation in the following 3 yr. The initially higher leaching below shallow disk injection in Pennsylvania may have been exaggerated because of the preferential flow of banded manure through soil voids that were left after the lysimeters were installed. Because soil structure was re-established with time, the large voids likely closed, and therefore the preferential flow diminished. Autumn injection of digested sewage sludge to a loamy sand soil in England resulted in a 32 to 500% more NO_3^- leaching to a 1-m depth than when the same material was surface applied (Shepherd, 1996). In addition to the high leaching potential of the loamy sand soil, large NH_4^+–N applications with the digested sludge

(60–300 kg ha^{-1}), and the subsequent NO_3^- formation, may have exacerbated leaching losses in the Shepherd (1996) study. Ball-Coelho et al. (2006) found increased leaching to tile drains with swine manure injection at high application rates, but they saw no impact of application method when manure N additions did not exceed crop N requirement. Weslien et al. (1998) found that NO_3^- leaching to tile drainage (~90 cm deep) was not affected by shallow, closed trench injection of swine slurry. Overall leaching losses to a 90-cm depth reported by Thompson et al. (1987) were low because N application rates were not excessive, and they found no significant effects of manure injection on leaching losses.

Although the reported research is very limited, available data suggest that in most cases manure injection is not likely to substantially increase leaching losses if manure is applied at rates consistent with crop needs. One hydrologic condition that could contribute to greater leaching from injected manure is injection over shallow tile drains or shallow water tables. However, additional research is required to fully understand how these conditions affect N leaching after subsurface manure applications.

Crop Response

In a 4-yr study at two sites in the Netherlands, Groot et al. (2007) found that shallow manure injection resulted in substantially greater N recovery by the grass crop than with surface application (42 vs. 26% of the total manure N). However, only a portion of the studies listed in Table 1 reported impacts of subsurface manure application on crop yield or crop N utilization. Thompson et al. (1987) reported that 40 to 50% of the N conserved through reducing NH_3 emissions was accounted for by increased N recovery in the crop. However, Dell et al. (unpublished) and Smith et al. (2000) saw no significant effect of manure application method on crop yields, whereas Rodhe et al. (2006) reported reduced forage yield due to damage to grass caused by the injectors. Despite a substantial reduction in NH_3 emissions, Misselbrook et al. (1996) found no increase in crop N uptake. They suspected that the lack of crop response may have been related to injector design and poor slurry distribution.

Developments in injector design could help to increase crop utilization of injected N by improving the placement of the manure and facilitating greater crop uptake. Schmitt et al. (1995) found that vertical knife and horizontal sweep injection of liquid dairy and swine manure resulted in significantly higher yields than surface broadcast manure. They also report that the horizontal sweep injection resulted in higher yields than the traditional vertical knife injection.

The findings cited above point out the inconsistent nature of crop responses to subsurface applications; however, crop response can also be affected by a range of additional site-specific environmental conditions (e.g., rainfall, temperature, and soil properties) and agronomic factors (e.g., crop varieties, weed pressures, and planting dates). With reductions in NH_3 emissions in many studies on the order of 20 to 30 kg N ha^{-1}, the increase in plant-available N may not be large enough to overcome the influence of the other yield-limiting factors. Additionally, the labor requirement and expense of gas emission and leaching measurements has often limited the amount of replication and size of experimental units in the reported studies. To fully evaluate crop response to manure injection technologies, field-scale manure application and yield evaluation would be beneficial.

Evaluating the Nitrogen Balance

Estimating the N balance for a complex N source like manure is challenging because manure interacts with many pathways in the soil–crop N cycle. Site-specific conditions and inherent measurement errors in the assessment of individual N fates also contribute variability among locations and to cumulative uncertainties in the estimation of the total N balance for a system. Given the ranges of losses and accumulation of manure N in the measured pools reported in the literature cited above, the range of estimates of the N losses to the various pathways for injected manure slurries is quite wide (Fig. 1).

Although several studies have addressed the impacts of manure injection on certain N fates, no reports of complete N balances are available in the literature. The most comprehensive N balance comparing manure injection with unincorporated surface manure application was reported by Thompson et al. (1987), who applied dairy slurry (~250 kg total N ha^{-1}) to perennial ryegrass (*Lolium perenne* L) on a well drained soil in Hurly, England. The study was replicated in early December and late April. Ammonia loss was monitored for 17 d after application, denitrification was estimated to 15 cm deep using acetylene inhibition and intact soil cores every 1 or 2 wk over 5-mo (winter application) or 2-mo periods (spring application), soil NO_3^-–N and NH_4^+–N were determined in

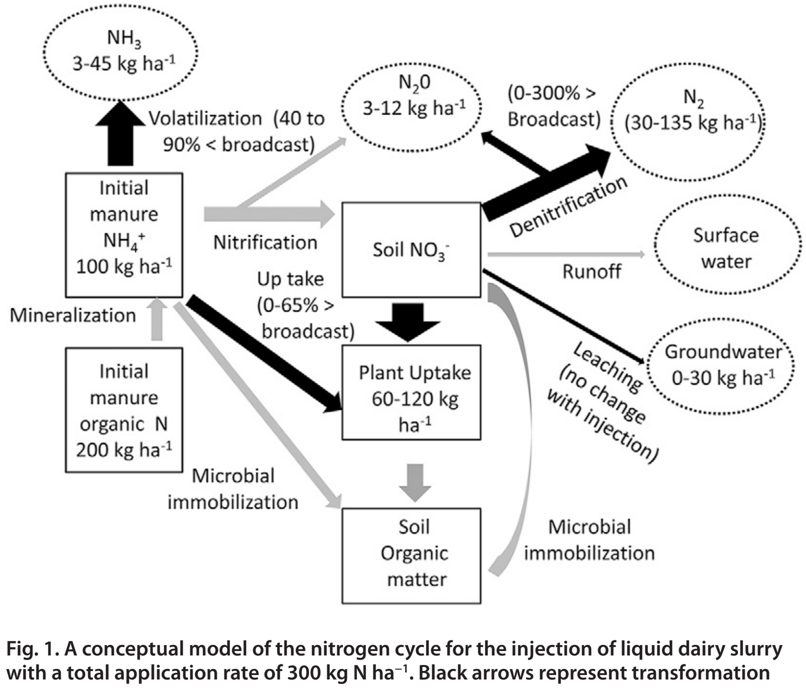

Fig. 1. A conceptual model of the nitrogen cycle for the injection of liquid dairy slurry with a total application rate of 300 kg N ha^{-1}. Black arrows represent transformation where data exist to estimate the impact of injection. Given data presented in the text, estimates of total losses from all pathways would range from about 36 to 222 kg N ha^{-1}.

soil cores each time denitrification measurements were made, NO_3^- leaching was estimated to 1.8 m deep after the growing season by measurement of NO_3^-–N and NH_4^+–N in 30-cm increments of soil cores, and ryegrass N uptake was determined from dry matter yield and N concentration at each of three forage harvests. Nitrogen not accounted for by measurement of the listed pools was assumed to be immobilized within the soil organic matter or roots.

The final N balance for the winter-applied slurry in Thompson et al. (1987) shows that NH_3–N losses from injection were 1%, compared with 31% of total N (TN) lost from surface application (Fig. 2a). However, the gaseous losses to denitrification were 21% of TN with injection, compared with 12% with surface application. Total gaseous losses (NH_3 plus denitrification) from the surface application were nearly double the losses from injection (43 vs. 22% of TN, respectively). The lower gaseous losses and greater conservation of plant-available N with injection were also expressed in greater crop N response, with ryegrass recovering an estimated 33% of the injected slurry N, compared with 20% from surface applications. The soil mineral-N profiles beneath manured plots were not significantly different from the control plots, leading to the conclusion that leaching losses were negligible in their grassland system. A supplementary treatment of injected slurry with nitrification inhibitor (nitrapyrin) was also studied (Fig. 2a), which produced a N balance of 1% for NH_3 loss, 9% denitrified, and 36% in crop uptake, leaving 54% for other N sinks.

In their spring study, Thompson et al. (1987) show that NH_3 losses were again higher with surface applications, amounting to 20% of the slurry TN or 48% of the slurry NH_4–N (Fig. 2b). The denitrification losses were much smaller after spring application (7% of TN for injection and 2% for surface application). These results were attributed to the lower soil moisture conditions and somewhat higher crop N utilization of the spring-applied slurry, as shown by crop N recoveries of 36% of TN for injection and 26% of TN for surface applications. The addition of nitrapyrin to the injected slurry increased crop N recoveries to 42% of TN but produced similar NH_3 losses (1%) and denitrification losses (5%) compared with the untreated injected slurry.

The dynamics of denitrification losses from injected manure appear to create wide variation in estimates of N balances among research studies. Other publications have reported lower denitrification from injection than measured by Thompson et al. (1987). This variability in denitrification losses among studies is likely due to different site factors that affect the underlying conditions needed to trigger denitrification events (e.g., soil moisture or texture, soil drainage, available C, and soil NO_3^-). In a study also conducted at Hurly, England, Thompson and Pain (1989) found low denitrification losses on a poorly drained soil where low soil NO_3^- levels resulted when poor aeration limited the nitrification of slurry NH_4^+–N. Misselbrook et al. (1996) observed a smaller impact of manure injection on denitrification rates than did Thompson et al. (1987) possibly because shallower injection and open injection slots resulted in larger NH_3–N losses in their study and, subsequently, a smaller accumulation of soil NO_3^- that could be denitrified. Additionally, Misselbrook et al. (1996) suggest that Thompson et al. (1987) may have overestimated overall denitrification losses from the field by measuring only direct losses from the injection slot without consideration for the areas between slots. To test the impact of available C on denitrification losses, Thompson (1989) injected dairy slurry with two levels of available C to the same soils at Hurly, England that were used for the Thompson et al. (1987) study. The low-C slurry was created by diluting raw slurry (1:1) and adding NH_4Cl to maintain the initial NH_4^+ concentration. Only 8% of the manure NH_4^+–N was lost to denitrification in the 3 mo after application when low-C slurry was applied, but 48% of the initial NH_4^+–N was lost from undiluted slurry.

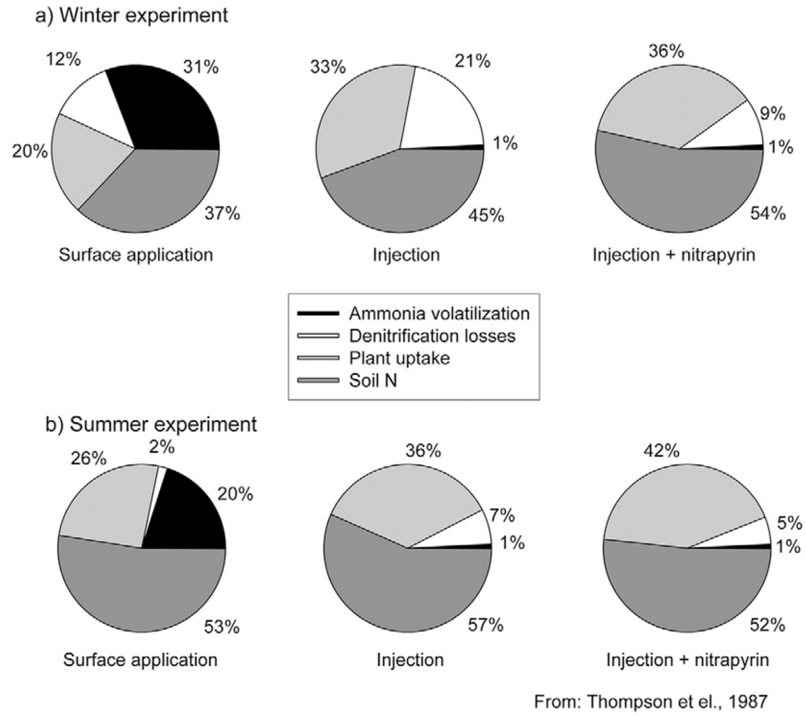

Fig. 2. Nitrogen budget for surface-applied and injected dairy slurry presented by Thompson et al. (1987). Leaching losses were negligible for each application treatment.

Nitrogen Dynamics within the Manure Band

The alteration of the microenvironment in and around the manure injection band and the subsequent effect on N cycling and fate has been studied to a limited extent. Comfort et al. (1988) reported changes in water content, soluble C, and inorganic N with time after dairy manure injection in the soil within grid points surrounding dairy manure injection slots. They observed that water content of the injection zone remained higher than the surrounding soil throughout a 99-d period after injection (Fig. 3a). Soluble C within the injection zone was initially threefold greater than the surrounding soil (750 vs. 190 mg kg^{-1}) but declined rapidly to 210 mg kg^{-1} soil by 26 d after injection (Fig. 3b). Rapid mineralization of organic N and nitrification were observed in the injection zone.

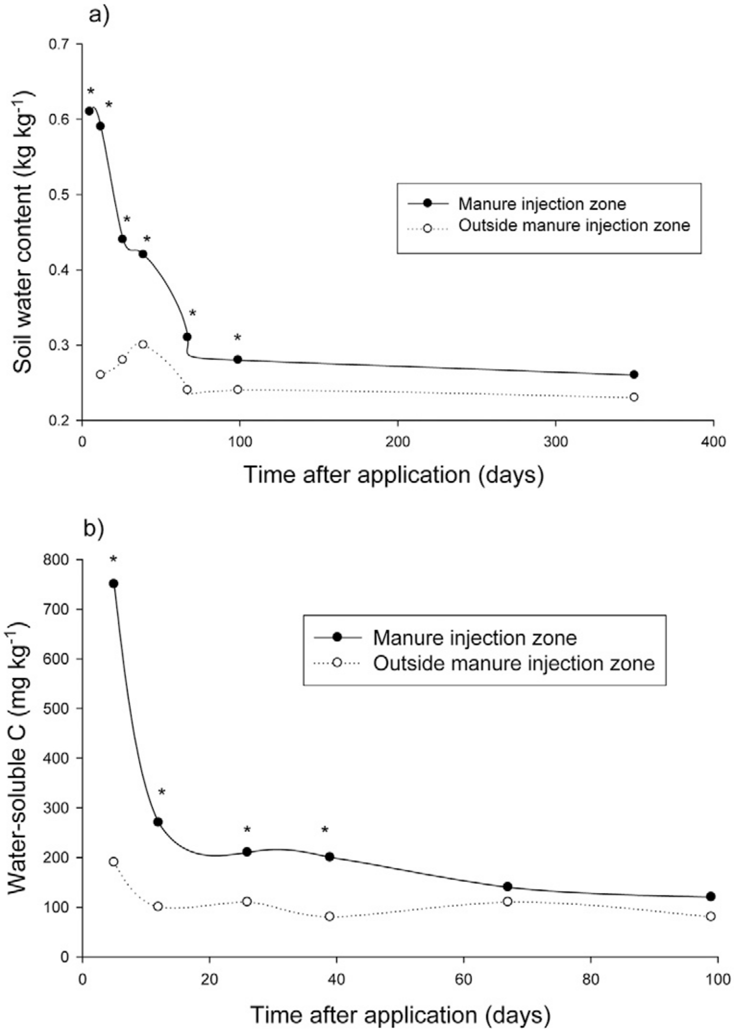

Fig. 3. Changes in (a) soil water content and (b) water-soluble carbon with time after the injection of dairy manure within and outside the injection zone. Outside samples were obtained at least 15 cm from the injection zone. *Values from within and outside the injection zone were significantly different (*P* < 0.05). (Adapted from Comfort et al., 1988.)

Initially, NH_4^+ was high (220–320 mg kg^{-1}) within the manure injection slot, whereas NO_3^- was low (generally <60 mg kg^{-1}) (Fig. 4). During the first 26 d after injection, NH_4^+ diffused away from the injection slot then diminished in the slot and surrounding soil. Conversely, NO_3^- concentrations increased greatly with concentrations in the slot, ranging from 320 to 780 mg kg^{-1} soil by 26 d after injection. Although denitrification rates were not directly measured in the Comfort et al. (1988) study, increased soil water content and the consumption of oxygen due to the high rates of microbial metabolism of soluble C compounds and nitrification produced conditions favorable to rapid denitrification. This hypothesis was further supported by a comparison of soil redox potentials within a manure injection slot and unamended soil presented by Flessa and Beese (2000). They observed highly reducing conditions (Eh ≈ −200 mV) within the injection slot for about 1 d after slurry application, followed by a rapid rise to positive Eh values that leveled out at Eh values of 350 to 400 by 5 d after application. Redox potentials within the injection slot remained between 350 and 400 mV for the remaining 5 d of measurement. Conversely, unamended soil remained well aerated, with redox potentials constantly between 600 and 700 mV.

In a subsequent study, Comfort et al. (1990) measured N_2O emissions directly from a simulated manure injection zone (with and without acetylene additions) and correlated those emissions with soil water content and carbon dioxide (CO_2) concentration of the soil atmosphere. The addition of acetylene blocked further reduction of N_2O to N_2 and inhibited N_2O production from nitrification, allowing the measurement of total denitrification as well as separate N_2O production measurement. The greatest gaseous N losses occurred 5 d after manure was injected and corresponded to the maximum CO_2 concentration in soil. Nitrous oxide was the primary gas emitted in the first 1 to 2 d after injection, with a shift to a greater proportion of N_2 thereafter. Comfort et al. (1990) also concluded that the availability of C was the main factor controlling denitrification rates within the injection zone and that the supply of readily metabolized C can be expected to be consumed within a narrow time frame (10–15 d at 12°C in their study).

Nitrogen transformations within the manure injection slot and dispersion of N into surrounding soil were studied by Chadwick et al. (2001) using ^{15}N tracers. A ^{15}N-labeled manure NH_4^+ pool was created by adding labeled urea to swine slurry and incubating to allowed urease enzymes to decompose the urea to form NH_4^+. The labeled manure was then placed into 5-cm-deep slots cut into 20-cm diameter intact soil cores or on the soil surface. Soil was sampled 8 d after manure application from injection slots, directly below the slots, and from the soil surrounding the injection slot at two depths (0–6 and 6–12 cm). During the 8-d period, labeled N moved to all portions of the cores, and a substantial portion of the ^{15}N was recovered as NO_3^- (~17% of total applied ^{15}N). The portion of added NH_4^+ that was nitrified and the movement of NO_3^- into the 6- to 12-cm depth were similar for injected and surface-applied N. However, the swine slurry used by Chadwick et al. (2001) had a low dry matter content (~3%), which may have facilitated greater infiltration of surface-applied manure than would be expected for manure with greater solids.

Petersen et al. (2003) further investigated slurry distribution with two injection methods (disk and harrow tine) and concluded that the relationship between slurry organic matter content and soil water potential can help describe that distribution. The measurement of water potential gradients and the movement of ^{13}C, ^{15}N, and Br^- showed greater retention of cattle manure in the injection slot compared with swine manure, likely due to the greater solids content of the cattle manure. More pronounced gradients in labeled C and N were also seen with disk injection compared with harrow tine injection. Although Petersen et al. (2003) did not measure N transformation rates, they hypothesized that differences in manure distribution with manure solids content and injection methods would affect N turnover rates because of the stimulation of nitrifying and denitrifying organisms at the manure–soil interface (Petersen et al., 1992; Frostegård et al., 1997).

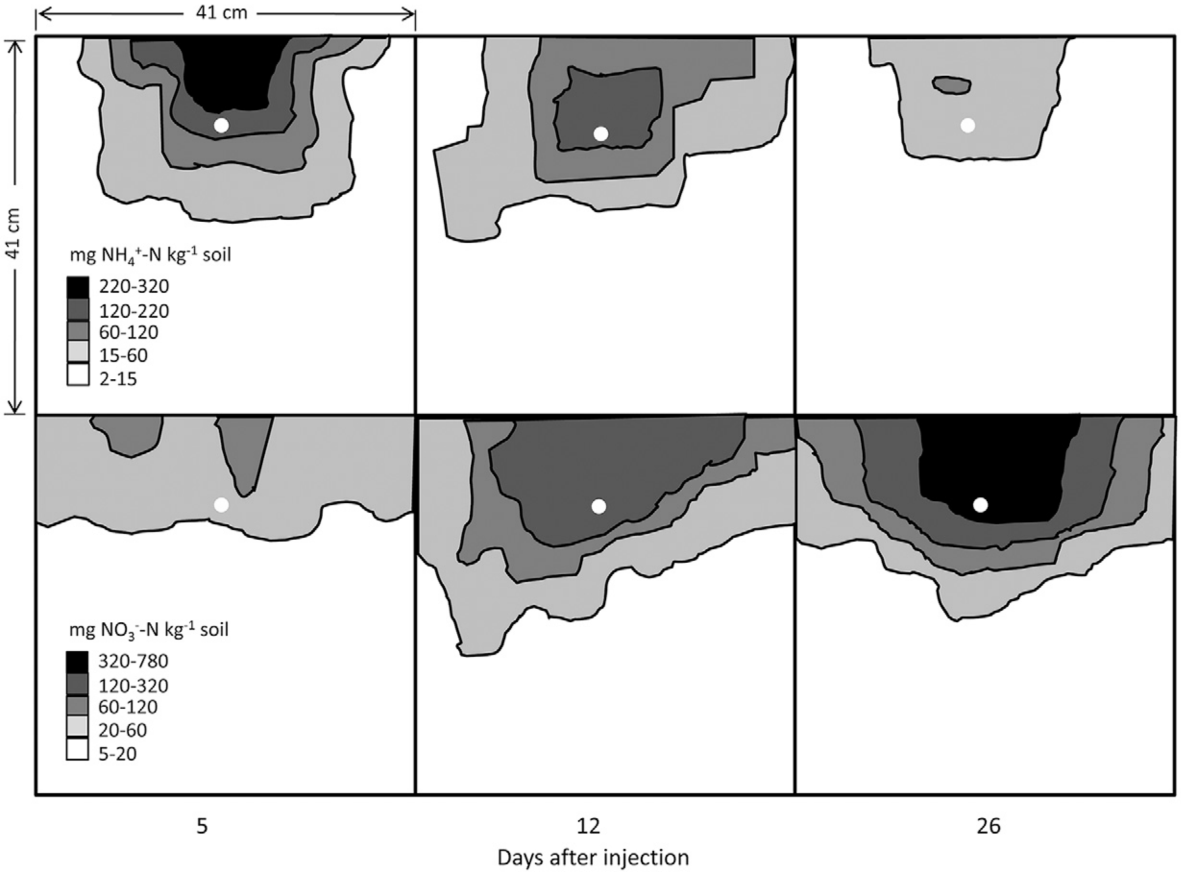

Fig. 4. Ammonium and nitrate concentration gradients surrounding manure injection zones 5, 12, and 26 d after injection. Circle indicates the location of the bottom center of the injection slot. (Adapted from Comfort et al., 1988.)

Implications for Greenhouse Gas Emissions

In addition to the loss of plant-available N, one must also consider the impact of subsurface applications on GHG production due to the potential for increased N_2O loss from additional nitrification and denitrification that are possible with injected manures. Although high rates of nitrification have been shown in response to manure application, there is no information in the literature to show an impact of manure injection on nitrification rates per se, although the NH_3 conserved by injection would contribute substrate for increasing the total quantity of N being nitrified. Other factors that might increase GHG production are the increased soil water content (Comfort et al., 1988) and lower redox potential (Flessa and Beese, 2000) within the manure injection band, which might increase methane (CH_4) gas emissions.

Relatively few studies have reported field measurements of the impacts of subsurface manure application on N_2O emissions, and reports of CH_4 measurement in the field are even more limited. Rodhe et al. (2006) reported 275% greater N_2O emission with injected cattle manure compared with surface banding, but overall N_2O emissions in their study were low (1.1 and 0.3% of total manure N for injected and band applied, respectively). Methane emissions in their study were not greatly affected by application method and were limited to the first 2 d after manure application, after which the soil remained a sink for CH_4. Injection of cofermentation product (70% dairy slurry, 30% organic household waste) in Germany resulted in two- to threefold greater N_2O, compared with splash plate application, in cropland and grasslands (Wulf et al., 2002b). Although N_2O emissions were not measured separately by Thompson et al. (1987) and Dosch and Gutser (1996), greater N_2O emissions were also likely associated with the significantly greater total denitrification activity observed with manure injection in those studies. Conversely, other field studies (Clemens et al., 1997; Vallejo et al., 2005; Velthof et al., 1997) found that manure injection did not significantly affect N_2O emissions.

Flessa and Beese (2000) suspected that high spatial and temporal variability in N_2O and CH_4 emissions may prevent detection of differences in gas emissions in the field. In response, they conducted continuous gas emission analysis (CO_2, N_2O, and CH_4) after surface application and injection of dairy slurry to mesocosms that were maintained under carefully controlled and constant conditions (67% water-filled-pore space and 14°C). They found that cumulative N_2O emissions during the 9-wk experiment were more than 10-fold greater with injected manure compared with surface application (0.2 vs. 3.3% of the total manure N). Injecting manure also resulted in greater CH_4 fluxes. Surface application and injection caused initial emissions of CH_4 before soils returned to being CH_4 sinks, but injection resulted in a small net emission (1 g ha^{-2}) during the 9-wk experiment, whereas surface application was a net sink (–34 g ha^{-2}). Carbon dioxide emissions were not affected by manure application method. Injection affected N_2O and CH_4 emissions much more strongly in the Flessa and Beese (2000) study than in previously reported studies. They attributed that difference to the constant high water content maintained in their mesocoms. Although mesocosm water content may have exceeded values commonly observed in the field, Flessa and Beese's (2000) findings illustrate the high potential for N_2O and, possibly, CH_4 emissions if manure injection is associated with wet or poorly aerated soils.

Further environmental and agricultural economic analysis of potentially greater N_2O emissions with manure injection is needed to assess the full environmental cost and the potential economic liability to farmers in the United States. Because quantities of N lost as N_2O are generally a small portion of the total applied N pool and do not constitute a large economic loss for the farmer, understanding the full cost of greater N_2O emissions lies primarily in the potential impact on global GHG budgets and the potential for global climate change. Nitrous oxide emissions are typically <10% of total denitrification products (Velthof et al., 2009), and the Intergovernmental Panel on Climate Change (IPCC, 2007) estimates that, on average, 1.25% of the TN in applied fertilizer or manure is emitted as N_2O. Therefore, a typical dairy slurry with 4 g total N L^{-1} surface applied at a rate of 75,000 L ha^{-1} would be expected to lead to emissions of 3.75 kg N_2O–N ha^{-1}. With reported increases in N_2O emissions of up to 300%, we could anticipate that injecting manure would lead to emissions of up to 12 kg N_2O–N ha^{-1}(~19 kg N_2O ha^{-1}). The fertilizer value of the N lost as N_2O would still likely be less than the value of the N conserved by reducing NH_3 emissions. However, the economic liability to farmers associated with N_2O emission could become much greater if GHG caps are adopted in the United States. With the current regulatory structure, GHG offset prices in the United States are low. In January 2010, the Chicago Climate Exchange (www.chicagoclimatex.com) paid \$0.10 per metric ton of CO_2 traded. Assuming 298 CO_2 equivalents per unit of N_2O, the cost to offset the N_2O emission calculated above would be up to \$0.60 ha^{-1} with the observed maximum emission rate for subsurface manure application. In Europe, the January 2010 price for GHG offsets was approximately \$24 per metric ton of CO_2 (www.ecx.eu). At the European rates, offsetting the same N_2O emissions would cost up to \$120 ha^{-1}. Although this is a rather simplistic estimation, it illustrates that the assessment of trade-offs among manure injection technologies is highly dependent on regulations in a country or region and how the liability associated with GHG emissions is valued.

Potential for Nitrification Inhibitors

The use of nitrification inhibitors (e.g., nitrapyrin, dicyandiamide [DCD], and 4- dimethylpyrazole phosphate [DMPP]) with injected manure has the potential to decrease denitrification losses and N_2O emissions by causing a disconnect between the NO_3^- production and the peak metabolism of readily metabolized manure C. Most readily available manure C can metabolize within 2 wk after application (Comfort et al., 1990); therefore, delaying nitrification until after that time can reduce the quantity of NO_3^- that is present during the period when high rates of microbial metabolism of manure C can lead to conditions that favor denitrification. In addition to the substantial reduction in denitrification losses with nitrification inhibitors discussed previously (Thompson et al., 1987; Thompson, 1989), Comfort et al. (1990) reported that nitrapyrin cut nitrification activity within a manure injection zone approximately in half during a 40-d period after application. However, they did not see significant reductions in denitrification in all replications of their experiments. de Klein et al. (1996) observed that the addition of DCD nearly eliminated additional denitrification losses resulting from cattle slurry injection. Adding DCD with injected manure resulted in 30% greater forage yield and 156% greater apparent N recovery in 1 yr of a 2-yr study conducted by Misselbrook et al. (1996) (no increase seen in the first year of the study). Vallejo et al. (2005) found that DCD addition to swine manure effectively inhibited nitrification for 20 to 30 d and reduced N_2O emissions by approximately 46% for injection and surface application. When DMPP was applied at a rate of 2 kg ha^{-1} with injected cattle slurry, Dittert et al. (2001) observed a 32% reduction in N_2O emission compared with injections without the nitrification inhibitor. Additionally, the use of ^{15}N labeling of the manure showed that emissions and the activity of the inhibitor were localized within the injection slot. The cited reductions in denitrification losses or increases in plant N utilization are an indication that the use of nitrification inhibitors should be studied further to determine if they are a cost-effective means to reduce N_2O emission from injected manures.

Research Needs

The biggest challenge to N research with subsurface manure application is understanding and managing the soil–manure transformations to improve crop N use efficiency. Improved N use efficiency for manure requires a holistic management approach that involves improving the N rate recommendations and allowing for all N sources (legume residues, soil organic N, etc.), the timing of manure applications in relation to the hydrologic cycle, and the placement of manure to reduce N losses through NH_3 volatilization, denitrification, and leaching. Improving N use efficiency will reduce accumulation of excess soil NO_3^-–N that can potentially be denitrified or leached. The lack of a crop response in many cases could be the result of N applications beyond the crop's requirement, a disconnect in timing between the availability of conserved N and plant demand, or poor placement or distribution of manure that limits plant N utilization. Improved crop N efficiencies from manure will likely be achieved through a combination of refined application rate guidelines, improved timing of applications to better synchronize N availability with plant N demand, and improved injector designs that optimize manure placement and distribution.

Complete comprehensive studies measuring the total N balance after subsurface manure applications would provide valuable information to scientists and nutrient managers. A full analysis of the N balance with subsurface manure application should address NH_3 volatilization, total denitrification losses, N_2O emissions, plant N uptake, N leaching, and N loss with surface runoff and eroded soil. These all-inclusive N balances remain a challenge for future agricultural research. Accounting for each of these N fates will be an expensive and time-consuming task; however, it is needed to facilitate a complete agronomic, environmental, and economic assessment of the value of subsurface manure application. Greater utilization of ^{15}N tracers could aid in the determination of complete N balances for injected manure. Although the expense and the quantities of labeled manure needed for field studies may limit ^{15}N use to small plot, laboratory, or greenhouse experiments, the ability to differentiate and track manure-derived N provided by ^{15}N labeling makes it a valuable tool for refining mechanisms of N transformations.

A better understanding of how subsurface application affects overall denitrification rates and the ratio of N_2O to N_2 is also needed. To achieve this, additional physical measurements are needed to better explain manure distribution and its effect on water content and oxygen diffusion within manure bands. This additional knowledge should facilitate the development of mechanistic models to estimate denitrification activity and N_2O emissions under a range of soil and management conditions.

Conclusions

Research has shown that subsurface manure applications with disks, knifes, or chisels can be expected to reduce NH_3 emissions, compared with traditional surface applications, by at least 40%. With well designed manure injectors and closure devises for the injection channel, NH_3 emissions can be reduced by over 90%. However, a portion of that N conservation can be expected to be offset by greater, although variable, denitrification loss. The potential also exists for greater leaching losses with injection than with surface manure application, in particular with shallow tile drainage, high water table conditions, or manure application in excess of plant N requirement. The limited data available on the trade-off between quantities of N conserved by reducing NH_3 emissions and the potential for greater NO_3^-–N losses from denitrification or leaching for injected manure suggests that the gains will be greater than the losses (Fig. 1). However, the quantities and costs of N_2O emissions associated with increased denitrification with subsurface application are not clearly established. Under the current regulatory scenario in the United States, the environmental and economic benefits derived from reducing NH_3 emissions appears to outweigh the economic liability to the farmer of potentially greater denitrification losses and corresponding higher N_2O emissions. A re-evaluation of the full environmental cost of N_2O emission and the establishment of GHG emission caps in the United States could greatly increase the liability to the farmer for N_2O emissions. Economic analysis (environmental and agricultural) is needed to better assess the cost of the trade-off between reduction in NH_3 emission and potentially greater N_2O emissions. Nitrification inhibitors have been shown to reduce denitrification activity and could be an important method to reduce N_2O emissions after manure injection, indicating the need for further research into the extent of their benefit.

Continued research to improve crop N use efficiency, by accounting for conservation of manure NH_3–N with appropriate adjustments in N management from other sources, should lower the levels of soil NO_3^-–N and reduce the risks for N loss through denitrification and leaching. Although the data are limited, it appears that subsurface application should not lead to additional leaching risks compared with incorporating manure with tillage. As with all types of manure application, special consideration should be given to shallow tile drainage and high water tables. Avoiding manure application at rates above crop N requirements should be a cornerstone of N management for maintaining crop yields and reducing N losses to water and air resources.

References

Ball-Coelho, B.R., R.C. Roy, and A.J. Bruin. 2006. Nitrogen recovery and partitioning with different rates and methods of sidedressed manure. Soil Sci. Soc. Am. J. 70:464–473.

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.

Bobbink, R., K. Hicks, J. Galloway, T. Spranger, R. Alkemade, M. Ashmore, M. Bustamante, S. Cinderby, E. Davidson, F. Dentener, B. Emmett, J.W. Erisman, M. Fenn, F. Gilliam, A. Nordin, L. Pardo, and W. Devries. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: A synthesis. Ecol. Appl. 20:30–59.

Clemens, J., R. Vandre, M. Kaupenjohann, and H. Goldbach. 1997. Ammonia and nitrous oxide emissions in landspreading of slurry as influenced by application technique and dry matter reduction: 2. Short-term nitrous oxide emissions. Z. Pflanzenernaehr Bodenkd. 160:491–496.

Chadwick, D.R., J. Martinez, C. Marol, and F. Beline. 2001. Nitrogen transformations and ammonia loss following injection and surface application of pig slurry: A laboratory experiment using slurry labeled with 15N-ammonium. J. Agric. Sci. 136:231–240.

Chesapeake Bay Program. 2009. Sources of nitrogen loads to the Bay. Available at: www.chesapeakebay.net/status_nitrogensources.aspx?menuitem=19797 (verified 18 Sept. 2010).

Comfort, S.D., K.A. Kelling, D.R. Keeney, and J.C. Converse. 1988. The fate of nitrogen from injected manure in a silt loam soil. J. Environ. Qual. 17:317–322.

Comfort, S.D., K.A. Kelling, D.R. Keeney, and J.C. Converse. 1990. Nitrous oxide production from injected liquid dairy manure. Soil Sci. Soc. Am. J. 54:421–427.

de Klein, C.A.M., R.S.P. van Logtestijn, H.G. van de Meer, and J.H. Geurink. 1996. Nitrogen losses due to denitrification from cattle slurry injected into grassland with and without nitrification inhibitor. Plant Soil 183:161–170.

Dittert, K., R. Bol, R. King, D. Chadwick, and D. Hatch. 2001. Use of a novel inhibitor to reduce nitrous oxide emission from 15N-labelled dairy slurry injected into soil. Rapid Commun. Mass Spectrom. 15:1291–1296.

Dosch, P., and R. Gutser. 1996. Reducing N losses (NH_3, N_2O, N_2) and immobilization from slurry through optimized application techniques. Fert. Res. 43:165–171.

Flessa, H., and F. Beese. 2000. Laboratory estimates of trace gas emissions following surface application and injection of cattle slurry. J. Environ. Qual. 267:117–127.

Frostegård, Å., S.O. Petersen, E. Bååth, and T.H. Nielsen. 1997. Dynamics of a microbial community associated with manure hot-spots as revealed by phospholipids fatty acid analysis. Appl. Environ. Microbiol. 63:2224–2231.

Groot, J.C.J., J.D. Van der Ploeg, F.P.M. Verhoeven, and E.A. Lantinga. 2007. Interpretation of results from on-farm experiments: Manure-nitrogen recovery on grassland as affected by manure quality and application technique. 1. An agronomic analysis. Wageningen J. Life Sci. 54:235–254.

Gordon, R., G. Patterson, T. Harz, V. Rodd, and J. MacLeod. 2000. Soil aeration for dairy manure spreading on forage: Effect on ammonia volatilization and yield. Can. J. Soil Sci. 80:319–326.

Hanna, H.M., D.S. Bund, J.C. Lorimor, S.K. Mickelson, S.W. Melvin, and D.C. Erbach. 2000. Manure incorporation equipment effects on odor, residue cover, and crop yield. Appl. Eng. Agric. 16:621–627.

Hansen, M., S.G. Sommer, and N.P. Madsen. 2003. Reduction of ammonia emission by shallow slurry injection: Injection efficiency and additional energy demand. J. Environ. Qual. 32:1099–1104.

IPCC. 2007. Climate change 2007. Synthesis report. Intergovernmental Panel on Climate Change. Available at www.ipcc.ch/publications_and_data/publications_ipcc_fourth_assessment_report_synthesis_report.htm (verified 18 Sept. 2010).

Klausner, S.D., and R.W. Guest. 1981. Influence of NH_3 conservation from dairy manure on the yield of corn. Agron. J. 73:720–723.

Lambert, B.D., and E.W. Bork. 2003. Ammonia volatilization trends following liquid hog manure application to forage land. J. Soil Water Conserv. 58:207–214.

Mattila, P.K. 1998. Ammonia volatilization from cattle slurry applied to grassland as affected by slurry treatment and application technique-first year results. Nutr. Cycling Agroecosyst. 51:47–50.

Misselbrook, T.H., J.A. Laws, and B.F. Pain. 1996. Surface application and shallow injection of cattle slurry on grassland: Nitrogen losses, herbage yield and nitrogen recoveries. Grass Forage Sci. 51:270–277.

Misselbrook, T.H., K.A. Smith, R.A. Johnson, and B.F. Pain. 2002. Slurry application techniques to reduce ammonia emissions: Results of some UK field-scale experiments. Biosystems Eng. 81:313–321.

Moseley, P.J., T.H. Misselbrook, B.F. Pain, R. Earl, and R.J. Godwin. 1998. The effect of injector tine design on odour and ammonia emissions following injection of bio-solids into arable cropping. J. Agric. Eng. Res. 71:385–394.

Morken, J., and S. Sakshaug. 1998. Direct ground injection of livestock waste slurry to avoid ammonia emission. Nutr. Cycling Agroecosyst. 51:59–63.

Myers, T.L. 2010. Master's thesis. Pennsylvania State Univ., University Park, PA.

Paerl, H.W. 1997. Coastal eutrophication and harmful algal blooms: Importance of atmospheric deposition and groundwater as new nitrogen and other nutrient sources. Limnol. Oceanogr. 42:1154–1165.

Petersen, S.O., A.L. Nielsen, K. Haarder, and K. Henriksen. 1992. Factors controlling nitrification and denitrification: A laboratory study with gel-stabilized liquid cattle manure. Microb. Ecol. 23:239–255.

Petersen, S.O., H.H. Nissen, I. Lund, and P. Ambus. 2003. Redistribution of slurry components as influenced by injection method, soil, and slurry properties. J. Environ. Qual. 32:2399–2409.

Pinder, R.W., P.J. Adams, and S.N. Pandis. 2007. Ammonia emission controls as a cost-effective strategy for reducing atmospheric particulate matter in the eastern United States. Environ. Sci. Technol. 41:380–386.

Rodhe, L., M. Pell, and S. Yamulki. 2006. Nitrous oxide, methane and ammonia emissions following slurry spreading on grassland. Soil Use Manage. 22:229–237.

Sawyer, J.E., M.A. Schmitt, R.G. Hoeft, J.C. Siemens, and D.H. Vanderholm. 1991. Corn production associated with liquid beef manure application methods. J. Prod. Agric. 4:335–344.

Schmitt, M.A., S.D. Evans, and G.W. Randall. 1995. Effect of liquid manure application methods on soil nitrogen and corn grain yields. J. Prod. Agric. 8:186–189.

Shepherd, M.A. 1996. Factors affecting nitrate leaching from sewage sludges applied to a sandy soil in arable agriculture. Agric. Ecosyst. Environ. 58:171–185.

Shipitalo, M.J., and F. Gibbs. 2000. Potential of earthworm burrows to transmit injected animal wastes to tile drains. Soil Sci. Soc. Am. J. 64:2103–2109.

Smith, K.A., D.R. Jackson, T.H. Misselbrook, B.F. Pain, and R.A. Johnson. 2000. Reduction of ammonia emission by slurry application techniques. J. Agric. Eng. Res. 77:277–287.

Sommer, S.G., and A.K. Ersbøll. 1994. Soil tillage effects on ammonia volatilization from surface-applied or injected animal slurry. J. Environ. Qual. 23:493–498.

Sommer, S.G., and N.J. Hutchings. 2001. Ammonia emission from field applied manure and its reduction-invited paper. Eur. J. Agron. 15:1–15.

Sutton, A.L., D.W. Nelson, J.D. Hoff, and V.B. Mayrose. 1982. Effects of injection and surface applications of liquid swine manure on corn yield and soil composition. J. Environ. Qual. 11:468–472.

Thompson, R.B. 1989. Denitrification in slurry-treated soil: Occurance at low temperatures, relationship with soil nitrate and reduction by nitrification inhibitors. Soil Biol. Biochem. 21:875–882.

Thompson, R.B., and J.J. Meisinger. 2002. Management factors affecting ammonia volatilization from land-applied cattle slurry in the Mid-Atlantic USA. J. Environ. Qual. 31:1329–1338.

Thompson, R.B., and B.F. Pain. 1989. Denitrification from cattle slurry applied to grassland. p. 247–260. *In* A.A. Hansen and K. Henrik (ed.) Nitrogen in organic wastes applied to soils. Academic Press, New York.

Thompson, R.B., J.C. Ryden, and D.R. Lockyer. 1987. Fate of nitrogen in cattle slurry following surface application or injection to grassland. J. Soil Sci. 38:689–700.

Vallejo, A., L. Garcia-Torres, J.A. Diez, A. Arce, and S. Lopez-Fernandez. 2005. Comparison of N losses (NO_3–, N_2O, NO) from surface applied, injected or amended (DCD) pig slurry of an irrigated soil in a Mediterranean climate. Plant Soil 272:313–325.

Velthof, G.L., O. Oenema, R. Postma, and M.L. Van Beusichem. 1997. Effects of type and amount of applied nitrogen fertilizer on nitrous oxide fluxes from intensively managed grassland. Nutr. Cycling Agroecosyst. 46:257–267.

Velthof, G.L., D. Oudendag, H.R. Witzke, W.A.H. Asman, Z. Klimont, and O. Oenema. 2009. Integrated assessment of nitrogen losses from agriculture in EU-27 using MITERRA-EUROPE. J. Environ. Qual. 38:402–417.

Webb, J., B. Eurich-Menden, U. Dämmgen, and F. Agostini. 2009. Review of published studies estimating the abatement efficacy of reduced-emission slurry spreading techniques. *In* M. Sutton, S. Reis, and S.M.H. Baker (ed.) Atmospheric ammonia. Springer Science + Business Media B.V., Dordrecht, the Netherlands.

Webb, J., and T.H. Misselbrook. 2004. A mass-flow model of ammonia emissions from U.K. livestock production. Atmos. Environ. 38:2163–2176.

Weslien, P., L. Klemedtsson, L. Svensson, B. Galle, Å. Kasimir-Klemedtsson, and A. Gustafsson. 1998. Nitrogen losses following application of pig slurry to arable land. Soil Use Manage. 14:200–208.

Wulf, S., M. Maeting, and J. Clemens. 2002a. Application technique and slurry co-fermentation effects on ammonia, nitrous oxide, and methane emissions after spreading: I. Ammonia volatilization. J. Environ. Qual. 31:1789–1794.

Wulf, S., M. Maeting, and J. Clemens. 2002b. Application technique and slurry co-fermentation effects on ammonia, nitrous oxide, and methane emissions after spreading: II. Greenhouse gas emissions. J. Environ. Qual. 31:1795–1801.