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Nitrogen Management of Forages in Relation to Gaseous Emissions – New Approaches and Considerations

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ABSTRACT

Forages have a high N demand, a long growing season, and an effective root system all contributing to effective nutrient capture. However forages are restrictive in methods available for mitigating gaseous losses both as NH_3 and as N_2O, due to both practical and cost considerations. Strategies are needed to address the challenges of both N efficiency and N losses. Agronomic techniques in long term experiments can enhance estimates of N loss pathways and N efficiency, and demonstrate the importance of integrated multi-nutrient approaches. The dual manure stream concept divides manure into a thin fraction suitable as an N source for grass and a sludge fraction suitable as a P source for corn. While this represents an integrated approach, questions remain about alternate loss pathways. While grazing greatly reduces ammonia emissions it is not clear that grazing improves N use relative to confinement systems. The current levels of prospective mitigation of emissions are perhaps modest. However new approaches such as acidifying manure, novel nitrification inhibitor products, more durable legume stands, ongoing improvements in manure application methodology with increasing adoption by farmers, and novel integrated approaches will continue to make incremental improvements in reducing losses of nitrogenous gases and other reactive N species and improving nutrient efficiency of forages.

Key words: Ammonia, Forages, Gaseous emission, Nitrogen efficiency, Nitrous oxide, Reactive N.

Introduction

Gaseous emissions are important pathways for loss of nitrogen (N) from agriculture and contribute to environmental degradation. In Canada, emission of ammonia (NH_3) to the atmosphere is the greatest loss pathway for N from agricultural systems, exceeding both nitrate (NO_3^-) leaching and denitrification (Clair e*t al.*, 2014). Much of the NH_3 from livestock production is emitted from land applied manure (37%), often spread on forage land, and urine deposition on pastures (7%) (Sheppard and Bittman, 2013). While total loss of N as nitrous oxide (N_2O) is much smaller than NH_3, there is international concern about the contribution of N_2O from agriculture to global climate change. Managing N inputs in crops, including forages, must address the dual goals of increasing nitrogen use efficiency (crop biomass or crop N uptake per applied N) and reducing losses of N to the environment where a single atom can have multiple deleterious effects (referred to as the N Cascade, Galloway *et al.*, 2003). The two goals are inextricably connected: inefficient N use on farms or fields will lead inevitably to environmental contamination, and practices that contribute to N losses will invariably reduce efficiency of nitrogen use.

There are three main sources of N in forage fields: manure, fertilizer and biological N fixation, although atmospheric deposition and irrigation water can be important in some locations, e.g. deposition contributes 49 kg N ha^{-1} on dairy farms in the Netherlands (Aarts *et al.*, 2000). All inputs of reactive N contribute to N_2O emissions from forage fields, while applications of manure and fertilizer contribute also to emissions of NH_3, as does direct emissions of NH_3 from plants. Moreover, losses of N both as gaseous NH_3 and leached nitrate lead to secondary N_2O emissions (emissions due to farm N that occur off the farm), which should be considered when assessing field practices.

There have been several detailed reviews published in recent years on losses of NH_3 and N_2O from production of agricultural crops, including forages (referred to below); here we will describe some new technologies and offer reflections on mitigating gaseous N losses associated with forage production in Canada and elsewhere.

On issues of measurements

There are generally accepted factors for gaseous emissions and abatement measures for NH_3 and N_2O from the Intergovernmental Panel on Climate Change (IPCC) and United Nations Economic Commission for Europe (UNECE), however, it must be recognized that there remain significant concerns about the methodology used to obtain the data used in developing these factors. For N_2O, it is recognized that static chambers impose an artificial environment and the measurement areas are quite small which leads to spatial variability. Also, measurements tend to be taken infrequently, and for short periods, which risks missing emission events and introduces errors when integrating emissions over time. And many studies are carried out for less than a full year, missing critical winter emissions (Holst *et al.*, 2008), or under non steady-state field conditions. Micrometeorological methods such as relaxed eddy accumulation (REA) and, more recently, the inverse dispersion method (backward Lagrangian stochastic or bLs), do not impose artificial conditions but they are costly, require a lot of space and are usually carried out for short periods of time (Denmead, 2008). At present 95% of published studies are with the static chamber method (Rochette 2011). It is important to note that N_2O flux values are sometimes used to estimate N_2 emissions from fields, especially for large upscaling studies (Drury *et al.*, 2007), since there are no fully accepted methods for measuring N_2 emissions that are widely used. In these cases, estimates of N_2O will have a significant bearing on N budgets.

There are some of the same questions raised with respect to NH_3 emission measurements, many of which are taken in wind tunnels, as opposed to static chambers, because of the importance of air movement. But conditions inside wind tunnels are certainly artificial (note that natural rainfall is often excluded), and it has recently been argued that use of chambers may lead to overestimates of emissions relative to less artificial micrometeorological measurements (Sintermann *et al.*, 2012). Such overestimates could lead to erroneous national emission inventories which are important to national governments due to the National Emission Ceilings regulations in Europe and the Gothenburg Protocol of the Convention on Long-range Transboundary Air Pollution (CLRTAP) in the UNECE. While NH_3 emission inventories tend to track well temporal changes in atmospheric concentrations (Bittman *et al.*, 2015), there are often differences between measured atmospheric NH_3 concentrations and predicted concentrations based on emission inventories; the differences are referred to as 'NH_3 gaps' and the gaps may be in part due to errors in emission measurements (Erisman *et al.*, 2009).

Inferring grassland N losses from agronomic data

Losses of N from grasslands can be calculated as the difference between N inputs (manure, fertilizers, etc.) and N outputs (mainly harvested crop), provided that there is no change in soil N. Change in soil N is difficult to ascertain in typical field studies lasting often no more than three years, but can be estimated in long-term studies where N applications are repeated consistently. For example, in a 7-year trial on a perennial sward of tall fescue we determined that of the 400 kg N ha^{-1} applied annually as dairy slurry about 32% accumulated in the soil (Bittman *et al.*, 2007). More recently we estimated that over 16 years, 13% of N applied in dairy slurry was retained in the soil (Bittman *et al.*, 2014). In this study, we estimated that apparent N recovery by the crop was 46% of applied N. Thus, including soil changes, 59% of the 400 kg N ha^{-1} applied as manure was accounted for while 41% was presumed lost to the environment. Apparent N recovery from manure is calculated as N uptake on the treated plots minus the N uptake on unfertilized (control) plots, to account for contribution of N from the soil. However, because there was no net release of soil N in this study, it may be more appropriate to use unadjusted values for N recovery, or simply crop N uptake values, to estimate losses. The 'unadjusted' loss rate from the applied manure was only 17%, although it may be about 20-23% if N deposition (estimated as 20 kg N ha^{-1}) were taken into account. This is somewhat greater than expected loss (15-20%) by NH_3 volatilization from banded cattle slurry based on 30-40% emission factor for ammonium N for banded dairy slurry (there were over 60 applications of manure containing about 50% ammonium N in our study) (Bittman *et al.*, 2007; Webb *et al.,* 2010; Thorman *et al.*, 2008). This suggests that NH_3 volatilization was the main loss pathway from dairy slurry in this study. Although the rate of N loss from ammonium nitrate fertilizer (applied at the same N rate) was similar to the slurry (20-25%), only a small proportion of the loss can be attributed to volatilization, since the emission factor for this fertilizer formulation is typically about 4% (Harrison and Webb 2001). Therefore, for ammonium nitrate, the loss was via a combination of leaching and denitrification. Since denitrification from fertilizer is generally not higher than from manure (Paul and Zebarth 1997) and denitrification from manure was determined to be quite low, it can be assumed that most of the loss from fertilizer was via leaching. The conclusion that there is relatively greater leaching losses from mineral fertilizer than manure is supported by lysimeters measurements (unpublished data) and the low rate of denitrification is consistent with the relatively coarse-textured soil and low levels of available soil nitrate in these highly productive grass plots. We expect that most leaching occurred slowly over fall and winter, and when there was intense rain after fertilizer application.

It is interesting to note that, in this trial, N recovery rates for perennial ryegrass were about 9% lower than for tall fescue, and this suggests a higher rate of leaching since there would be little difference in gaseous emissions between the grasses. This illustrates the importance of matching N application rates to the N requirements of crops, which remains a difficult challenge, since lacking reliable long term weather forecasts we cannot predict either crop growth or soil N mineralization. Higher manure applications (800 kg N ha^{-1}) in our study resulted in 30-35% (unadjusted) N loss from tall fescue plots, with this additional loss likely due to both more emissions and more leaching. Another interesting inference can be made from the treatment that combined 200 kg N ha^{-1} as mineral fertilizer with 400 kg N ha^{-1} as manure; compared to manure alone, there was an increase in crop growth and N uptake but reduced soil N accumulation so that there was an overall greater N loss than from manure alone, likely by leaching.

Mitigating ammonia losses in grassland

In the long term manure study mentioned above, a trailing shoe applicator was used to improve the efficacy of the applied nutrients and to reduce NH_3 emissions compared to broadcast applications (Webb *et al.*, 2010). We had earlier investigated the effect of the trailing shoe relative to conventional broadcast applications in terms of short term crop N recovery in three seasons: spring, summer, and fall (Bittman *et al.*, 1999). We found that in 4 of 9 trials the sliding shoe significantly improved apparent N uptake over broadcasting whereas method of application did not have an effect in 5 of 9 trials. The yield and N uptake values from the sliding shoe were consistently close to ammonium nitrate fertilizer at equivalent rates of mineral N. Hence, our N fertilizer replacement value from banded manure, based on total applied N, was about 50% which is similar to short-term shallow injection values reported by Schroder *et al.* (2007). Much lower fertilizer replacement values for banded manure were recently reported by Lalor *et al.* (2011) but this can be attributed to slurries with much higher solids concentration (over 20% compared to 5-6%; see below). Overall, there was about a 38% increase in apparent N recovery with the trailing shoe, but perhaps an equally important practical outcome was that the sliding shoe provided consistent response to manure N, which cattle producers require if they are to depend on manure as their primary N source for their crops. We attribute the greater and more consistent agronomic response by grass to generally lower and less variable rates of NH_3 emission with banded than broadcast manure.

The effectiveness of various slurry applicator technologies for reducing emission from grassland has been extensively reviewed in recent years (Webb *et al.*, 2010; Webb *et al.*, 2014; Webb *et al.*, 2015; Maguire *et al.*, 2011). It is generally accepted that of the applicators available for perennial forages, emission reductions relative to broadcasting are least for trailing hoses (30-35%), intermediate for trailing shoes (30-60%) and greatest for shallow injection (70-80%) (Webb *et al.*, 2014). The efficacy of the banding methods varies with the height or development of the canopy at the time of application (Sommer *et al.*, 1997; Thorman *et al.*, 2008). It is interesting that we did not find increased N recovery by delaying manure application for 7-10 days, allowing for a greater canopy, as might be expected with reduced NH_3 loss (Bittman *et al.*, 1999). Instead, delayed application of manure withheld N from the crop initially slowed growth and resulted in lower yield, higher N concentrations and no difference in N uptake. So, if delaying manure application had also reduced NH_3 emissions (this was not measured), it could have resulted in more leaching.

Aggressive mitigation methods like injection are often problematic for perennial forages because they may limit application volumes, injure crops, or be difficult to implement in stony or hard soils and they may have unintended effects like creating runoff channels on sloped land. Tools are gradually being developed to overcome these limitations for grassland, like the tubulator in Sweden (Rodhe and Etana, 2005), pressurized injection in Finland (Morken and Sakshaug, 1998) and the Aerway SSD in North America (Bittman *et al.*, 2005).

The costs associated with NH_3 mitigation methods tend to be somewhat greater than the value of the conserved N, generally -0.5 to 1.5 Euros per kg of NH_3 conserved, or perhaps less if the implement can be locally fabricated (Webb *et al.*, 2014; Webb *et al.*, 2015). Investing in new application equipment is more cost effective for manure contractors or for very large livestock operations than for typical farms. When considering the cost-benefit of these technologies, it is useful to consider co-benefits relative to conventional broadcasting: more consistent grass response, more uniform nutrient application, less risk of drift to sensitive areas, more time for spreading manure without contaminating the canopy, reduced odour, and greater palatability for grazing.

In lieu of these methods, or in combination with them, there are other approaches for abating emission from grassland. Because NH_3 emissions are greatest in warm, windy, dry weather it may be possible to reduce emission by avoiding spreading of manure, especially by broadcasting, during these conditions, a technique now referred to as the Application Timing Management System (ATMS) described by Webb *et al.* (2014). While appealing because there is no cost for additional equipment, there are risks of unintended consequences such as increased runoff, leaching and N_2O emissions due to greater rainfall. Also, with multiple grass harvests there may not be sufficient N for the crop during the warmer months, although the longevity of available N in manure may be extended, and losses abated, by using (still quite costly) nitrification inhibitors, notably DCD (dicyandiamide), and more recently DMPP (3,4-dimethylpyrazole phosphate) and nitrapyrin (2-chloro-6-(trichloromethyl) pyridine). So while these products do not reduce NH_3 emissions from manure they might have an indirect role in NH_3 abatement. Nitrification inhibitors have been more extensively considered for abating emissions of N_2O (see below).

Where practical, NH_3 emission during warm dry periods can be reduced by irrigating after manure application (Webb *et al.*, 2014). Emissions may also be reduced by diluting manure as there appears to be linear relationships between slurry DM concentration and emissions; diluting slurry by 50% will on average reduce emissions by 30%. However, hauling additional water to the field in tanks can be costly and time consuming, although less so if transported through pipes (irrigation or umbilical systems). It has long been known that acidification reduces NH_3 emissions from manure but recent technical advances in Denmark have made slurry acidification more practical, if still quite expensive (Webb *et al.*, 2015). Due to stringent emission limits in Denmark, acidification of slurry is now widely practiced and, of special interest to forage producers, this method can replace mandatory injection.

Removing solids from slurry manure by filtration or other means (centrifugation, digestion) will have the effect of reducing manure viscosity which can facilitate infiltration into the soil, without the need for additional hauling. Separated liquid manure has lower NH_3 emission and higher N uptake than whole slurry (Bhandral *et al.*, 2009). However, processing can also increase emissions by raising manure pH. If the processed manure is more dilute, higher application rates will be required which will delay full infiltration, especially on wet soils. Finally, there needs to be a way for effectively utilizing the remaining high-solids fraction without generating additional emissions (Webb *et al.*, 2013).

There are comparatively few options for mitigating emissions from solid manure on established grassland. Prior to forage establishment, NH_3 can be conserved by manure incorporation, however application rates must be kept low to match low N uptake by the seedlings. There is perhaps less concern about emissions from solid cattle manure (also called farmyard manure or FYM) because it may be depleted of ammonium (Qian and Schoenau, 2002) due to high emission rates in feedlots (Flesch *et al.*, 2007). As the ammonium content of these manures is very low, there is comparatively little environmental or commercial benefit in adopting measures to mitigate losses. Here the mitigation effort should be further up the manure-handling stream. The economic benefit of NH_3 abatement is also less clear when manure is applied on stands composed largely of legumes, such as alfalfa (Lucerne), which is a major forage crop on many dairy and some beef operations across North America. Since added N will not contribute to increased crop or crude protein production in alfalfa swards, the manure is valued much less for its N than its P, K and S contents, and abating NH_3 loss will be more difficult to justify to farmers. In contrast to solid cattle manure, solid poultry litter is very high in ammoniacal N. While there are few options for mitigating NH_3 emission from poultry litter applied to perennial forages, a recent technology for injecting poultry litter holds promise (Pote *et al.*, 2011). Also, it is possible to moderately curtail emissions by applying poultry manure under a tall (20 cm) grass canopy (Bittman *et al.*, 2008).

There is considerable concern also about NH_3 emissions from mineral fertilizers on grassland, particularly where urea-based products are replacing ammonium nitrate because of lower costs and safety concerns. The best strategy may be to apply the fertilizer in cool damp weather (see ATMS above) and to perhaps use products that contain urease and nitrification inhibitors or coatings to prolong the effectiveness of the N (unpublished data). Urease inhibitors may reduce sharp rises in pH that would enhance emissions. At present these types of products are not widely used for perennial forages due to cost, although there is some penetration in the grass turf sector. Unlike manure, banding of urea fertilizer may lead to high NH_3 emissions due to a rise in pH associated with hydrolysis (Rochette *et al.*, 2009).

An integrated approach for abating ammonia and utilizing nutrients: the dual manure stream concept

It is evident that where manure is applied as the main source of N, even with low emission technology, there will likely be an over application of other nutrients, especially P, leading to soil accumulation (Bittman *et al.*, 2007 and 2011). Accumulation of soil P can be reduced by supplementing or replacing manure with commercial N fertilizer, as this lowers manure doses and enhances crop yield and P recovery. We observed in our long-term study that adding 200 kg of N ha^{-1} as commercial fertilizer to 400 kg N ha^{-1} of dairy slurry increased yield and recovery of both N and P hence reducing P loading (Bittman *et al.*, 2007). However, this approach is clearly not suitable for farms with surplus N.

For these farms, an alternative approach for avoiding excessive soil P from slurry manure is to remove some of the P-rich solids before application. Gravitational settling of slurry (<7% solids) stored in tanks or multi-stage lagoon systems will, over time, reduce solid particles and lower P concentrations in the supernatant (Burton, 2007). We compared grass response to long term applications of whole vs. separated slurry from commercial farms; the liquid fraction came from a two stage lagoon system (Bittman *et al.*, 2011). The whole and separated fraction had N:P ratios of 5.2:1 and 9.7:1, mineral N fractions of 0.51 and 0.65, and solids contents of 5.9 and 1.6%, respectively. We attributed higher apparent N recovery by grass receiving the thin supernatant fraction than whole slurry to a higher percentage of available N. Significantly higher (8.8 to 16.1%) apparent N recovery per applied mineral N from the thin fraction suggests lower NH_3 emissions due to more rapid manure infiltration into the soil compared with whole slurry. Also, there was much higher P loading with whole slurry than with the separated fraction (81 and 47 kg P ha^{-1} at the 400 kg total N rate); crop P uptake from the separated fraction was nearly sustainable.

The sludge accumulating at the bottom of manure tanks over several months of storage was relatively high in P and successfully used to replace side-banded 'starter' commercial fertilizer for corn (Bittman *et al.*, 2012). This was done by first injecting the sludge at corn-row spacing (75 cm), then precision-planting the corn < 10 cm from the manure furrows a few days later. The sludge provided all the required P and some of the required N, and because it was injected, there was likely little NH_3 emitted from this fraction. As the sludge remained in the tank until use, additional NH_3 emissions during storage from this fraction is not expected. The effect of this practice on leaching and N_2O emissions needs to be assessed.

Nitrous oxide emissions from grassland

It is well known that emission of N_2O from a soil is regulated by soil moisture (especially water filled pore space), soil temperature, and soil mineral N, especially NO_3^-. While it has been proposed that the amount and duration of soil nitrate largely determines N_2O emissions (Burton *et al.*, 2008), it is our observation that larger emission spikes occur only when all three factors favour emissions (unpublished data). Except soon after application of fertilizer or manure, perennial forage soils tend to have relatively low soil nitrate concentrations because of the presence of abundant roots and soil microbes, the latter promoted by labile C originating from root exudates, soil organic matter and manure. Microbes compete strongly with grass for mineral N, more so for ammonium than for nitrate, thus removing ammonium from the mineral N pool (Jackson *et al.*, 1989). Therefore, N_2O emissions from unfertilized forages tend to be low; in our moist maritime environment annual year-round emission from unfertilized grass plots averaged 0.24 kg N_2O-N ha^{-1} over 5 years. This is important because forages are often grown with little fertilization; we estimated that the large beef cattle sector in Canada applies commercial N fertilizer on just 19% of their perennial forage land, with an average N rate on the fertilized land of 50 kg ha^{-1}, while another 19% of land receives manure (Sheppard *et al.*, 2015). The relatively small proportion of forage land receiving N is due to low cattle densities and extensive use of legumes, especially alfalfa, in forage mixes; beef farmers in Canada reported that on average legumes comprise about 40% of their forages. It is now accepted that N_2O emissions from legumes are not directly associated with the *de novo* production of reactive N but rather the turnover of organic residues and roots (Rochette and Janzen, 2005) resulting in relatively low emission rates (0.4 kg N_2O-N ha^{-1} yr^{-1}) for mixed grass legume swards. This emission value is only slightly higher than for unfertilized grass (above). Legumes are being better maintained in mixed stands due to improvements in genetics and in-stand seeding (sod-seeding) (Cuomo *et al.*, 2001; McLeod *et al.*, 2009).

We studied the effects of long term applications of whole and separated liquid slurry fraction and commercial fertilizer on N_2O emission (unpublished data). As mentioned above, manure separation improved grass production and reduced P loading. The emission factor (year-long) was highest for commercial fertilizer (0.84%) which is sometimes reported in C-rich forage systems (e.g. unsaturated soils in Luo *et al.*, 2010). Emission factor was lowest for the low rate of whole slurry (0.38%) but similar for high rate of whole slurry and both high and low rates of separated slurry (0.64%). Note that whereas NH_3 emissions from products are typically compared on total ammoniacal N, N_2O emissions are based on total N, although sometimes also on expected available N (Ball *et al.*, 2004). In our study, there was no difference in N_2O emissions per kg of yield or N uptake between separated and whole manure, which indicated that the long term gain in efficiency and sustainability from the liquid fraction did not come at a cost of greater N_2O. There were emission surges that followed each of the four equal manure doses each year but these spikes were much lower in cut 1 (18% of annual) than cut 2 (25-28%) and cut 3 (30-32%) probably due to the warmer soil temperatures during cuts 2 and 3. It may be possible to somewhat mitigate N_2O emissions by reducing manure application during the warmer part of the year but this may reduce grass productivity in midseason.

All emission rates for manure in our long-term trial were lower than IPCC values of 2% and this may be due to the fairly coarse textured soil, a productive grass crop, and relatively low soil NO_3^- concentrations. It may be due also to the negative correlation between soil temperature and moisture so that there are few days when there are high levels of both parameters (Bhandral *et al.*, 2008). There may also have been unmeasured leaching of N_2O during high rainfall periods in winter, although such losses are probably fairly small given that only small quantities of NO_3^- were released as determined with anion exchange resin membranes. Wintertime emissions accounted for 20, 25 and 30% of annual emissions for the separated fraction, whole slurry and control, respectively. It is important to measure N_2O in winter as well as summer to properly quantify annual emissions.

In determining overall emission factors for non-permanent perennial grass stands, N_2O emitted during stand renovation must also be considered (Luo *et al.*, 2010). We found that there was very little emission from renovation of unfertilized grass. Emissions were 2-3 times higher from historical manure than from fertilizer at equivalent N rates and emissions were greatly reduced when the stand was terminated by tillage compared with a herbicide. This was somewhat unexpected because tillage released more soil NO_3^- than herbicide, but the tillage also reduced soil bulk density and, likely, water filled pore spaces.

At present we do not know if separated slurry sludge injected into corn fields as starter N and P fertilizer will cause substantial N_2O emissions, but we found that injecting whole slurry in a similar fashion increased N_2O emissions by 2-3 fold compared to broadcasting, as well as increasing N leaching (unpublished data). The large increases may be due to moist, anaerobic conditions in the furrows due to the large quantities of slurry needed to provide sufficient P for the corn (~30 kg P ha^{-1}). We have initiated a trial to assess if emissions from the injected manure can be significantly reduced, and crop N uptake increased, by adding a nitrification inhibitor to the injected manure. Use of the nitrification inhibitor DCD has been extensively examined for abating N_2O emissions on grassland and pastures and there is increasing interest in DMPP and Nitrapyrin and, for fertilizers, also urease inhibitors and various fertilizer coating products used to suppress nitrogen losses (Saggar *et al.*, 2009; Selbie *et al.*, 2015; Luo *et al.*, 2010).

Emissions from pastures

There has been a major shift by beef cattle farmers in Canada over the past decade towards more winter grazing on frozen, often snow-covered, soils to reduce operating costs associated with maintaining winter feedlots (McCartney *et al.*, 2004; Sheppard *et al.*, 2015). Feed is provided on winter pastures as fall-swathed cereals, stockpiled standing forages or conserved hay. There is evidence that there is more N recovery from winter grazing systems than from winter feedlots (Jungnitsch *et al.*, 2011) and this is probably associated with lower NH_3 emissions. However there is concern about leaching and N_2O losses from these sites, especially during spring thaw when there is little crop growth. The effect of freezethaw cycles on N losses in these systems needs particular research attention.

Grazing continues to be an enigma. Ammonia losses from urine patches are low (12.9% of applied N, 8% when measured by micrometeorological methods) due to the low ammonium content and rapid infiltration of urine (Selbie *et al.*, 2015). This is much lower than the combined NH_3 losses from housing, storage and land spreading in confinement operations. For Canadian beef cattle, NH_3 loss from grazing systems was 9.3% in contrast to 82.5% total losses in confinement systems coming from housing (28.6%), storage (9.4%) and landspreading (44.5%) (Sheppard and Bittman, 2013). However N recovery rates from urine patches are often less than 50%, and lower during periods of poor growth or if there is scorching. Urine patches at typical deposition rates of 500-1000 kg N ha^{-1} are N_2O hotspots emitting 11-21 kg N_2O-N ha^{-1} (based on 2.1% emission factor), and greater in areas heavily trafficked by cattle where there is compaction and loss of stand (Brùèek *et al.*, 2009; Sheppard and Bittman, 2011). Nitrate leaching from urine patches is estimated at 20% and possibly greater in areas of poor growth (Selbie *et al.*, 2015). These observations are consistent with the conclusions that halving grazing time each day and eliminating grazing in October reduced N loss and contributed to greater farm N use efficiency on dairy farms (Aarts *et al.*, 2000). However, grazing is recognized by the UNECE as a Category 1 method for abating NH_3 loss compared to confined cattle (with up to 50% NH_3 saving from housing alone) even given the assumption that N_2O emissions are higher on pastures than ungrazed forage (Groenestein *et al.*, 2014, Webb *et al.*, 2014). There is a worldwide effort to mitigate losses of N_2O from pastures using a range of strategies which include reducing N in feeds, treating pastures with nitrification inhibitors, and altering or reducing the grazing season (Selbie *et al.*, 2015; Luo *et al.*, 2010).

Conclusions

Forage systems are a paradox in terms of gaseous N emissions. Forages have a high N demand, a long growing season, and an effective root system all contributing to effective nutrient capture. However forages are restrictive in methods available for mitigating gaseous losses both as NH_3 and as N_2O. This is due both to practical and cost considerations. The animal-plant-soil system is complex and elusive in terms of research but also offers a variety of opportunities for efficiencies such as balancing farm feed and nutrient production to animal requirements. Particularly enigmatic for containment of reactive N are grazing systems.

At present, the levels of prospective mitigation of emissions are perhaps modest. However new ideas like acidifying manure in storages, novel inhibitor products, more durable legumes, better animal nutrition technology, ongoing improvements in manure application methodology with increasing adoption by farmers, and integrated approaches will continue to make incremental improvements in reducing losses of nitrogenous gases and other reactive N species from forages.

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