Agriculture, Ecosystems and Environment 171 (2013) 90–102

Contents lists available at SciVerse ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Estimated net application of ammoniacal and organic N from manure, and potential for mitigating losses of ammonia in Canada

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ARTICLE INFO

Article history:
Received 19 December 2012
Received in revised form 14 March 2013
Accepted 15 March 2013
Available online 29 April 2013

Keywords:
Ammonia
Greenhouse gas
BMP
Emission inventory
Nitrogen balance

ABSTRACT

Manure nitrogen (N) includes what can be generalized as organic N, which includes undigested N from the feeds; ammoniacal and easily hydrolysable N, which includes urea and uric acid; and nitrate/nitrite species, which are the least abundant. From excretion to landspreading, the largest change in N concentration occurs because of volatilization of ammonia (NH_3) from the ammoniacal and easily hydrolysable fraction. This process can be highly dependent on manure management, and some management strategies such as manure injection are largely designed to decrease NH_3 loss. This paper utilizes recent models of NH_3 emission from beef, dairy, swine and poultry production to estimate the net organic and ammoniacal N content of manure in Canadian Ecoregions before and after land spreading. Confinement versus grazing for beef is a major factor for overall net manure N application, and slurry versus solid manure is next most important. There are distinct differences among Ecoregions in the proportions of organic and ammoniacal N, so that generic assumptions are not appropriate. The estimates are mapped for all of Canada based on 2006 animal census. Several best management practices (BMPs) are evaluated using recent costing information (dollars per kg of NH_3-N saved from emission). Relatively low-cost BMPs related to slurry manure applied nation-wide could save 16 Gg NH_3-N year^{-1} for an estimated cost of $13 M. Other low-cost BMPs could increase this to a saving of 79 Gg NH_3-N year^{-1} or 26% of present emissions.

1. Introduction

Nitrogen is a problematic element in agriculture: it is both a costly plant nutrient when obtained as fertilizer and a critical environmental contaminant because of the inevitable emissions of ammonia (NH_3) and N_2O from fertilized crop land and manure, and the possibility of nitrate/nitrite and organic N leaching to groundwater and runoff to surface water. As a result, there is considerable effort to model N in agriculture, and at least some of this is directed toward guiding regulatory policy (Eilers et al., 2010; Lillyman et al., 2009). From a policy perspective, it is important to know the spatial distribution and intensity of land application of manure N, and the distribution between organic and ammoniacal forms of manure N retained in the soil. This paper describes the use of recent farm practices information and NH_3 emission models to map manure N application to agricultural land in Canada and to estimate costs of abatement measures.

Most models used to estimate national-level NH_3 emissions are simple linear accounting computations (Misselbrook et al., 2004; Velthof et al., 2012). Estimated N excreted by various livestock types is partitioned into total ammoniacal N (TAN), which is taken to include NH_3 generated by rapid hydrolysis reactions after excretion, and other N. The manure N is further partitioned to various manure management pathways. The NH_3 emission is estimated as a fraction of the remaining TAN (e.g., Lau et al., 2008; Dämmgenl et al., 2010), and the TAN is sequentially decreased by losses during housing, storage and landspreading (e.g., Misselbrook et al., 2004). Mechanistic models have been proposed for specific emission processes (e.g., Sommer et al., 2006; Smith et al., 2009; Ogejo et al., 2010), but such models are less well adapted for long-term and spatially extensive estimates and are often used as sources of emission fractions.

According to recent estimates Canadian agriculture produces manure with 1.08 Tg N year^{-1} (Drury et al., 2007; Huffman et al., 2008) relative to 1.54 Tg N year^{-1} applied as commercial fertilizers (Yang et al., 2011). Yang et al. (2011) estimated manure N produced in 2006 at 1.10 Tg of which 710 Gg went into storage systems and 390 Gg was directly deposited on pastures. They estimated that total available manure N after losses was 664 Gg N year^{-1}. New data on farm practices are now available thanks to the Livestock Farm Practices Survey (LFPS, Sheppard et al., 2009a, 2010, 2011a; Sheppard and Bittman, 2011, 2012). These data allow estimates of emissions from different types of housing (including pasture,

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http://dx.doi.org/10.1016/j.agee.2013.03.013

corrals, feedlots and barns) and storage systems and provide more detailed information on land application practices for manure across all sectors than previously possible. These data also provide better estimates of N excretion rates because of more detailed information on feeding practices. The LFPS farm data were stratified by Ecoregions, which also allows for temperature corrections for emissions factors as well as recognition of regional differences in husbandry and agronomy.

The objective of this paper is to determine the N excretion and NH_3 volatilization losses from manure through the entire manure handling system from housing through storage and land application. In addition, we estimate loss reductions and associated costs that may be possible and implications for crop use of manure N.

2. Methods

2.1. Livestock Farm Practices Survey (LFPS)

The LFPS survey was administered by Statistics Canada (Ottawa, ON, Canada) in 2006. Details of the survey and data handling, and an Ecoregion map, are included in Sheppard et al. (2009a). The Ecoregions sampled were 1: Atlantic Canada, 2: St. Lawrence Lowlands, 3: Manitoulin-Lake Simcoe-Frontenac Axis, 4: Lake Erie Lowland, 5: Boreal Shield, 6: Brown Soil Zone, 7: Dark Brown Soil Zone, 8: Black Soil Zone, 9: Lake Manitoba Plains, 10: Boreal Plains, 11: Montane Cordillera and 12: Pacific Maritime. Beef, dairy, swine, broiler, layer and turkey farmers were interviewed in each of at least 8 Ecoregions: Ecoregions were excluded when that livestock sector was not very economically significant. A total of 2981 farms responded to the survey. The sampling of farms was stratified so that within each Ecoregion, the farms chosen represented the range of farm sizes based on numbers of animals. The questions about manure storage and handling were the same for all livestock sectors. Data were closely vetted for errors. Log transformation was used on certain data that were obviously skewed, such as animal numbers per farm that ranged >10,000 fold. Analysis of covariance with Ecoregion as the categorical factor and a non-parametric index of farm size based on numbers of animals as the covariate was used to aid interpretation, with statistical significance accepted if $P < 0.05$. The survey results for poultry, pigs, dairy and beef sectors are discussed by Sheppard et al. (2009a, 2010, 2011a), Sheppard and Bittman (2012), respectively.

2.2. Ammonia emission models

Nitrogen excretion rates for each class of animals were determined separately for each Ecoregion based on protein content of feeds reported in the LFPS and published models (Sheppard et al., 2009b, 2010, 2011b; Sheppard and Bittman, 2012) and these differ somewhat among Ecoregions, and from the average USA values (ASAE, 2005) used by Yang et al. (2011) and Huffman et al. (2008). It should be noted that separate excretion factors were used for beef cattle on pasture and in confinement taking into account the contrasting feeds and especially seasonal variation in protein concentration in pasture herbage (Sheppard and Bittman, 2011). The feed-based excretion models allow for incorporating changes in feeding practices such as increased use of specific amino acids in poultry and swine feeds, and changes in feeding practices related to a shift from tie-stall to free-stall housing of dairy animals. Also important is the increased use of high-protein distiller's grain which is a waste product from ethanol facilities. The models at present do not vary the fraction of excreted N that is ammoniacal or urea with feeding and this remains an important development for future models.

The NH_3 emission models were as described by Sheppard et al. (2009b, 2010, 2011b) and Sheppard and Bittman (2011), and were designed to use as much of the LFPS data as possible. A schematic of information sources and density is given by Sheppard and Bittman (2011). Although the specifications required detailed computations, the basic model is:

$$E_C = TAN_{EX} \cdot \left[\sum_H f_H \cdot EF_H + \left(1 - \sum_H f_H \cdot EF_H\right) \cdot \sum_S f_S \cdot EF_S + \left(1 - \sum_H f_H \cdot EF_H\right) \cdot \left(1 - \sum_S f_S \cdot EF_S\right) \cdot \sum_L f_L \cdot EF_L \right] \quad (1)$$

where: E_C = NH_3 emission from the livestock sector C (kg NH_3/t, where t is some unit of time-usually a month), TAN_{EX} = excretion of total ammoniacal N (kg TAN/t), $\sum$ = the sum across the subscript management practices for housing/grazing (H), manure storage (S) or manure land spreading (L), f_H = fraction of the animal population that follows housing/grazing management practice H (unitless), EF_H = the emission fraction that applies to housing/grazing management practice H (unitless, expressed as fraction of TAN that is emitted), f_S = fraction of the animal population that follows manure storage practice S (unitless), EF_S = the emission fraction that applies to manure storage practice S (unitless, expressed as fraction of TAN that is emitted), f_L = fraction of the animal population that follows manure land spreading practice L (unitless), and EF_L = the emission fraction that applies to manure landspreading practice L (unitless, expressed as fraction of TAN that is emitted). The $(1 - x)$ formulation ensures mass balance: emissions at each subsequent stage are limited to a fraction of the TAN that has not been emitted until that stage. There is a separate model for each of the 6 livestock sectors from the LFPS. Emissions are calculated for each month of the year based on farming practices such as the timing of manure spreading, temperature and the probability of rain within a few days of spreading.

The models were modified for this study to also account for the non-ammoniacal N (non-TAN), which is mostly organic N and assumed to pass quantitatively from excretion through to soil application following the same manure management pathways already incorporated in the models. In fact, there will be losses because of mineralization of organic N to TAN, and denitrification and leaching of NO_3, both highly dependent on manure handling practices and neither considered to be major losses of N prior to landspreading (although both may have important environmental consequences). Additionally, there may be some increases in non-TAN because of biological incorporation of TAN depending on availability of C and O_2. Again, this is considered a minor process. Because nitrate and nitrite N are generally at low concentrations in manure, the non-TAN is largely organic N and we use these terms as synonyms here.

The models track a progressive gaseous loss of TAN as NH_3, and so the ratios among total N, TAN and organic N continue to change until at least several days after landspreading of the manure. The fraction of total manure N that is retained in soil initially as TAN (most of this will be eventually nitrified) varies strongly with farm practices, and so this is the important predicted quantity from the models.

Grazing is different from other animal production systems because the ammoniacal and readily hydrolysable N from the urine (all considered TAN) will often infiltrate the soil directly and may not even contact feces, so that the hydrolysis of urea is delayed until the urine is in the soil. Corrals are intermediate in this regard between grazing and full confinement such as barns. There remain some emissions of NH_3 from urine patches in pastures, these are accounted in the beef and dairy models, and the net TAN application to pasture land is computed.