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REVIEW

Methane and nitrous oxide emissions from Canadian dairy farms and mitigation options: An updated review

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Abstract: This review examined methane (CH_4) and nitrous oxide (N_2O) mitigation strategies for Canadian dairy farms. The primary focus was research conducted in Canada and cold climatic regions with similar dairy systems. Meta-analyses were conducted to assess the impact of a given strategy when sufficient data were available. Results indicated that options to reduce enteric CH_4 from dairy cows were increasing the dietary starch content and dietary lipid supplementation. Replacing barley or alfalfa silage with corn silage with higher starch content decreased enteric CH_4 per unit of milk by 6%. Increasing dietary lipids from 3% to 6% of dry matter (DM) reduced enteric CH_4 yield by 9%. Strategies such as nitrate supplementation and 3-nitrooxypropanol additive indicated potential for reducing enteric CH_4 by about 30% but require extensive research on toxicology and consumer acceptance. Strategies to reduce emissions from manure are anaerobic digestion, composting, solid–liquid separation, covering slurry storage and flaring CH_4, and reducing methanogen inoculum by complete emptying of slurry storage at spring application. These strategies have potential to reduce emissions from manure by up to 50%. An integrated approach of combining strategies through diet and manure management is necessary for significant GHG mitigation and lowering carbon footprint of milk produced in Canada.

Key words: dairy cattle, greenhouse gas, enteric fermentation, manure management, methane, nitrous oxide.

Résumé : Cet article de revue examine les stratégies pour mitiger les émissions de méthane (CH_4) et d'oxyde nitrique (N_2O) dans les fermes laitières canadiennes. L'objectif principal était la recherche effectuée au Canada et des régions climatiques froides avec des systèmes laitiers similaires. Des méta-analyses ont été effectuées pour évaluer l'impact d'une stratégie particulière lorsque des données suffisantes étaient disponibles. Les résultats indiquent que les options pour réduire le CH_4 entérique provenant des vaches laitières augmentaient la teneur en amidon alimentaire et la quantité de suppléments lipidiques alimentaires. Remplacer l'ensilage d'orge ou de luzerne avec l'ensilage de maïs à plus forte teneur en amidon diminuait le CH_4 entérique par unité de lait de 6 %. Augmenter les lipides alimentaires de 3 % à 6 % des matières sèches (DM — « dry matter ») à réduit rendement de CH_4 entérique de 9 %. Les stratégies comme les suppléments de nitrate et l'additif 3-nitrooxypropanol indiquent un potentiel de réduction du CH_4 entérique d'environ 30 %, mais nécessitent davantage de recherche sur la toxicologie et l'acceptation par le consommateur. Les stratégies pour réduire les émissions provenant du fumier sont la digestion anaérobie, le compostage, la séparation solide–liquide, la couverture du réservoir à lisier et le torchage du CH_4 ainsi que la réduction de l'inoculum de méthanogènes par vidange complet du réservoir à lisier lors de l'application au printemps. Ces stratégies ont le potentiel de réduire les émissions provenant du fumier jusqu'à 50 %. Une approche intégrée en combinant les stratégies au moyen de la gestion des aliments et du fumier est nécessaire

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Abbreviations: 3NO, 3-nitrooxypropanol; **AD**, anaerobic digestion; **ADF**, acid detergent fiber; **BW**, body weight; **C**, carbon; **C:N**, carbon to nitrogen; **CDDGSs**, corn-dried distiller grains with solubles; **CH_4**, methane; **CP**, crude protein; **CS:GS**, corn silage to grass silage; **DM**, dry matter; **DMI**, dry matter intake; **EE**, ether extract; **FCM**, 4% fat-corrected milk; **GEI**, gross energy intake; **GHG**, greenhouse gas; **HRT**, hydraulic retention time; **MCF**, methane conversion factor; **MCR**, methyl-coenzyme M reductase; **MDP**, management and design parameter; **N_2O**, nitrous oxide; **na**, not available; **NC**, not calculated; **nd**, not determined; **NDF**, neutral detergent fiber; **NH_3**, ammonia; **NSC**, nonstructural carbohydrates; **OMd**, organic matter digestibility; **PAM**, polyacrylamide; **SD**, standard deviation; **SF_6**, sulfur hexafluoride; **SSD**, separated solid; **TAN**, total ammoniacal N; **VFA**, volatile fatty acid; **VS**, volatile solid; **Y_m**, proportion of gross energy intake lost as enteric CH_4.

pour mitiger significativement la production de gaz à effet de serre et réduire l'empreinte carbone de la production de lait au Canada. [Traduit par la Rédaction]

Mots-clés : vaches laitières, gaz à effet de serre, fermentation entérique, gestion du fumier, méthane, oxyde nitrique.

Introduction

Livestock-related greenhouse gas (GHG) emissions account for about 60% of Canada's total agricultural GHG emissions and primarily consist of methane (CH_4) and nitrous oxide (N_2O) emissions from enteric fermentation and manure management, respectively (Environment Canada 2013). Based on a life cycle analysis (International Dairy Federation 2010), total GHG emissions from Canadian milk production were estimated at 9.4 Mt CO_2 equivalents yr^{-1} (Quantis, AGÉCO and CIRAIG 2012), representing about 13% of the total agricultural GHG emissions in Canada. When expressed relative to the total milk production, the GHG intensity or carbon (C) footprint of milk produced in Canada has been estimated to range from 0.67 to 1.2 kg CO_2 eq kg^{-1} fat- and protein-corrected milk depending on the study (see Table 1 in Jayasundara and Wagner-Riddle 2014). The largest contribution to emissions was CH_4 from enteric fermentation (46% of the total), followed by CH_4 and N_2O from manure management (27%), N_2O emissions associated with feed crop production (20%), and CO_2 emissions associated with on-farm energy use and transportation of inputs to farms (7%) (Quantis, AGÉCO and CIRAIG 2012).

At the global scale, GHG emissions from livestock were estimated at 7.1 Gt CO_2 eq yr^{-1}, representing 14.5% of the total anthropogenic GHG emissions (Gerber et al. 2013). As ruminants (mainly dairy and beef cattle) contribute the largest proportion (61%) to livestock-related GHG emissions (Gerber et al. 2013; Eshel et al. 2014), there is an increased pressure to reduce their C footprint. In response, major milk and meat producing countries have taken significant steps to mitigate GHG emissions from their production systems (e.g., US "cow of the future" program which targets 25% reduction of current level of GHG by 2025; Australian Carbon Farming Initiative that provides incentives for farmers to generate C offset credits through GHG mitigation activities). Although GHG emissions from the Canadian dairy farming sector have decreased over the last two decades (National GHG Inventory report, Environment Canada 2013), due to decrease in dairy cow population, there is an urgent necessity for identifying additional GHG reduction strategies for Canadian dairy farms.

Several comprehensive reviews describing the potential strategies for mitigating GHG emissions from ruminant livestock have been published during the past few years (Beauchemin et al. 2009*a*; Eckard et al. 2010; Martin et al. 2010; Grainger and Beauchemin 2011; Hristov et al. 2013; Montes et al. 2013). These reviews discussed mitigation strategies with a global focus covering both milk and meat production from ruminants, while Moate et al. (2014) focused on mitigation of CH_4 from enteric fermentation by dairy cattle in Australia and Knapp et al. (2014) in the United States. The Australian dairy system is largely a pasture-based milk production system; therefore, mitigation strategies may not be directly applicable to the Canadian dairy system which is predominantly a year-round confined system. In comparison, the review by Knapp et al. (2014), although applicable to the Canadian dairy system to a certain extent, is focused on enteric CH_4 mitigation strategies only. Therefore, there is a need for a comprehensive review of recent Canadian relevant research evaluating options for mitigating GHG emissions from enteric fermentation and manure management.

The objectives of this paper were to (1) review recent Canadian research on enteric CH_4 emissions, and CH_4 and N_2O emissions from stored manure related to dairy cattle, and (2) identify strategies for GHG mitigation that can currently be used in Canadian dairy farms and promising technologies that have potential to be used as mitigation strategies in the future. Although the primary focus is on Canadian research, applicable research from other cold climatic regions with similar dairy production systems was considered. In addition, efforts were taken to identify gaps in knowledge and recommend priorities for research and development for the reduction of the C footprint of Canadian milk.

Methane Emissions from Enteric Fermentation

Enteric CH_4 production arises principally from anaerobic microbial fermentation in the gastrointestinal tract of ruminants. Typically, 4.0–7.5% of gross energy intake (GEI) by dairy cows is lost as CH_4 in North America (Kebreab et al. 2006). There are two primary mechanisms that explain variation in enteric CH_4 production in ruminants: dietary carbohydrate fermented in the rumen, and the mechanism regulating availability of hydrogen for CH_4 production related to the stoichiometry of volatile fatty acid production (Johnson and Johnson 1995). Acetic acid production releases metabolic hydrogen whereas production of propionic acid is a net user of metabolic hydrogen. Therefore, increasing fractions of propionic acid produced relative to acetic acid are negatively associated with enteric CH_4 production. Biohydrogenation of fatty acids provides an alternative hydrogen sink to methanogenesis. Moreover, microbial protein synthesis from dietary proteins can result in either net consumption or net production of hydrogen (Czerkawski 1986).

Several methods have been used to measure CH_4 emissions from ruminants. These range from sophisticated chambers equipped with various types of gas analyzers to tracer techniques such as sulfur hexafluoride (SF_6) method and open-path laser method. These methods have been comprehensively discussed and compared in several previous reviews (e.g., Kebreab et al. 2006; Storm et al. 2012); here, we focus on measured enteric CH_4 emissions from Canadian dairy cows in response to evaluated mitigation options.

Strategies for mitigating enteric CH_4 emissions from dairy cows in Canada

A recent review by Knapp et al. (2014) classified studies on enteric CH_4 mitigation strategies into three categories: (1) feeding and nutrient management, (2) rumen modifiers, and (3) increasing animal production through genetic and management approaches. In recent years, several research groups in Canada conducted studies examining various enteric CH_4 mitigation strategies falling into one or more of those categories. The strategies studied include (1) increasing dietary starch content through increasing corn silage content in the forage fraction (Hassanat et al. 2013; Lettat et al. 2013; Benchaar et al. 2014); (2), increasing fat content in the diet via supplementation of specific fatty acids (Odongo et al. 2007*a*), adding oilseeds (Beauchemin et al. 2009*b*) or using lipid containing by-products that may be economically feasible, such as distiller's grain with solubles (Benchaar et al. 2013); and (3) through the use of feed additives such as yeast (Chung et al. 2011), fibrolytic enzymes (Chung et al. 2012), ionophores (Odongo et al. 2007*b*), plant bioactive compounds such as saponin (Holtshausen et al. 2009) or inhibitors such as 3-nitrooxypropanol (Haisan et al. 2014). In the following sections, these strategies are reviewed in terms of effectiveness (e.g., decrease of CH_4 per unit of milk) and limitations (e.g., nutrient digestibility declines) compared with the findings from the same or similar strategies applied outside Canada. Additionally, observed or potential effects of these strategies on manure CH_4 emissions are discussed in the context of whole farm carbon foot print.

Increasing corn silage content in the diet

Forages constitute the major proportion of dairy cow diets; however, few studies have investigated the effect of forage type on enteric CH_4 emissions. Canadian dairy feed consists mainly of barley silage in western Canada and corn silage and alfalfa silage or alfalfa hay in Ontario and Quebec, the two provinces with over 70% of Canada's dairy industry (McCartney and Horton 1997). Corn silage usually contains greater amounts of starch [e.g., 30% of dry matter (DM); Maizex 2015] than silages from other forages (e.g., 9.4% of DM in barley silage; Oba and Swift 2013). Feeding more starch without compromising rumen health (i.e., acidosis) and (or) production (e.g., milk fat depression) has been shown to be associated with less CH_4 losses (Mills et al. 2003) and improved milk yields (Khorasani et al. 1994). Therefore, increasing the proportion of corn silage at the expense of cereal or legume silage is considered a promising enteric CH_4 mitigation strategy, provided that the desired maturity stage of corn corresponding to high starch contents is achieved. Hassanat et al. (2013) and Benchaar et al. (2014) investigated the impact of increasing dietary corn silage at the expense of barley or alfalfa silage on enteric CH_4 production by Canadian dairy cows and observed declines of ~6% in emissions per kg of milk at 100% replacement rates. However, these emission intensity declines were primarily due to improved milk production (by 6%–16%) as the absolute CH_4 production (g cow^{-1} d^{-1}) did not change significantly.

We conducted a meta-analysis on the impact of increasing corn silage relative to the other silages following the methodology described in Alvarez-Fuentes et al. (2016). Forty-seven treatment means of enteric methane emission measurements from lactating cows fed corn silage (17.7%–53.3% of DM) and grass silage (5.5%–53.3% of DM)-based diets in the US and Europe (Dohme et al. 2004; Hindrichsen et al. 2006; O'Neill et al. 2011; van Zijderveld et al. 2011*a*, 2011*b*; Hollmann et al. 2012, 2013; Brask et al. 2013; Haque et al. 2014; Reynolds et al. 2014; Aguinaga Casañas et al. 2015; Livingstone et al. 2015; van Gastelen et al. 2015) were used. The results indicated that the ratio of corn silage to grass silage (CS:GS) (ranging from 0.5 to 7.0 with a mean of 3.2) was negatively associated (P = 0.007) with CH_4 production (g cow^{-1} d^{-1}) and explained 15% of variability in the treatment means. The CS:GS ratio was positively associated with milk yield ($P < 0.001$) and explained 39% of the variability. Consequently, CS:GS ratio had a strong negative relationship with CH_4 intensity (CH_4:milk, $P < 0.001$) and explained 54% of the variability. Overall, increasing dietary corn silage at the expense of other silages appeared to reduce enteric methane emission intensity primarily through improved milk production.

Feeding increased levels of corn silage can also change nutrient digestibility and excretions, which would ultimately affect CH_4 emission potential of manure (Külling et al. 2002; Nousiainen et al. 2009; Appuhamy et al. 2014). In Hassanat et al. (2013) and Benchaar et al. (2014), increasing corn silage at the expense of alfalfa silage or barley silage linearly increased apparent total-tract digestibility of organic matter (OMd) in Canadian cows, likely resulting in low C excretions in feces and thereby potentially low manure CH_4 emissions. In contrast, O'Mara et al. (1998) and Bernard et al. (2002) demonstrated that replacing grass silage with corn silage reduced OMd suggesting that increased levels of corn silage relative to grass silage in diet may lead to high manure CH_4 emissions. Hellwing et al. (2014) also demonstrated that dairy cows fed corn silage excreted more organic matter in feces per unit of dry matter intake (DMI) and emitted more CH_4 from manure (7.2 vs. 3.9 g per kg of energy corrected milk) than cows fed grass silage. The increased manure CH_4 emissions in Hellwing et al. (2014) counteracted the corn silage-induced enteric CH_4 emission reductions (19.4 vs. 21.8 g per kg of energy corrected milk) leading to similar total (enteric plus manure) emission intensities for both groups of cows. Overall, CH_4 emissions from manure can differentially respond to increasing dietary corn silage level depending on the forage it replaces, even though the enteric CH_4 emission intensities would decrease as a result of improved milk production. Therefore, if enteric CH_4 mitigation achieved with increasing dietary corn silage was to be meaningful, it is necessary to implement a suitable mitigation strategy to address potential increases of CH_4 from manure. In fact, this increased CH_4 production potential of dairy manure may be desirable with a mitigation strategy such as anaerobic digestion (AD), where CH_4 can be captured and used for energy generation (discussed in detail under manure CH_4 mitigation below). It is also necessary to consider the overall impact of additional upstream and soil GHG emissions resulting from increased corn silage production vs. decreased grass or alfalfa silage production to avoid possible GHG trade-offs at the whole farm level.

Adding lipids into the diet

Added dietary fat could decrease methanogenesis in several ways including (1) lowering the quantity of organic matter fermented in the rumen; (2) hindering the activity of rumen methanogens; and (3) through biohydrogenation of lipids rich in unsaturated fatty acids. Supplementation of dairy cow diets with lipids has been one of the most extensively experimented enteric CH_4 mitigation strategies. A systematic review by Eugene et al. (2008) concluded that lipid-supplemented diets containing, on average, 6.4% ether extract (EE) reduced CH_4 production in lactating dairy cows by 9% (~30 g cow^{-1} d^{-1}) compared with diets containing 2.5% EE. Furthermore, they observed that this reduction was mainly a consequence of decreased DMI, although milk production and milk composition were not affected. However, a meta-analysis by Grainger and Beauchemin (2011) showed a persistent reduction in enteric CH_4 per unit of DMI for dietary lipid supplementations. In another meta-analysis, Patra (2014) examined the impact of the composition of added lipids on enteric CH_4 production and reported that fats with high concentrations of C12:0, C18:3, and polyunsaturated fatty acids had marked inhibitory effect on CH_4 production independent of DMI in cattle. Odongo et al. (2007*a*) fed Canadian dairy cows with myristic acid (C14:0) at 5% of dietary DM. They observed that CH_4 intensity decreased by 29% without altering DMI, milk yield, or milk fat percentage. Beauchemin et al. (2009*b*) examined the impact of adding three sources of long-chain fatty acids: crushed sunflower, flax, and canola seeds on enteric CH_4 production in lactating dairy cows. All three oil seed supplementations reduced CH_4 production by 10%–17% without altering milk yield or milk composition; however, canola seed appeared to be more promising as it did not reduce digestibility. Benchaar et al. (2013) tested fat supplementation using increased levels of corn-dried distiller grains with solubles (CDDGSs) ranging from 0% to 30% of DM, which in turn increased crude fat content from 4.0% to 7.2% of DM, respectively. Milk yield increased linearly while enteric CH_4 intensity decreased linearly (15% decline at 30% CDDGS level). However, because the CDDGS supplementation suppressed OMd and N utilization efficiency (Benchaar et al. 2013), volatile solids (VS) and N excretion in manure could be increased, potentially leading to increased CH_4 emissions as well as increased NH_3 and N_2O emissions from manure. Therefore, CDDGS supplementation needs to be evaluated at the whole farm level not only for total GHG emissions, but also for other potential environmental issues resulting from increased NH_3 emissions. Furthermore, it is worthwhile to note that fat content in CDDGS available commercially to dairy farms in North America is not constant, as companies extract a greater percentage of the oil for biodiesel and human consumption compared with a few years ago. Benchaar et al. (2013) used CDDGS with 16.3% EE; however, most commercially available CDDGSs contain 7–10% fat. Therefore, further research is necessary using current commercially available CDDGS to evaluate the overall impact on enteric CH_4 emissions from dairy cows and CH_4 and N_2O emissions from manure.

To quantitatively summarize the impact of dietary EE on enteric CH_4 emissions, a metaregression analysis was conducted using a data set of measured CH_4 emissions from Canadian lactating dairy cows. This data set included 30 treatment means [with corresponding standard deviations (SD)] of CH_4 emission measurements from 11 published Canadian studies (Table 1). The data also included average dietary ingredient and nutrient composition, average milk yield and milk composition, days in milk, and body weight (BW) of each treatment group. Treatment means related to supplementation of nonconventional dietary ingredients (e.g., monensin and 3-nitrooxypropanol) were excluded. Furthermore, the data set included diverse dairy diets in Canada that range from a grass-legume forage-based diet dominant in Atlantic Canada (Fredeen et al. 2013), to corn silage and corn grain diet used in central Canada (Odongo et al. 2007*a*, 2007*b*; Benchaar et al. 2013, 2014; Hassanat et al. 2013) and barley silage and small grain-based diet used in western Canada (Beauchemin et al. 2009*b*; Holtshausen et al. 2009; Chung et al. 2012; Haisan et al. 2014). The metaregression was conducted following the approach described in

Table 1. A summary of the literature data (n = 30 treatment means from 11 studies[a]).

Variable[b]	Mean ± SD	Minimum	Maximum
DMI (kg cow^{-1} d^{-1})	21.4 ± 2.9	15.2	27.2
GEI (MJ cow^{-1} d^{-1})	402 ± 53	292	513
Milk yield (kg cow^{-1} d^{-1})	30.6 ± 5.4	14.9	39
FCM (kg cow^{-1} d^{-1})	28.8 ± 4.9	15.4	35.3
Milk fat (%)	3.6 ± 0.3	3.1	4.2
Body weight (kg cow^{-1})	652 ± 54	591	762
Diet composition (% of DM)			
Concentrate	45.6 ± 7	34.0	60.0
OM	92.4 ± 1.3	89.5	94.5
CP	17.0 ± 1.2	14.7	20
NDF	33.3 ± 3.3	26.5	40.8
ADF	21.6 ± 2.2	17.0	26.4
NSC	37.4 ± 3.7	31.2	46.6
Starch	21.4 ± 6.6	9.7	35.5
EE	4.7 ± 1.4	3.0	7.3
Enteric CH_4 (g cow^{-1} d^{-1})	411 ± 81	241	540
Y_m (% of GEI)	5.7 ± 0.9	3.9	8.2

[a]Studies included: Odongo et al. (2007*a*, 2007*b*); Beauchemin et al. (2009*b*); Holtshausen et al.'s (2009) chamber based study and SF6 study; Chung et al. (2012); Benchaar et al. (2013); Fredeen et al. (2013); Hassanat et al. (2013); Benchaar et al. (2014); Haisan et al. (2014).

[b]DMI, dry matter intake; GEI, gross energy intake; FCM, 4% fat-corrected milk; DM, dry matter; OM, organic matter; CP, crude protein; NDF, neutral detergent fiber; ADF, acid detergent fiber; NSC, nonstructural carbohydrates; EE, ether extract; Y_m, gross energy intake lost as enteric CH_4.

Table 2. Metaregression model describing factors associated with enteric methane emissions (g cow^{-1} d^{-1}) from lactating dairy cows in Canada.

Predictor variable	Estimate	*P*-value	95% Confidence interval
Dry matter intake (kg d^{-1})	12.4	<0.001	6.5 to 18.3
Body weight (BW)	0.46	<0.001	0.25 to 0.68
Dietary starch (DM %)	–6.00	<0.001	–7.63 to –4.39
Dietary ether extract – 3.0 (DM %)	–12.5	0.005	–21.2 to –3.8

Viechtbauer (2010). Dry matter intake, dietary nutrient composition, milk yield and milk composition, and BW were used as potential predictor variables. A new predictor variable EEc (EEc = EE – 3.0) was developed to represent EE increasing above 3.0% of DM, the common dietary EE content in dairy cow diets. The best-fitting model explained 92% of variability in CH_4 emission measurements using DMI, BW, dietary starch, and EE contents, which were significantly related to enteric CH_4 production (Table 2). A one unit increase in EE from 3.0% of DM was associated with a 12.5 g cow^{-1} d^{-1} reduction in CH_4 production, implying that total CH_4 production reduction associated with increased dietary EE from 3.0% to 6.0% of DM would be, on average, 37.5 g cow^{-1} d^{-1}. Given the average CH_4 emission of Canadian dairy cows receiving EE at the 3.0% levels was 417 g cow^{-1} d^{-1} (data not shown), the reduction indicates a 9% decrease in CH_4 production. When data from studies outside Canada were analyzed following the same method, a one unit increase in EE from 3.0% of DM was associated with a 16.5 g cow^{-1} d^{-1} reduction in CH_4 production, implying, on average, a 12% reduction in total enteric CH_4 production due to EE increasing from 3.0% to 6.0% of DM (data not shown). Overall, increasing dietary lipid up to ~6.0% of DM does not compromise rumen fiber digestibility, milk yield, and milk composition and appears to be a promising enteric CH_4 mitigation strategy for the Canadian dairy sector.

Dietary supplementation of ionophores

Monensin has been commonly used in Canadian dairy rations since 1996 and the benefits include improvements in milk production, antiketogenic effects, and reduced risk of acidosis (Duffield and Bagg 2000). Moreover, increased propionate to acetate ratio and reduced numbers of protozoa-generating hydrogen in the rumen with ionophores have indicated the potential for using monensin as a CH_4 mitigation strategy (McGuffey et al. 2001). While a large number of studies have evaluated the benefits of monensin on energy and N metabolism, production performance, health, and reproduction in dairy cows (see systematic reviews by Duffield et al. 2008*a*, 2008*b*, 2008*c*), a relatively smaller number of studies have investigated the effect of monensin on enteric CH_4 emissions from dairy cows. Appuhamy et al. (2013) conducted a meta-analysis based on 11 experiments on lactating dairy cows and concluded that the effectiveness of monensin for directly reducing enteric CH_4 production was small (6.0 g cow^{-1} d^{-1} or about 2% decrease in methane production relative to the control diet when cows were fed at a dose of 21 mg kg^{-1} DM), but the antimethanogenic effect of the monensin may be enhanced by dietary modifications, especially with dietary lipid supplementation and increased monensin dose. Furthermore, evidence for the long-term antimethanogenic effects of monensin is inconclusive. Appuhamy et al. (2013) found the antimethanogenic effect of monensin to be persistent over a period varying from 11 to 72 d after feeding in the majority of studies included in the meta-analysis while only one study indicating the effect lasting for 180-d period. In a separate meta-analysis covering 71 studies with >9000 cow records, Duffield et al. (2008*b*) concluded that monensin at current recommended levels (16-24 mg kg^{-1} feed DM) reduced daily DMI by 2% while improving the milk production by 2.3% in high producing lactating dairy cows. Considering the conclusions from these two meta-analyses, it is likely that adding monensin to dairy cow diets may have overall positive effects in reducing CH_4 intensity which need to be assessed using a whole system analysis approach (e.g., Capper and Hayes 2012).

Dietary supplementation of monensin could potentially affect CH_4 emissions from manure. Up to 40% of dietary supplemented monensin may be found in feces in its active form (Hilpert et al. 1984), indicating a possibility of antimethanogenic effects during manure fermentation. Several studies have been carried out to examine the impact of monensin on methanogenesis in small prototypes of anaerobic digesters. In those studies, a single or a continuous addition of monensin reduced CH_4 production (e.g., by 50%) initially but did not show an affect after 40 d suggesting that monensin was broken down or the microorganisms had adapted to it over time (Hilpert et al. 1984; Wildenauer et al. 1984; Fleming and Soos 2009). Monensin may affect CH_4 emissions from manure indirectly, by improving feed digestibility and composition of VS excreted in manure. For example, dietary supplementation of monensin was related to numerical increases of apparent total-tract OMd, crude protein, and NDF in Canadian cows (Plaizier et al. 2000; Benchaar et al. 2006; do Prado et al. 2015) indicating that the manure may have low CH_4 emission potential. Therefore, improved understanding of the impact of monenesin on both enteric and manure CH_4 emissions would assist in drawing more sensible conclusion of its impact on the whole farm C footprint.

Yeast supplementation

Commercially available yeast (*Saccharomyces cerevisiae*) products fall under a group of feed additives called direct-fed microbials which are recommended for use in cattle to mitigate rumen disorders and improve feed efficiency, cow health, and production performance (McAllister et al. 2011). Several systematic reviews have provided strong evidence for their beneficial effects in improving feed digestibility, milk production, and milk fat content in high producing dairy cows (Desnoyers et al. 2009; Robinson and Erasmus 2009; Poppy et al. 2012). It has also been suggested that yeast products have the potential to alter the fermentation process in the rumen in a manner that reduces CH_4 production (Boadi et al. 2004). Yeast has been found to cause a shift in H_2 utilization from methanogenesis to reductive acetogenesis in vitro (Chaucheyras et al. 1995). Chung et al. (2011) reported two strains of *S. cerevisiae* varied in their ability to mitigate enteric CH_4 emissions indicating that there may be a possibility to select yeast strains that lower enteric CH_4 emissions intensity. The authors fed nonlactating Holstein cows in Canada (five cows per treatment) with a 50% forage diet (solely barley silage) supplemented with two yeast strains. One strain did not affect CH_4 production although it was shown to do so in vitro. The other strain modified rumen toward a more glucogenic and a more acidic (pH = 6.2 ± 0.1 vs. 5.9 ± 0.1) environment providing less favorable conditions for methanogenesis and was associated with a 7% reduction (15.7 ± 0.6 vs. 16.9 ± 0.6 g kg^{-1} DMI) in enteric CH_4 yield. Because acidic and glucogenic rumen conditions often lead to acidosis, Chung et al. (2011) suggested further evaluation of this strategy in cows fed diets for which the risk of acidosis is low but CH_4 emissions are high. In a study using lactating cows in Finland (four cows per treatment), Bayat et al. (2015) supplemented two yeast strains to diets having the same forage (solely grass silage) content as the diets in Chung et al. (2011) but with less starch (12% vs. 28% of DM) and greater NDF (40.1% vs. 33.1% of DM) contents implying low acidosis and high CH_4 production risks, respectively. The rumen pH was 6.65 and not altered by yeast supplementations, but enteric methane yield (19.7 ± 1.3 vs. 21.4 ± 1.3 g kg^{-1} DMI) and intensity [13.8 ± 1.3 vs. 15.6 ± 1.3 g (kg of milk)$^{-1}$] decreased numerically by 8% and 12%, respectively. Moreover, as Bayat et al. (2015) pointed out, further investigations with greater number of replicates would provide a basis for true potential of yeast supplementation as an enteric CH_4 mitigation strategy for dairy cows.

Addition of fibrolytic enzymes

Adding fibrolytic enzymes to ruminant diets has been investigated as a method of enhancing fiber digestion for many years (Beauchemin et al. 2004). Increased substrate degradability by fibrolytic enzyme additives reported in in vitro evaluations has often been accompanied by concurrent decreases in acetate:propionate ratio in ruminal fluid indicating a lower availability of metabolic H_2 for methanogenesis. Consequently, Beauchemin et al. (2008) proposed supplementing diets with enzymes as a means of mitigating enteric CH_4 production. Moreover, the in vivo fermentation balance calculations in Arriola et al. (2011) showed that supplementation of fibrolytic enzyme could reduce rumen methanogenesis in lactating dairy cows. Chung et al. (2012) conducted an in vivo study involving enteric CH_4 emission measurements to understand the potential mode of action of fibrolytic enzymes in the rumen of dairy cows with respect to volatile fatty acid (VFA) profiles, pH, microbial populations, and CH_4 production in rumen. They did not observe any change in VFA profiles or pH but change in methanogen species and increase in CH_4 yield and intensity. A review by Meale et al. (2014) concluded that the application of fibrolytic enzymes to dairy cow diets has shown extremely variable results and, in the majority of studies, fibrolytic enzymes failed to improve milk production efficiency.

Saponin-rich compounds

Three groups of plant-derived bioactive compounds (tannins, saponins, and essential oils) have received increasing interest as potential feed additives for enteric CH_4 mitigation (Beauchemin et al. 2008). They are generally perceived as "natural" alternatives to chemical feed additives used in ruminant diets (Martin et al. 2010). These compounds have been shown to inhibit rumen protozoa that assist in hydrogen transfer for methanogenesis or have shown to exert direct inhibitory effects on methanogens (Beauchemin et al. 2008; Martin et al. 2010). In a Canadian study, Holtshausen et al. (2009) examined the impact of two major commercial sources of saponin: *Yucca schidigera* and *Quillaja saponaria*, first in vitro on CH_4 production and fermentation, and second in vivo with a slightly lower dose of saponin (1.0% of DM to avoid potential negative effects on digestion) on CH_4 production, nutrient digestibility, and milk production of lactating dairy cows. Increasing levels of both saponin sources in vitro decreased CH_4 production and acetate: propionate ratio in the buffered rumen fluid, despite being with an accompanied reduction in DM digestibility. A meta-analysis using data from 23 in vitro experiments also showed that saponin-rich sources decreased CH_4 production per unit of DM significantly and tended to be related to low OM digestibility. Seventy-five percent of these experiments used the saponin-rich sources such as *Yucca schidigera*, *Quillaja saponaria*, and tea at doses >4% of DM (Jayanegara et al. 2014). In the in vivo experiment, there was no change in enteric CH_4 production, milk yield, DM, energy, and CP digestibility due to feeding the saponin additives. Consistently, adding *Yucca schidigera* to *Pennisetum purpureum* grass at doses <1.0% of DM did not change enteric CH_4 production and OM digestibility in sheep (Canul-Solis et al. 2014). Moreover, supplementation of purified extract of *Yucca schidigera* to a prairie hay-based (1 or 2 g d^{-1}) diet did not alter digestibility of DM and NDF, and manure N excretions of beef steers. Overall, feeding saponin-rich compounds to ruminants does not appear to significantly affect enteric and manure CH_4 emissions.

Addition of 3-nitrooxypropanol to diet

Several chemical compounds such as bromochloromethane, 2-bromoethane, sulfonate, chloroform, cyclodextrin, and 3-nitrooxypropanol (3NO) have been tested for potential antimethanogenic effects in ruminants. Some have shown promising results although there have been concerns related to animal health, food safety, or environmental impact (Hristov et al. 2015). Nonetheless, 3NO has been recognized as one of the efficacious methane inhibitors and is speculated to inhibit a key enzyme of methanogenesis, methyl-coenzyme M reductase (MCR). Prakash (2014) studied the impact of 3NO on MCR and found 3NO quenching the active form of MCR via a radical type mechanism. Haisan et al. (2014) tested 3NO on enteric CH_4 production in mid-lactating Holstein cows by mixing (2500 mg d^{-1}) to a barley silage-based diet fed over 4 wk. The 3NO supplementation was associated with a 60% decline in enteric CH_4 yield and intensity (per kg of DMI or milk yield) without compromising DMI or milk production. In another study, Romero-Perez et al. (2015) observed 59% reduction in CH_4 emissions from Canadian beef heifers fed a similar barley silage-based diet supplemented with 3NO over 16 wk indicating the antimethanogenic effects of 3NO to be persistent. Consistently, Hristov et al. (2015) observed CH_4 yield and intensity to reduce constantly over a period of 12 wk in lactating dairy cows in the US fed a corn silage-based diet supplemented with three doses of 3NO (1100, 1700, or 2200 mg d^{-1}). The decrease in enteric CH_4 emissions improved from 28% to 33% as 3NO (supplementation increased from 1100 to 1700 mg d^{-1}), while DMI and milk yield were constant. Increasing 3NO supplementation to 2200 mg d^{-1} did not have an impact over the 1700 mg d^{-1} supplementation. Nonetheless, the CH_4 emission intensity decline achieved in Hristov et al. (2015) was half of the decline in Haisan et al. (2014). This discrepancy could be explained partly by the differences in DMI of cows. The average DMI of cows in Haisan et al. (2014) was considerably less than that of cows in Hristov et al. (2015) allowing slower passage rates and thereby greater rumen retention times for 3NO to exert most of its antimethanogenic effects relative to the dose supplemented. In both studies, 3NO was hand mixed into the TMR daily simulating an on-farm feeding scenario, allowing a gradual introduction of 3NO into the rumen as the animals consumed the feed. This synchrony between feed digestion and 3NO consumption appears to be critical in obtaining maximum antimethanogenic potential of 3NO (Romero-Perez et al. 2015). For example, Reynolds et al. (2014) dosed 3NOP directly into the rumen through a rumen cannula and observed a minor reduction in enteric CH_4 production by lactating dairy cows. Overall, 3NO supplementation appeared to be a very promising enteric CH_4 emission mitigation strategy. However, this strategy is still at early stage of development and requires extensive future research on its consistency for reducing enteric CH_4 emissions as well as issues related to toxicology and consumer acceptance.

Alternative hydrogen sinks (nitrates)

Promoting alternative biochemical pathways to remove metabolic hydrogen produced during the VFA formation in the rumen is another strategy suggested for mitigating enteric CH_4 production in ruminants (McAllister and Newbold 2008). A number of chemical agents that act as electron accepters have been tested for this purpose with variable results (McAllister and Newbold 2008); however, nitrate has received the most widespread and renewed interest due to its efficacy, consistency, and persistency in reducing enteric CH_4 production (see review by Lee and Beauchemin 2014). To the best of our knowledge, there are no published Canadian studies investigating the effect of nitrate additive in vivo with dairy cows. van Zijderveld et al. (2011*c*) observed 16% decreased CH_4 yield, persisting for about 107-d experimental period during which a nitrate supplemented diet was administered to lactating dairy cows. Use of nitrate directly or in any other form in dairy cow diets has not been approved in Canada, and there are major concerns regarding its potential toxicity to the animal.

Methane and Nitrous Oxide Emissions from Manure Management

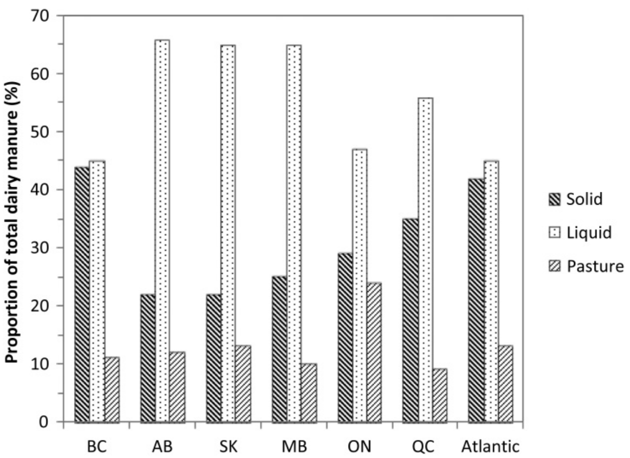
Dairy manure management systems in Canadian dairy farms

Manure management is the second largest source of GHG emissions after enteric fermentation in dairy farms in Canada (Quantis, AGÉCO, and CIRAIG 2012). In the Canadian dairy cattle system animals, spend a large proportion of their life confined, with only a few hours (<5 h) per day spent on small pasture areas in the summer (Sheppard et al. 2011*a*). Therefore, a major proportion of dairy manure is accumulated in barns that require systematic removal, storage, and disposal. Dairy manure contains ammoniacal nitrogen (resulting from hydrolysis of excreted urea), degradable carbon, and water — three major prerequisites for the production of N_2O and CH_4 (Chadwick et al. 2011). In addition, manure can emit ammonia (NH_3) which may contribute to N_2O emissions indirectly in the wider environment beyond the farm [Intergovernmental Panel on Climate Change (IPCC) 2006]. Greenhouse gases can be emitted from the point of manure excretion to the time of application to crops as a nutrient source. Activities that include the routine cleaning of barns or animal-confinement areas, delivery of manure to main storage, and land application are considered part of manure management. In this review, we focus on CH_4 and N_2O emissions from dairy housing and storage of the manure management continuum. A recent modelling study has indicated that about 28% of total ammoniacal N (TAN) excreted by dairy cattle can be lost as NH_3 at the housing and storage stages under average dairy farm practices in Canada (Sheppard et al. 2011*b*). Hristov et al. (2011) comprehensively reviewed mitigation strategies for reducing NH_3 emissions from dairy farms (and from beef feedlots) with special focus on Canada and USA. The major GHG emitted after dairy manure is applied to crops is N_2O emissions from soil (IPCC 2006), and strategies to mitigate these emissions were recently reviewed by VanderZaag et al. (2011*a*). Therefore, we will confine our discussion to potential implications from GHG mitigation strategies applied to housing and storage on the subsequent N_2O emissions from soil, whenever data related to these aspects are available from the reviewed studies.

Manure management systems in dairy farms can be broadly grouped into three systems: liquid manure, solid manure, and manure excreted directly on pasture. The anaerobic nature of liquid manure systems increases the potential for CH_4 production and decreases N_2O production, whereas solid manure systems can be substantial sources of N_2O while contributing relatively smaller amounts of CH_4 (IPCC 2006). Manure directly excreted on pasture by grazing dairy cattle can be a substantial source of N_2O while contributing negligible amounts of CH_4 (Chadwick et al. 2011; not reviewed here). Therefore, the type of manure management system has an important implication on the baseline GHG emissions from dairy manure management at a given farm and their mitigation potential.

There are appreciable regional differences in the proportion of total dairy manure managed using different manure management systems (Fig. 1). This variation is largely associated with regional differences in farm size and housing types used for dairy cattle (Sheppard et al. 2011*a*). In western Canada (British Colombia, Alberta, Saskatchewan, and Manitoba), average farms' size (number of dairy cows per farm) is relatively higher than that in eastern Canada (150 vs. 70 in Ontario, Quebec, and Atlantic provinces) (Canadian Dairy Information Centre 2014), and larger farms tend to use free-stall housing with liquid slurry systems due to practical and economic reasons (Sheppard et al. 2011*a*). In addition to these regional differences, data from Ontario have indicated that over the two decades between 1990 and 2011, consolidation of dairy farms has led to an increase in the average farms' size accompanied with increasing

Fig. 1. Regional differences in the proportion of total dairy manure managed using different manure management systems in Canada. Abbreviations: BC (British Columbia), AB (Alberta), SK (Saskatchewan), MB (Manitoba), ON (Ontario), QC (Quebec), Atlantic (all maritime provinces) [redrawn from data provided in Quantis, AGÉCO, and CIRAIG (2012), which are based on a national survey on dairy farm practices in 2005 reported by Sheppard et al. (2011*a*)].



proportions of dairy manure being managed using liquid systems (Jayasundara and Wagner-Riddle 2014). Furthermore, the survey data reported by Sheppard et al. (2011*a*) showed that within a single dairy farm, manure from lactating cows is likely to be managed as liquid while manure from dry cows and heifers is managed as solid. Therefore, mitigation strategies cannot be assumed to apply uniformly across all regions and need to be customized to groups of dairy farms that are similar in characteristics.

Methane and N_2O emissions from dairy housing

Methane and N_2O emissions from manure within dairy animal housing (before removing to main storage) can be highly variable, depending on several factors, e.g., barn floor type, bedding, manure cleaning method, and frequency. The data summarized in Table 3 present the measured CH_4 and N_2O emissions from dairy housing in cold climates (annual average temperature <10 °C), including three studies completed in Canada (Kinsman et al. 1995; van Vliet et al. 2004; Ngwabie et al. 2014). Methane emissions ranged from 321 to 599 g CH_4 cow^{-1} d^{-1} in these whole barn studies (Table 3) and comprised enteric fermentation and emission from manure within the barn except for a few studies that measured or estimated these sources separately (e.g., Kinsman et al. 1995; Marik and Levin 1996; Ngwabie et al. 2014). In these three studies, manure was temporarily stored (usually for 2–3 wk) in subfloor pits or pits within the barn before moving into the long-term storage, and CH_4 emissions from manure contributed 6%–20% (or 25–56 g CH_4 head^{-1} d^{-1}) of the whole barn CH_4 emissions (Table 5). In contrast, barn floor measurements (animals excluded) obtained using the dynamic chamber method indicated considerably lower manure CH_4 emissions from dairy housing. For example, Van Vliet et al. (2004) reported about 1.4 g CH_4 cow^{-1} d^{-1} from the concrete alley floor of a free-stall dairy barn (225 Holstein cows, sand or saw dust bedding, manure cleaned from the floor six times per day) in British Colombia (Canada). Similarly, Adviento-Borbe et al. (2010) measured 8.0 g CH_4 cow^{-1} d^{-1} emitted from the barn floor of a free-stall barn housing 60 lactating cows with manure cleaned twice daily in Pennsylvania, USA. Hence, significant CH_4 emissions from manure prior to moving into outdoor storage appear to occur mostly when manure is temporarily held within barns but not from excreta on the barn floor.

The impact of different floor types and manure cleaning practices on N_2O emissions from dairy barns is clearly evident from the summarized results (Table 3). For example, Zhang et al. (2005) studied whole barn emissions from nine dairy barns with different floor types and manure handling methods in Denmark, and reported N_2O emissions ranging from 0.1 to 7.0 g N_2O cow^{-1} d^{-1}. Of the barns with solid floors, grooved concrete elements on the floor coupled with liquid drain (that facilitated quick drain off of urine) and solid scraper systems emitted relatively low rates of N_2O (<1.0 g N_2O cow^{-1} d^{-1}), compared with, e.g., "hot rolled asphalt floor" with liquid drain and solid scraper (~3.0 g N_2O cow^{-1} d^{-1}, Table 3). Of barns with slatted floors, a back flushing system resulted in lower N_2O emissions (0.1 g N_2O cow^{-1} d^{-1}) compared with "scraper in gutter" system (2.0 g N_2O cow^{-1} d^{-1}). These authors reported the same general trend for NH_3 emissions as observed for N_2O, indicating the possibility of mitigating both direct N_2O emissions and potential indirect N_2O emissions by matching floor types with manure cleaning methods.

In a whole barn study in Ontario (Canada), Ngwabie et al. (2014) reported N_2O emissions averaging 1.6 g N_2O cow^{-1} d^{-1} from a free-stall dairy barn with a mixture of solid and slated concrete floors with scraper in manure alley, cleaned six times a day, close to values reported by Zhang et al. (2005) (Table 3). In contrast, van Vliet et al. (2004) reported considerably lower N_2O emissions (<0.02 g N_2O cow^{-1} d^{-1}) from the solid concrete alley floor in a study conducted in British Colombia, Canada. These limited Canadian studies support the notion that N_2O emissions from dairy cattle housing can be reduced by matching certain floor types with cleaning methods. Uncharacteristically high levels of whole barn N_2O emissions (22 g N_2O cow^{-1} d^{-1}) were reported by Leytem et al. (2013) for a southern Idaho (USA) dairy farm. However, these measurements included emissions from the adjacent exercise yards which had accumulated solid manure mixed with urine due to less frequent manure removal compared with frequently cleaned barn floors.

Table 3. Summary of research that quantified CH_4 and N_2O emissions from dairy housing in cold climatic (annual average temperature <10 °C) countries.

Reference and study location	Housing, ventilation,[a] and floor type	Manure cleaning method	Bedding	Season	CH_4 (g cow^{-1} d^{-1})	N_2O (g cow^{-1} d^{-1})
Whole barn measurements using tracer gas or CO_2 balance methods						
Kinsman et al. (1995); ON, Canada	Tie-stall, MV, concrete floor	Gutter scraped to subfloor tank that emptied/23 wk	Straw	Summer–fall	420.3 (6%)[b]	nd[c]
Marik and Levin(1996); Germany	Tie-stall, MV, slatted floor	Drained to subfloor tank that emptied/23 wk	na[d]	Spring	376.3 (~20%)[b]	nd
Snell et al. (2003); Germany	Free-stall, NV, grooved concrete, slatted alley	Urine drained into slats, feces scraped	na	Winter	450.7	nd
	Free-stall, NV, mastic asphalt floor	Alley scraped to outside	na	Winter	352.5	nd
	Free-stall, NV, (floor type not explained)	Alley scraped to outside	na	Winter	599.2	nd
	Free-stall, NV, slatted floor	Not explained	na	Winter	456.1	nd
Zhang et al. (2005);	Free-stall, NV, solid concrete (b1)[e]	Delta scraper	na	Summer–fall	505.5	0.6
Denmark	Free-stall, NV, hot rolled asphalt (b2)	Solid scraper and liquid drain	na	Summer–fall	456.4	3.0
	Free-stall, NV, concrete (grooved) (b3)	Solid scraper and liquid drain	na	Summer–fall	321.1	0.7
	Free-stall, NV, concrete (profiles) (b4)	Solid scraper and liquid drain	na	Summer–fall	387.3	1.8
	Free-stall, NV, slatted (b5)	Scraper in 40-cm-deep slurry channel	na	Summer–fall	483.5	1.9
	Free-stall, NV, slatted (b6)	Alley back flushing	na	Summer–fall	414.5	0.1
	Free-stall, NV, slatted (b7)	Circulation (no acid)	na	Summer–fall	488.1	7.0
	Free-stall, NV, slatted (b7)	Circulation (with acid)	na	Summer–fall	336.1	1.3
	Free-stall, NV, slatted (b8)	Scraper on slatted floor and circulation	na	Summer–fall	444.9	2.0
	Free-stall, NV, slatted (b9)	Circulation (with/without additive)	na	Summer–fall	455.6	4.9
Ngwabie et al. (2009); Sweden	Free-stall, NV, concrete floor, slatted alley	Gutter (under slatted alley) scraped to outside twice per day	na	Winter–spring	327.2	nd
Ngwabie et al. (2011); Sweden	Free-stall, NV, concrete floor	Gutter scraped hourly to inside pit that is emptied twice per day	Peat on rubber mat	Winter–spring	311.0	nd
Samer et al. (2012); Germany	Free-stall, NV, concrete floor	Alley scraped to outside several times per day	na	Winter	465.3	50.3
Wu et al. (2012); Denmark	Free-stall, NV, concrete floor, slatted alley	Gutter (under slatted alley) scraped to outside, 12 times per day	na	Spring, summer, fall	353.6	nd
Leytem et al. (2013); Idaho, USA	Free-stall, NV, concrete floor (exercise yards included)	Alleys flushed 23 times per day	SSD[f]	All seasons	409.9	22.1
Ngwabie et al. (2014); ON, Canada	Free-stall, NV, concrete and slatted floors	Scraped to inside pit 6 times per day, pit emptied biweekly	Straw/SSD on rubber mat	Spring–fall	419.8 (6%)[b]	1.6
Barn floor measurements using chamber method						
van Vliet et al. (2004); BC, Canada	Free-stall, NV, concrete floor	Bedding raked, alley scraped, 6 time per day	Sawdust or sand	Spring, summer	1.4	0.002
Adviento-Borbe et al. (2010); PA, USA	Free-stall, NV, grooved concrete	Alleys scraped 2× per day	Sand	Spring, summer	7.9	0.004

[a]Ventilation types: MV, mechanically ventilated; NV, naturally ventilated.
[b]For studies that reported CH_4 emissions from manure present in the barn separately, the value in parentheses is the proportion of total CH_4 attributed to manure.
[c]nd, not determined.
[d]na, not available (whether bedding was used or not used has not been included in the original publication).
[e]b1 to b9: barn 1 – barn 9.
[f]SSD, separated solids from digested dairy manure.

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Furthermore, exercise pens had soil surfaces which may have facilitated N_2O emissions.

Methane and N_2O missions from stored dairy manure

We compiled peer-reviewed research from Canada and other cold climates reporting CH_4 and (or) N_2O emissions from farm-scale and pilot-scale storages closely mimicking farm-scale manure storages (Table 4). The intent was to summarize baseline GHG emissions from dairy manure, so results from experimental treatments designed to mitigate GHG emissions (e.g., use of artificial covers) could be compared (next section). Published studies have not measured emissions for a complete year at one location; therefore, deriving annual emission factors (unit mass of gas emitted cow^{-1} yr^{-1}) directly from the measured data is challenging. Another serious limitation was the lack of sufficient background information such as the "volume of manure present in storage" when measurements were made, particularly in studies that only presented emissions per unit of storage surface area. Nevertheless, for liquid manure storages, we found seven studies that measured emissions during the warm season (June to November), and two studies during the cold season (December to May). Two year-round studies that measured CH_4 emissions from farm-scale liquid dairy manure storages at Askov, Denmark, and Hokkaido, Japan, were also included (Table 4) as climatic conditions were similar to those in major dairy regions in Canada.

The length of storage for dairy manure is generally >100 d at Canadian farms (Sheppard et al. 2011*a*), in compliance with environmental regulations on manure application timing; thus, there is sufficient time to build enough methanogen population in manure to cause CH_4 formation and emissions (Massé et al. 2003). In this context, ambient temperature has a profound influence on the rate of CH_4 emissions from stored dairy manure (Fig. 2). Measured CH_4 emissions from liquid dairy manure during the warm season averaged 21.8 ± 4.0 g CH_4 m^{-3} d^{-1}, while during the cold season, emissions were 75% lower at 5.1 ± 2.1 g CH_4 m^{-3} d^{-1} (Table 4). Corresponding seasonal values for CH_4 conversion factor (MCF, the proportion of maximum CH_4 production potential that is achieved in a specific manure storage system) are calculated using the measured CH_4 emissions averaged at $24.7\% \pm 6.6\%$ for the warm season and $3.9\% \pm 1.4\%$ for the cold season storage (Table 4). These MCF values would amount to an annual average MCF of about 14%, which is lower than the IPCC (2006) recommended MCF values of 17%–25% for liquid dairy manure stored without a natural crust in cold climates, but compares well with 10%–15% recommended for liquid manure stored with a natural crust, which is typical of Canadian conditions.

Only two longer term studies on solid manure were available for consideration (Husted 1994; Pattey et al. 2005) indicating CH_4 emissions from solid dairy manure stored for 100–180 d averaged about 12.8 ± 2.3 g CH_4 m^{-3} d^{-1} or 40% lower relative to the warm season emissions from liquid manure (Table 4). Based on data from Husted (1994), CH_4 emission from solid dairy manure during the cold season was even smaller, only about 0.1 g CH_4 m^{-3} d^{-1} (Table 4). Corresponding seasonal values of MCF for solid dairy manure were $4.8\% \pm 0.5\%$ for the warm season and 0.1% for the cold season (Table 4). In comparison, the recommended annual average MCF for solid manure stored in cold climates is 2% (IPCC 2006). Emissions of N_2O from liquid dairy manure storages were minimal (less than 0.1 g N_2O m^{-3} d^{-1}) even during the warm season; however, N_2O emissions from solid manure were >20 times higher than that from liquid dairy manure (Table 5).

Models for estimating CH_4 and N_2O emissions from dairy manure

Monitoring emissions from the large number of farms is not practically possible; therefore, prediction models are required for quantifying CH_4 and N_2O emissions from manure management on different farms before and after adoption of any potential mitigation strategy. At present, the IPCC tier 2 model (IPCC 2006) is the most widely used model for estimating CH_4 and N_2O emissions from dairy manure (or any other livestock manure). Several farm models that are available in the literature for estimating GHG emissions from dairy farms at the whole farm scale also use IPCC tier 2 algorithms directly or with minor variations [e.g., Farm GHG (Olesen et al. 2006), Dairywise (Schils et al. 2007*a*), Farm Sim (Saletes et al. 2004), Sims Dairy (Schils et al. 2007*b*), Holos (Little et al. 2008), Dairy Dyn (Lengers and Britz 2012), IDEAM (Doole et al. 2013)].

The IPCC (2006) tier 2 model is an empirical model that uses four parameters for estimating CH_4 emissions from stored manure [VS excretion rate, maximum CH_4 generation potential (Bo), manure management system distribution factor (MS), and MCF for a given manure management system] and three parameters for estimating N_2O emissions from stored manure [N excretion rate (Nex), MS, and N_2O emissions' factor for each manure management system ($N2O\text{-}_{EFMS}$)]. These parameters can be derived using country-specific production characteristics related to dairy production, while MCF is based on the average ambient temperature and type of manure management system (IPCC 2006). One important criterion for any model is its ability to predict GHG emissions within an acceptable accuracy (Kebreab et al. 2006). In this regard, VanderZaag et al. (2013) reviewed the application of the IPCC (2006) tier 2 model for estimating CH_4 emissions from manure management under the context of Canadian GHG inventory. They concluded that the inability of MCF (as applied in the IPCC model) to accurately account for variations in manure retention time and inoculum level (old manure left over after emptying) was a major weakness. Another limitation

Table 4. Daily rate of CH_4 emissions, total CH_4 emissions over the storage period expressed per unit weight of volatile solids (VS), and methane conversion factor (MCF) calculated using the measured total CH_4 emissions from liquid dairy manure in cold climatic countries.

Reference; research location;[a] sampling frequency; technique	Storage condition, manure retention time (no. of days)	Season, average air temperature (°C)	Daily average CH_4 emissions		Total CH_4 production ($g\ CH_4\ kg^{-1}$ VS)	MCF (%)[b]
			$g\ CH_4\ m^{-2}\ d^{-1}$	$g\ CH_4\ m^{-3}\ d^{-1}$		
CH_4 Emissions from liquid dairy manure						
Studies involving pilot-scale, batch loaded storage						
Pattey et al. (2005); ON, Canada; twice a week full-day; flow-through chamber	Open, natural crust, 150 d	Warm season,[c] 19.8	26.4	26.4	26.9	16.8
VanderZaag et al. (2009); NS, Canada; continuous; flow-through chamber	Open, natural crust, 121 d	Warm season, 16.9	36.6	28.2	64.3	40.2
VanderZaag et al. (2010*a*); NS, Canada; continuous; flow-through chamber	Open, natural crust, 165 d	Warm season, 15.3	24.4	18.8	48.0	30.0
VanderZaag et al. (2010*b*); NS, Canada; continuous; flow-through chamber	Open, natural crust, 158 d	Cold season, −0.5	0.9	0.6	1.5	1.0
Wood et al. (2012); NS, Canada; continuous; flow-through chamber	Open, natural crust, 180 d	Warm season, 18.4	64.9	42.5	148.0	61.6
Wood et al. (2014); NS, Canada; continuous; flow-through chamber	Open, natural crust, ~155 d	Warm season, 14.2	14.1	8.8	17.1	7.1
Studies involving farm manure storages						
Husted (1994); Askov, Denmark; biweekly full-day sampling; dynamic chamber	Above-grade open, natural crust in some months, ~182 d	Warm season, 13.4 Cold season, 4.4	na[d] na	13.1 3.2	16.6 9.5	10.4 5.9
Kaharabata et al. (1998); QC, Canada; trace gas technique; daily at 12:30–2:30 pm	Above-grade open, natural crust, ~180 d	Warm season, 17.0	202.7	NC[e]	NC	NC
VanderZaag et al. (2011*b*); Bright, ON, Canada; continuous; micromet mass balance	Above-grade open, natural crust, ~180 d	Cold season, −0.6	4.0	5.1	4.9	2.1
C. Wagner-Riddle (unpublished data); Embro, ON, Canada; continuous; micromet mass balance	Open, earthen tank, natural crust, ~180 d	Warm season, 14.2	28.5	21.3	32.4	20.3
Minato et al. (2013); Hokkaido, Japan; 3-wk-long seasonal measurements in all seasons; floating dynamic chamber	Above-grade, open, natural crust, ~182 d	Warm season, 13.6 Cold season, 1.1	54.8 23.9	14.9 11.0	18.3 10.2	11.4 6.4
VanderZaag et al. (2014); Prescott-Russell, ON, Canada; 2–3 wk sampling in October to November and April to May; inverse dispersion technique	Above-grade open, natural crust, ~180 d	October to November, 9.7 April to May, 10.1	33.7 12.5	NC NC	NC NC	NC NC
CH_4 Emissions from solid dairy manure						
Pattey et al. (2005); ON, Canada; twice a week full-day; flow-through chamber	Stockpile, straw bedding included, 98 d	Warm season,[c] 19.8	14.4	14.4	8.1	5.1
Husted (1994); Askov, Denmark; biweekly full-day sampling; dynamic chamber	Stockpile, straw bedding included, ~180 d	Warm season, 13.1 Cold season, 4.2	na na	11.2 0.11	7.0 0.1	4.4 0.1

[a]QC, Quebec; ON, Ontario; NS, Nova Scotia.

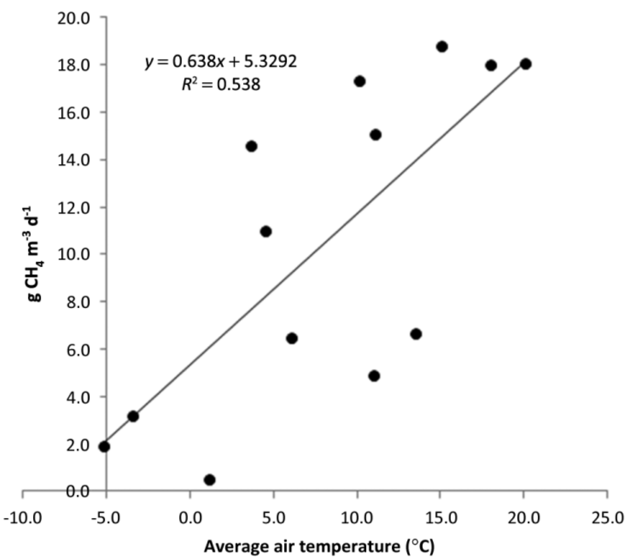
[b]MCF = [measured total CH_4 emissions ($L\ CH_4\ kg^{-1}$ VS)/maximum CH_4 production potential of dairy manure ($L\ CH_4\ kg^{-1}$ VS)] × 100, where maximum CH_4 production potential (B_0) of dairy manure was assumed to be 240 $L\ CH_4\ kg^{-1}$ VS (IPCC 2006).

[c]Warm season: late spring–summer–early fall (June to November); cold season: late fall–winter–early spring (December to May).

[d]na, not available in the original publication.

[e]NC, not calculated, because required supporting data were not provided in the original paper.

Fig. 2. Relationship between ambient temperature and the rate of CH_4 emissions from farm-scale dairy manure slurry tanks [Data were taken from four studies (C. Wagner-Riddle, unpublished data; Husted 1994; VanderZaag et al. 2011*b*; Minato et al. 2013) reporting measured CH_4 emissions from farm manure tanks under general farm practices (continuous loading of manure to storage as produced and manure being removed for application to crops in April/May and October/November). Reported monthly CH_4 emissions and monthly air temperature were averaged over four distinct seasons (winter, spring, summer, and autumn) for deriving the relationship].



was the fact that predicted emissions obtained using this model had not been verified independently, using measured emissions from farm-level manure storages. VanderZaag et al. (2011*b*) reported predicted IPCC tier 2 emissions to be about 50% higher than measured emissions due to the large discrepancy in MCF derived using the approach of Mangino et al. (2001) as recommended by IPCC (2006). Similar observations were reported in two other farm-level studies involving liquid swine manure storages in Canada (Park et al. 2006; Flesch et al. 2013).

The approach of Mangino et al. (2001) for deriving MCF has been developed for anaerobic manure treatment lagoons. This approach considers manure loading rate adjusted by a poorly defined management and design parameter (MDP), the average monthly air temperature, and van't Hoff Arrhenius function to estimate CH_4 production. It does not consider other critical factors affecting methanogen activity and CH_4 generation, e.g., inoculum level and manure chemical characteristics (Kebreab et al. 2006). Differences in functional aspects between commonly used manure storages in Canadian farms (i.e., earthen, concrete or steel tanks that simply hold manure until disposal), and "anaerobic manure treatment lagoons" that use intermittent loop flush system designed to optimize AD (USEPA) may also affect the accuracy of MCF predictions using the Mangino et al. (2001) approach. A major difference between tanks and lagoons is the VS loading rate which is more than 300-fold higher in earthen, concrete, or steel tanks than in anaerobic lagoons (37.0 vs. 0.12 kg VS d^{-1} m^{-3}) (Zahn et al. 2001). Methane flux was more than 25-fold higher in a lagoon system compared with a concrete slurry tank even though manure was originated from an identical swine population (Zahn et al. 2001). Higher VS loading rate inevitably is associated with higher TS and NH_4^+ loading rates: two critical factors affecting CH_4 formation (Kebreab et al. 2006). It has been reported that CH_4 production in manure with high TS is inhibited by high levels of NH_4^+ (Massé et al. 2003); thus, for improvements in deriving MCF accurately, critical manure properties and functional differences in manure management systems need to be considered in addition to the effect of ambient temperature.

To overcome the limitations in empirical models such as those found in the IPCC tier 2 model, dynamic mechanistic (process based) models that can accurately describe various biochemical pathways involved in GHG generation from manure were recommended (Kebreab et al. 2006). Currently, two such models covering the complete manure management chain (handling, storage, and land application) have been developed [i.e., integrated farm system model (Rotz et al. 2015), Manure-DNDC (Li et al. 2012)]. This is promising; however, their wide-scale application for estimating GHG emissions and assessing mitigation strategies requires satisfactory validation using on-farm GHG measurements.

Strategies for mitigating CH_4 and N_2O emissions from dairy manure

Anaerobic digestion

Anaerobic digestion (AD) is a process that promotes degradation of organic matter in biological wastes under an oxygen-free environment using microbial consortia composed of hydrolytic and fermentative bacteria as well as acetogens and methanogens (Massé et al. 2011). This process is used to stabilize biological waste resulting in a biogas (a mixture of gas mainly composed of CH_4 and CO_2) and a nutrient-rich, relatively stable and odorless sludge called digestate. Biogas contains 60%–70% CH_4 (Massé et al. 2011) and can be captured to be used as renewable fuel. When designed and operated properly, ensuring against gas leakages from the digester and emissions from the digested slurry, AD offers one of the most effective methods for reducing CH_4 emissions from dairy manure management (Frear et al. 2011). Additionally, AD offers other benefits such as reducing pathogens and odor in manure.

Ideally, AD is suited for mitigating GHG emissions from manure in dairy farms that already use liquid manure handling systems. Implementation of AD in these farms can be done with relatively less complicated

Table 5. Rates of N_2O emissions and experimentally derived N_2O emission factors for liquid dairy manure and solid cattle manure in Canada.

Reference; research location;[a] gas measurement technique	Storage condition, manure retention time (no. of days)	Rate of N_2O emissions (g N_2O m^{-3} manure d^{-1})	Emission factor (kg N_2O-N kg N)
N_2O emissions from liquid dairy manure			
Pattey et al. (2005); ON; twice a week full-day measurement using flow-through chamber	Open (pilot) storage, batch filled, natural crust on surface, 150 d	0.2	NC[b]
VanderZaag et al. (2009); NS; continuous measurement using flow-through steady-state chamber	Open (pilot) storage, batch filled, natural crust on surface, 121 d	0.045	0.002
VanderZaag et al. (2010*a*); NS; continuous measurement using flow-through steady-state chamber	Open (pilot) storage, batch filled, natural crust on surface, 165 d	0.064	0.003
VanderZaag et al. (2010*b*); NS; continuous measurement using flow-through steady-state chamber	Open (pilot) storage, batch filled, natural crust, and ice on surface, 158 d	0.0003	0
Wood et al. (2012); NS; continuous measurement using flow-through steady-state chamber	Open (pilot) storage, batch filled, natural crust on surface, 180 d	0.06	0.0012
N_2O emissions from solid cattle manure			
Brown et al. (2002); MB; continuous micrometeorological mass balance method	Manure with straw bedding, stockpile ~180 d	0.99	0.106
Pattey et al. (2005); ON; twice a week full-day measurement using flow-through chamber	Stockpile, 90 d	0.91	NC[b]

[a]AB, Alberta; MB, Manitoba; ON, Ontario; NS, Nova Scotia.
[b]NC, not calculated because the supporting data required for calculation were not provided in the original paper.

modifications to the existing manure handling system relative to modifications required for farms using solid manure handling. Moreover, AD systems are effective in generating C credits (Alberta Environment 2007) because the baseline GHG emissions for liquid manure systems are significantly higher than those for solid manure systems (IPCC 2006). In theory, implementing an AD system could reduce CH_4 emissions from dairy manure very effectively, as CH_4 emitted from manure is captured and destroyed in the process of generating energy. Many studies estimated substantial reductions of CH_4 emissions from manure management (about 80% or more reductions) associated with AD implementation on dairy farms (e.g., Kaparaju and Rintala 2011; Maranon et al. 2011; Battini et al. 2014). However, these studies are based on life cycle analysis (LCA) and mostly used parameters derived from a few short-term pilot-scale or laboratory-scale digesters. It is recommended that the evaluation of energy generation and verification of actual GHG mitigation potential are conducted for one year, after the start-up phase is completed (ASERTTI 2007). As an example, CH_4 emissions from stored digestate at a US dairy farm with 6200 Holstein cows were reduced by 47% compared with emissions from the reference manure management system estimated using the IPCC tier 2 model (Artrip et al. 2013). Studies involving municipal organic waste digesters have identified two major factors influencing the effectiveness of an AD system for GHG mitigation: uncontrolled losses of CH_4 from the anaerobic digester itself (fugitive CH_4 emissions) and CH_4 emissions from the stored digestate (Møller et al. 2009). Fugitive CH_4 emissions can occur (1) at times when the reactor is opened for maintenance, (2) leaking from valves and fittings, (3) intentional release of biogas through safety valves due to overpressure, (4) emissions associated with the use of biogas (burning in the engine or upgrading), and (5) inefficiency in the flare operation (Møller et al. 2009). Research-reporting measured fugitive GHG emissions are extremely rare because of the uncertainty associated with timing, location, and the rate of fugitive emissions (Møller et al. 2009). In one published study on a Canadian farm digester, Flesch et al. (2011) reported fugitive CH_4 emissions to be around 3% of the total CH_4 produced in the digester over two "6-d" periods in the autumn and winter. In contrast, protocols developed under the clean development mechanism (UNFCCC 2014), recommend a fugitive emission rate of 15% of the total CH_4 generated in an AD when calculating GHG reductions. Measurements taken over longer periods and under different operational conditions are necessary to ascertain the extent of fugitive GHG emissions from dairy farm digesters.

The other critical factor influencing the effectiveness of a farm anaerobic digester for GHG mitigation is the magnitude of emissions from stored digestate. Hydraulic retention time (HRT) during the AD process typically ranges from 20 to 30 d, and all VS are not degraded so only part of the CH_4 potential of the dairy manure is realized (Kaparaju and Rintala 2011). Once the effluent is transferred to storage, methanogenic microbes in partially digested slurry may continue to produce CH_4. The difference between the maximum and actual CH_4 generation in an AD can be viewed as an indication for the remaining CH_4 generation potential (Brown et al. 2008). As an example, Clemens et al. (2006) observed higher total emissions (CH_4, N_2O, and NH_3) from raw dairy slurry (90 kg CO_2 eq m^{-3}) as expected; however, digested dairy slurry (29-d HRT) also emitted a significant amount of GHG (40 kg CO_2 eq m^{-3}) over 140 d. Covering the storage tank and capturing the CH_4 released during digestate storage may therefore be an important factor to avoid digestate becoming a significant source of GHG in an AD system (Amon et al. 2006; Clemens et al. 2006; Gioelli et al. 2011).

Chemical changes during the digestion process such as increased NH_4^+ concentration and pH in the digestate relative to raw dairy manure have potential to increase NH_3 emissions during storage and NH_3 and N_2O emissions after land application. Thus, it is important to account for these implications when assessing the overall GHG mitigation due to AD; however, very few studies have been done on this aspect. Clemens et al. (2006) accounted for CH_4 and N_2O emissions from all stages of AD (digester, storage, and land application) showing an overall decrease of 46% relative to untreated manure, with increased N_2O emissions from the digestate application offsetting only 3% of the reductions achieved at digester and storage stages combined.

Although AD could be an effective way to reduce GHG emissions while also generating an additional income for dairy farms by means of renewable energy, its widespread adoption has not been satisfactory, largely due to high capital cost associated with digester construction and limited competitiveness of biogas with other fuels used for heat and power generation (Frear et al. 2011). Moreover, dairy manure as a single substrate has resulted in variable economic returns due to relatively conservative rates of biogas yield relative to other substrates (Atandi and Rahman 2012). Adding off-farm organic sources such as waste oil and fats from food processing industry can optimize C:N ratio in the dairy manure and increase the CH_4 yield (Frear et al. 2011; Atandi and Rahman 2012); however, it is highly critical that fugitive CH_4 emissions and emissions from digestate are prevented because these emissions can increase proportionally with increased CH_4 production in the digester with codigestion (Møller et al. 2009).

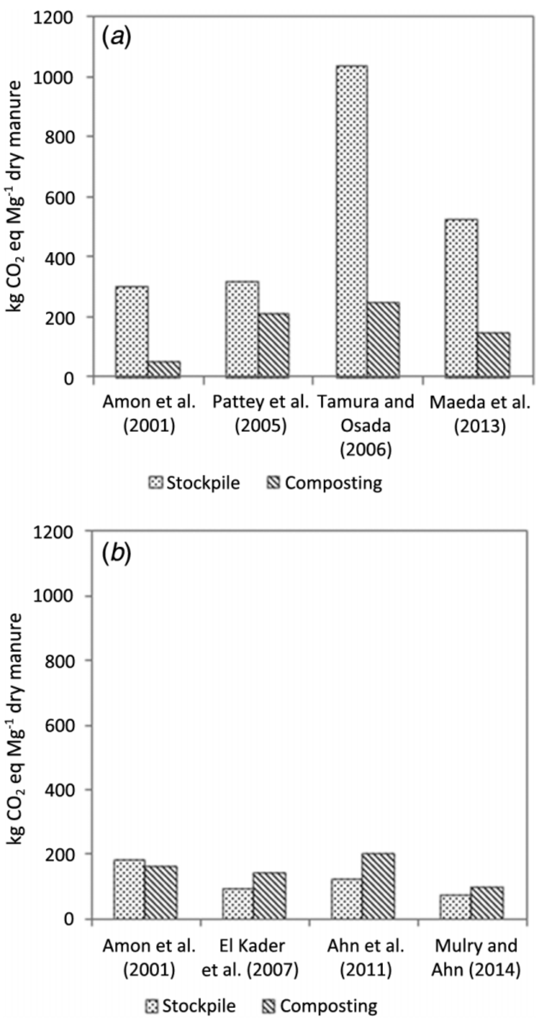
Composting

Composting is an aerobic process that transforms biological waste materials into a stable humus-like material through microbial decomposition and is generally regarded as an environmentally friendly waste management process (Brown et al. 2008). Because 20%–45% of dairy manure is managed as solid manure among different provinces of Canada, composting may be regarded as a viable GHG mitigation strategy for dairy farms. The composting process can be "active" with forced aeration or aeration supplied by frequent turning, or "passive", with only natural aeration provided by the "chimney effect" (Peigne and Girardin 2004). In some instances, passive aeration is enhanced by placing open-ended perforated PVC pipes at the bottom of the compost pile (Hao et al. 2001; Pattey et al. 2005). About 38% of farms across Canada used composting as a manure management method in 2001, but this may be an overestimate as many of the surveyed farms may have considered solid manure piles as composting (Statistics Canada 2004). Piling solid manure without intermittent turning to promote aeration is not considered as composting (Peigne and Girardin 2004). In the Canadian dairy sector, a 2005 survey reported that composting was explicitly done on 14%–30% of dairy farms with solid manure in the prairie Ecoregions and much less (2%–11%) in other Ecoregions (Sheppard et al. 2011*a*).

Greenhouse gases (CH_4 and N_2O) can be generated during composting as a result of microbial breakdown of organic matter. Oxygen can be depleted quickly within the composting pile creating heterogeneous anaerobic zones near the center of the pile and progressively increasing aerobic zones toward the surface (Hao et al. 2001). Nitrous oxide can be generated through nitrification of NH_4^+ in the aerobic zones, while both CH_4 and denitrification-based N_2O (when NO_3^- formed in aerobic zones diffuses to anaerobic zones) can be generated in anaerobic zones (Gilroyed et al. 2011). In some circumstances, methanogenic microbes within the center may be inhibited by high concentrations of NH_3 leading to reduced CH_4 formation (Brown et al. 2008). Moreover, some of the CH_4 generated in anaerobic zones can be consumed by aerobic microbes active near the surface. Therefore, GHG emissions during composting are the net result of the dynamic balance between GHG formation and consumption regulated by management practices such as turning and active aeration (Gilroyed et al. 2011).

The majority of studies evaluating GHG emissions from composting in Canada have focused on solid beef manure (e.g., Hao et al. 2001, 2004, 2011). The results from these studies have some applicability to dairy manure composting; however, they do not include reference manure storage for comparing change of emissions due to composting; therefore, the GHG mitigation potential cannot be determined. Hence, only studies on dairy manure composting were considered (Fig. 3).

Fig. 3. Comparison of greenhouse gas emissions (CH_4 and N_2O) from composting of solid dairy manure relative to emissions from stockpiled dairy manure (reference manure storage) (*a*) experiments involving warm season composting, (*b*) experiments involving cold season composting. Composting pile aeration provided by frequent turning except in Pattey et al. (2005) in which passive aeration provided by perforated steel pipes placed at the bottom of the pile.



Two important trends were clear from the results from these studies. First, GHG emissions from stockpiled dairy manure were mitigated substantially by composting (72%–83% reduction of total GHG) relative to emissions from the stockpiled dairy manure during the warm season (Fig. 3*a*). Second, management practices such as timing and frequency of turning appeared to be very critical for composting during the winter season, because mixed results for GHG mitigation have been observed for winter season composting: one study indicating a modest mitigation of GHG (about 10% reduction relative to reference manure storage, Amon et al. 2001) while two other studies (El Kader et al. 2007; Ahn et al. 2011) indicated increased GHG emissions from composting (Fig. 3*b*). Ahn et al. (2011) observed that cold and wet conditions can adversely affect the performance of the composting process, because aerobic microbial community can be subjected to cold shock during cold season composting. However, as the baseline GHG emissions (emissions from the reference system) are low in the cold season, the overall impact of a small increase of GHG due to composting in the cold season is minimal, because larger reductions in GHG emissions can be achieved in the warm season. Based on these results, composting can be considered as an effective strategy for mitigating GHG emissions from solid dairy manure in cold climates, but a significant loss of N, mostly as NH_3 and loss of C that reduce the fertilizer value of manure (Hao et al. 2004) should also be assessed before final recommendations can be made.

Reducing the methanogen inoculum by complete emptying of the storage

When manure is utilized for application to crops, complete emptying or cleaning of manure storage is not usually practiced, and some old manure may be left in the storage while it is being refilled with new manure (Sommer et al. 2007; VanderZaag et al. 2011*b*; Wood et al. 2014). The old manure left in the storage can act as an inoculum facilitating the rapid re-establishment of the methanogen population and CH_4 emissions. Previous studies have noted 2–5-mo long lag phases of low CH_4 emissions before the onset of increased CH_4 fluxes, when manure was loaded into clean storage vessels (Massé et al. 2008), or pilot-scale structures (VanderZaag et al. 2008, 2009, 2010*b*; Wood et al. 2012). This lag phase may be associated with the time required for methanogen and syntropic anaerobic microbial communities to establish. Massé et al. (2008) used modeling to extend the results of laboratory incubations and recommended more frequent and complete emptying of manure storages as an effective management practice to reduce CH_4 emissions. Using pilot-scale concrete manure storages, Wood et al. (2014) evaluated the impact of complete vs. partial (50%) emptying of manure storage on CH_4 and N_2O emissions from dairy manure over a 155-d-long summer storage period in Nova Scotia, Canada, and observed about 50% lower total GHG emissions (CH_4 and N_2O) from the completely emptied manure storage compared with emissions from the partially emptied storage. As the baseline CH_4 emissions over the summer manure storage period are high, this strategy appears especially effective if complete emptying of storage is done at the time of spring manure application (generally in April or early May in Canada). It is not clear to what extent the storage should be emptied to prevent old manure from becoming an effective inoculum. In another pilot level study, Sommer et al. (2007) noted that the presence of a small quantity of inoculum (~8%) could support the immediate production of appreciable amounts of CH_4 from manure within 10 d after refilling the storage with new manure, indicating that the storage should be emptied nearly completely to prevent old manure becoming an effective inoculum. Further research is necessary especially at the farm level, for fine tuning this strategy and also to evaluate the practical implications in terms of time and land available for manure application.

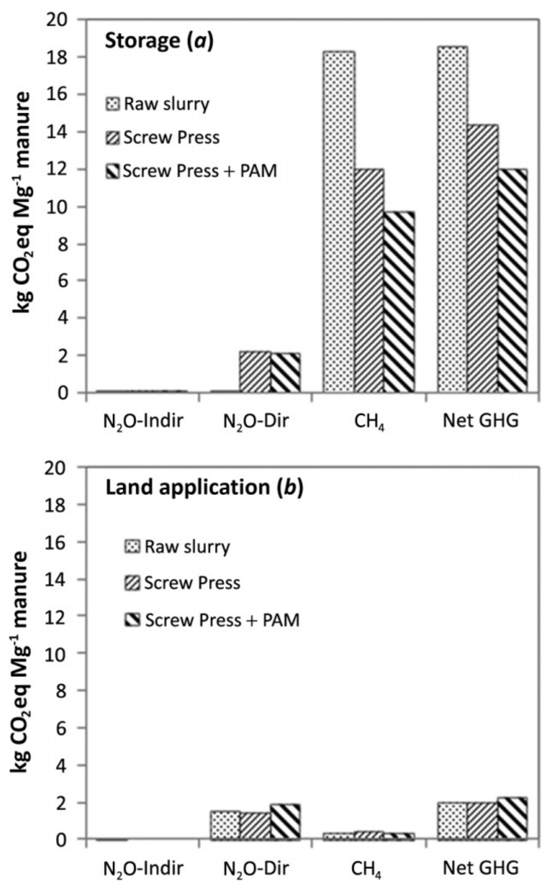
Solid–liquid separation

Solid–liquid separation is a process that results in a solid fraction rich in DM and nutrients, and a liquid fraction relatively low in both DM and nutrients (Møller et al. 2000). Solid–liquid separation has additional benefits with regard to a farm's nutrient management plan. For example, (1) the nutrient-rich solid fraction can be transported greater distances (to be used as fertilizer) at a relatively low cost (compared with transporting untreated manure slurry), which may help avoid nutrient overloading on crop lands within the farm, and (2) solid separation could potentially increase the storage capacity for the liquid fraction (Møller et al. 2000). Several different systems for separating manure slurry into solid and liquid fractions are available, and their performance under practical farm conditions has been reviewed recently by Hjorth et al. (2010).

Methane and N_2O emissions from the separated solid and liquid fractions during storage can behave in a different manner. For example, total N_2O emissions during the storage of separated solid and liquid fractions increased by over 1100% compared with N_2O emissions from the stored untreated dairy slurry (Fangueiro et al. 2008*a*). This increase is mainly from the higher N_2O emissions from the solid fraction due to the existence of aerobic/anaerobic zones in the stored solids, similar to conditions in solid manure piles (Fangueiro et al. 2008*a*). In contrast, the total CH_4 emissions from the two fractions (separated solids and liquids) were reduced by about 34% relative to the emissions of CH_4 from the untreated slurry (Fangueiro et al. 2008*a*). Because CH_4 constitutes close to 90% of the total GHG when expressed in CO_2 equivalents, a net decrease of about 23% of the sum of two gases ($N_2O + CH_4$) can be achieved by solid–liquid separation when compared with emissions from untreated dairy slurry (Fig. 4*a*). Although a significant increase in NH_3 emissions from the stored solid fraction has been observed relative to emissions from untreated dairy slurry, its contribution through indirect N_2O emissions [calculated using IPCC (2006) default emissions factor] to the total GHG budget was less than 2% (Amon et al. 2006; Fangueiro et al. 2008*a*). In another study, only a modest mitigation (<10%) of CH_4 emissions under warm ambient temperature (25 °C) and slightly higher (~4%) emissions from the separated fractions at colder temperature (5 °C) were observed (Dinuccio et al. 2008), but reasons for this were unclear.

Separation efficiency seems to be an important factor influencing the effectiveness of solid–liquid separation

Fig. 4. Comparison of greenhouse gas emissions after solid–liquid separation of dairy manure with emissions from untreated dairy manure during a 48-d storage (*a*) and after land application (*b*). Two methods of solid–liquid separation are shown (screw-press separation and screw press combined with flocculation by polyacrylamide (PAM). Abbreviations: N_2O-Indir: indirect N_2O emissions due to NH_3 emissions, N_2O-Dir: direct N_2O emissions, CH_4: methane emissions, Net GHG: $N_2O + CH_4$. Indirect N_2O emissions were calculated as per IPCC (2006) on the basis of measured NH_3 emitted in the study (data source: Fangueiro et al. 2008*a*, 2008*b*).



on overall GHG mitigation. For example, CH_4 mitigation due to solid–liquid separation was amplified with improved separation efficiency achieved by combining mechanical separation with chemically induced settling of suspended solids (enhanced removal of VS from the liquid phase) to obtain a supernatant liquid fraction (Fig. 4*a*).

Following the application of separated solid and liquid dairy manure fractions to soil, total N_2O emissions were 20%–25% higher than the N_2O emissions from untreated dairy slurry application (Amon et al. 2006; Fangueiro et al. 2008*b*). However, when total GHG emissions (CH_4 and N_2O) from the storage phase and soil application were combined, net GHG emissions from separated solid and liquid fractions were still 20%–37% lower than that from untreated dairy manure (Fig. 4*b*). This implies that,

Table 6. Summary of research results that studied the effect of manure storage covers on greenhouse gas emissions from stored dairy manure.

References	Treatment	CH_4	N_2O	$CH_4 + N_2O$	% change
		kg CO_2 eq m^{-3} manure			
Clemens et al. (2006); 100 d, winter	Raw slurry + no cover	4.1	13.4	17.5	
	Raw slurry + wooden cover	3.6	11.6	15.0	–14
	Digested slurry + no cover	2.8	12.2	15.0	
	Digested slurry + straw cover	2.9	12.1	15.0	No change
	Digested slurry + wooden cover	2.0	12.3	14.3	–4
Clemens et al. (2006); 140 d, summer	Raw slurry + no cover	89.9	14.9	104.7	
	Raw slurry + wooden cover	75.0	17.7	92.7	–12
	Digested slurry + no cover	28.9	22.4	51.3	
	Digested slurry + straw cover	29.8	23.0	52.8	+3
	Digested slurry + wooden cover	25.5	18.6	44.1	–14
Amon et al. (2006); 80 d, summer	Raw slurry + straw cover[a]	122.6	12.8	135.4	
	Raw slurry + wooden cover	101.1	6.2	107.3	–21
VanderZaag et al. (2009); 121 d, summer	Raw slurry + no cover	85.2	2.0	87.1	
	Raw slurry + 15 cm straw cover	63.5	2.6	66.1	–24
	Raw slurry + 30 cm straw cover	61.5	3.2	64.8	–26
VanderZaag et al. (2010*a*, 2010*b*); 165 d, summer	Raw slurry + no cover	77.6	3.5	81.1	
	Raw slurry + permeable synthetic cover	79.4	1.0	80.5	–1

[a]The "raw manure + no cover" control treatment was not included in this study.

by implementing additional mitigation strategies for reducing soil emissions, further reductions of net GHG emissions could be potentially achieved with solid–liquid separation of dairy manure.

Manure storage covers for GHG mitigation

This strategy involves creating a barrier over the free surface of liquid slurry by covering it with either fixed structures or floating material. Early research on covering manure storages has mainly focused on mitigating odors and NH_3 emissions from manure (see the review by VanderZaag et al. 2008). However, several recent experiments evaluating the effect of various cover types on CH_4 and N_2O emissions from dairy manure have indicated appreciable reductions of these two gases combined (up to 26% reduction) compared with emissions from uncovered slurry (Table 6).

Two types of covers can be used in manure storages: (1) natural covers and (2) artificial covers. Natural covers can be the "crust" that formed naturally on the surface of liquid manure due to coalescing of larger solid particles in manure or a layer of natural material (e.g., straw or wood chips) purposely added to the surface of storage to encourage the crust formation. Artificial covers can be impermeable or permeable and both natural and artificial permeable covers allow water and gas molecules to pass through the cover (VanderZaag et al. 2008). Moreover, permeable covers can be colonized by a diverse community of microorganisms and can create differential anaerobic and aerobic zones within which microbial processes such as NH_3 and CH_4 oxidation, and denitrification can take place (Petersen and Miller 2006; Nielsen et al. 2013). As a result, N_2O emissions from liquid manure can be increased relative to those from uncovered liquid manure, while CH_4 emissions can be decreased (Petersen and Miller 2006). The balance of these two opposing processes determines the magnitude of the net change of GHG emissions, and most of the available studies indicate that covering liquid manure with natural covers results in an overall reduction in GHG emissions, relative to emissions from uncovered liquid manure (Table 6). However, the results summarized in Table 6 indicate that the effectiveness of permeable covers/crust in mitigating GHG emissions from liquid manure is not always consistent. Several other factors such as rapid degradation of cover material when in contact with manure, possible settling of cover material to the bottom of storage, wind drifting affecting the coverage of material on manure surface (Guarino et al. 2006), and a mismatch between the period of peak CH_4 oxidation in the crust and period of peak CH_4 production from manure (Nielsen et al. 2013), may contribute to this inconsistency.

In contrast with permeable covers, impermeable covers can predictably prevent GHG emissions from dairy manure slurry storages, provided that they are properly installed and trapped CH_4 is captured and destroyed (Stenglein et al. 2011). Impermeable covers can be built using concrete and wood but typically covers made of plastic are widely used (English and Fleming 2006; Stenglein et al. 2011). Plastic impermeable covers can sit on the surface of manure (floating), have a longer life span and can capture most gases emitted from manure if installed properly (Stenglein et al. 2011). Captured gas may be treated using gas-phase biofilters, flared, or combusted to generate useful energy (Stenglein et al. 2011). If ensured that no gas leaks from the cover, their effectiveness in GHG mitigation depends on the destruction or combustion efficiency of the flare or device used for gas destruction (USEPA 2008).

Conclusions

A substantial amount of research on enteric CH_4 emissions from dairy cows has been conducted in Canada during the last decade (2005–2014), improving our understanding of potential mitigation strategies for reducing emissions from high producing dairy cows. This review examined effective mitigation options available for reducing enteric CH_4 emissions, options other than the emission intensity reductions achievable by improved herd management or genetic improvement in milk production. In this regard, a few practical strategies are available that can be applied currently without affecting milk productivity. These readily applicable strategies include manipulation of forage type and forage quality by increased use of whole plant corn silage with high starch content, and feeding rations supplemented with lipid-rich by-products (e.g., CDDGS). It may be possible to decrease enteric CH_4 yield by up to about 15% with these strategies; however, possible trade-offs from increased CH_4 emissions from manure and implications related to increased production of whole corn silage in place of other silage crops need to be evaluated at the whole farm scale. Two other potentially promising strategies that were identified include nitrate supplementation and administration of specific inhibitors of rumen methanogenesis such as 3NO. However, these strategies are still at early stages of development and require extensive future research on their efficacy and consistency for reducing enteric CH_4 emissions as well as issues related to toxicology and consumer acceptance.

With continued consolidation of dairy farms in Canada, increasingly larger proportion of dairy manure is managed using liquid slurry systems. This change inevitably leads to increased CH_4 contributions from dairy manure to whole farm GHG emissions. However, surprisingly few studies quantifying dairy manure GHG emissions have been published, especially studies involving on-farm measurements over a complete year. This lack of year-round data has significantly hampered the accurate estimation of baseline GHG emissions and verification of the recommended empirical model (IPCC 2006, tier 2 model), as well as accurate assessment of mitigation strategies. This is a major knowledge gap. Further research is required, both in terms of establishing baseline GHG emissions from dairy manure management and verifying estimated emissions. Year-round on-farm measurements are also required for improving and verifying process-based models that are capable of describing various biochemical pathways involved in GHG generation from stored dairy manure.

Based on pilot-scale Canadian research as well as several farm-level studies from other cold climates with similar intensive dairy production systems, several promising strategies for mitigating GHG emissions from dairy manure were identified. These strategies include AD, solid–liquid separation, composting, manure storage covers, and complete emptying of liquid manure storage at spring application. With implementation of these strategies, it may be possible to achieve up to a 50% reduction in CH_4 and N_2O emissions from dairy manure; however, these strategies need to be evaluated using on-farm research for their practical and economic feasibility of adoption.

It is clear that substantial reductions in GHG emissions from milk production may not be possible by applying one strategy alone. An integrated approach of combining several strategies both at the feeding stage and in the manure management chain could result in significant reductions in GHG emissions leading to potential reductions in C footprint of milk produced in Canada.

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