FARMERS FOR CLIMATE SOLUTIONS

Technical Emissions Report:

Agricultural Policy Framework (APF) Recommendation

Contents:

** Please note that not all of the beneficial management practices (BMPs) considered in this technical report are included in the list of recommended BMPs in the summary report. The Task Force examined a number of BMPs that are not recommended for incentivization in the short term because they were found to be less cost effective, more difficult to incentivize or less effective than the recommended BMPs.*

Improved Nitrogen Management

EnviroTerra Consulting

for

FARMERS FOR CLIMATE SOLUTIONS

February 2022

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1. Introduction

The Canadian government has made clear its commitment to reducing greenhouse gas emissions in A Healthy Environment and A Healthy Economy. In particular, the government has identified a target of reducing absolute levels of greenhouse gas (GHG) emissions arising from nitrogen fertilizer by 30% below 2020 levels by 2030. This represents an approximate 4 Mt CO_2e emissions reduction associated with improved nitrogen management. In their March 4 discussion document *Reducing Emissions Arising from the Application of Fertilizer in Canada's Agriculture Sector*, Agriculture and Agri-Food Canada identified 11 beneficial management practices (BMPs) relating nitrogen and conservation management that could result in the desired emissions reduction. The analysis presented here parallels that analysis providing detail on the implementation of BMPs through policy measures in the upcoming Agriculture Policy Framework (APF).

The AAFC discussion document argues that a 30% reduction is doable, and without significant reduction in yield if improved nitrogen management is used to increase the efficiency of nitrogen use from all sources and rates of fertilizer nitrogen reduced accordingly. This program is not in conflict with 4R Nutrient Stewardship or 4R Climate Smart Protocol being promoted by Fertilizer Canada. One of the primary tenants of 4R nitrogen management is to determine *Right Rate* by quantify all sources of nitrogen and consider realistic yield goals.

Setting the scene, how much N do we use?

As a first step it is useful to examine temporal and spatial trends in N use in agriculture in Canada. For this analysis we are relying on data reported to the UN Food and Agriculture Organization (UNFOSTAT) and Canada's National Inventory Report (NIR). For this report, Environment and Climate Change Canada (ECCC) had provided us with a breakdown of the N use information for 2019 that is used in building the NIR reports. The amount of N used in agriculture in Canada is increasing (Fig. 1). Synthetic N fertilizer is the dominant source of N additions to agriculture and, according to data reported to the FAO represents 67% of all N inputs.

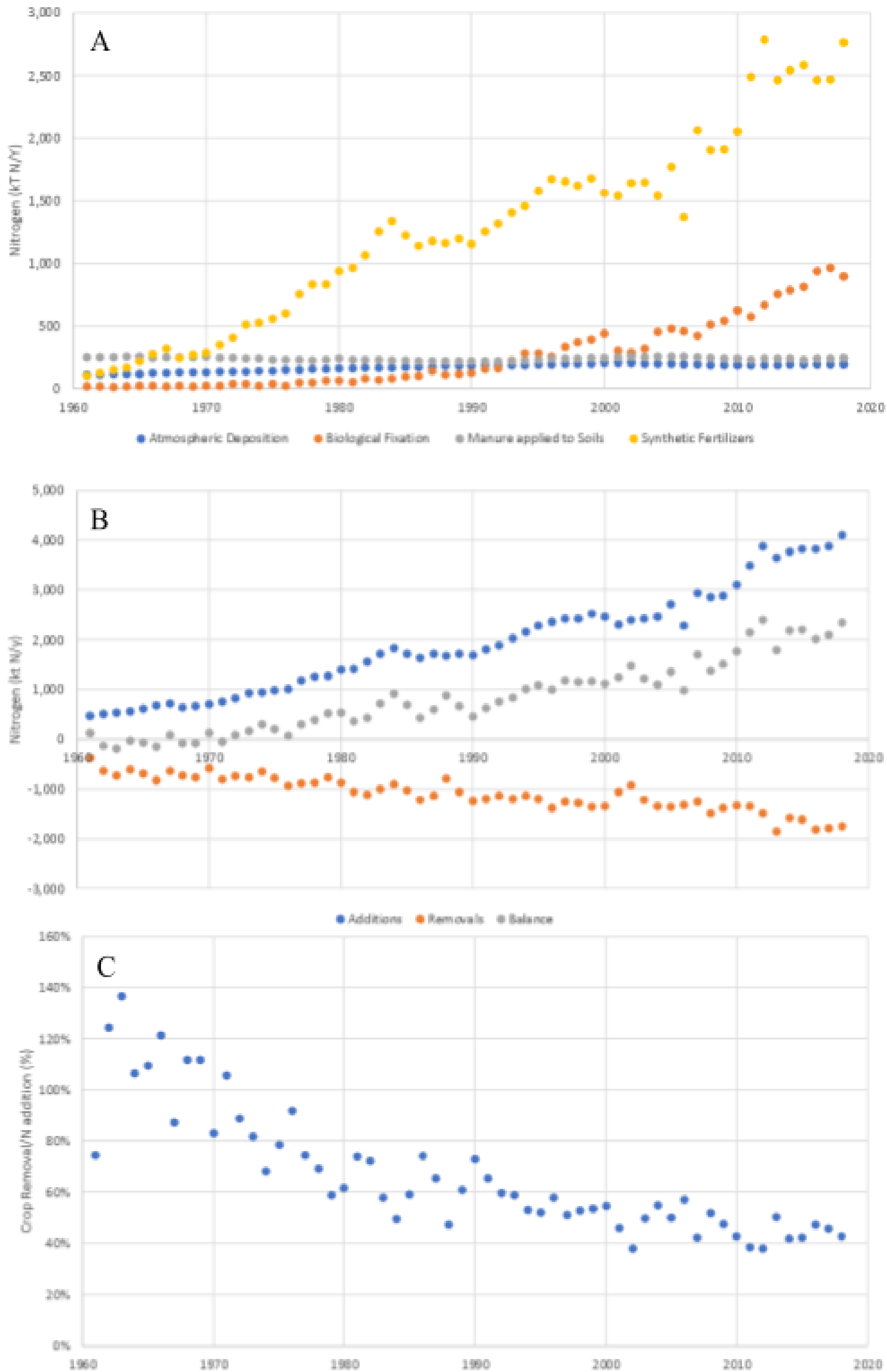


*Figure SEQ Figure * ARABIC 1: Annual nitrogen additions from atmospheric deposition, biological N fixation, manure N and fertilizer N (A), balance between N additions and removals in crop harvest (B), and crop N removals as a percentage of N additions (C) for Canadian agriculture from 1961 to 2019 (FAOSTAT 2022).*

2. Methods for estimating Emissions

Manure Management (NIR Category 520)

Description of emissions sources and mechanisms

The United Nations Food and Agriculture Organization Statistical Branch (FAOSTAT) reports that in 2019 a total of 742 kT N was excreted by the cattle, chickens, and swine. Beef dominates (~60%) the production of manure N in Canada with the role of dairy declining over the past number of decades such that it is now equivalent to chicken and swine, each at approximately 10-15% (Fig. 2A).

How manure management is measured and reported in the National Inventory Report

The handling, storage, and land application of animal manure results in both methane (CH_4) and N_2O emissions. Methane emissions are largely associated with the storage of manure. Nitrous oxide emissions occur both during manure management (handling and storage) and following land application. Direct emissions of N_2O during manure management are reported as emissions from Manure Management, NIR category 520, also referred to as Animal Waste Management Systems (AWMS) emissions. Indirect N_2O emissions associated with ammonia (NH_3) loss from storage and leaching and run off from storage are also estimated based on animal species, storage system and ecozone.

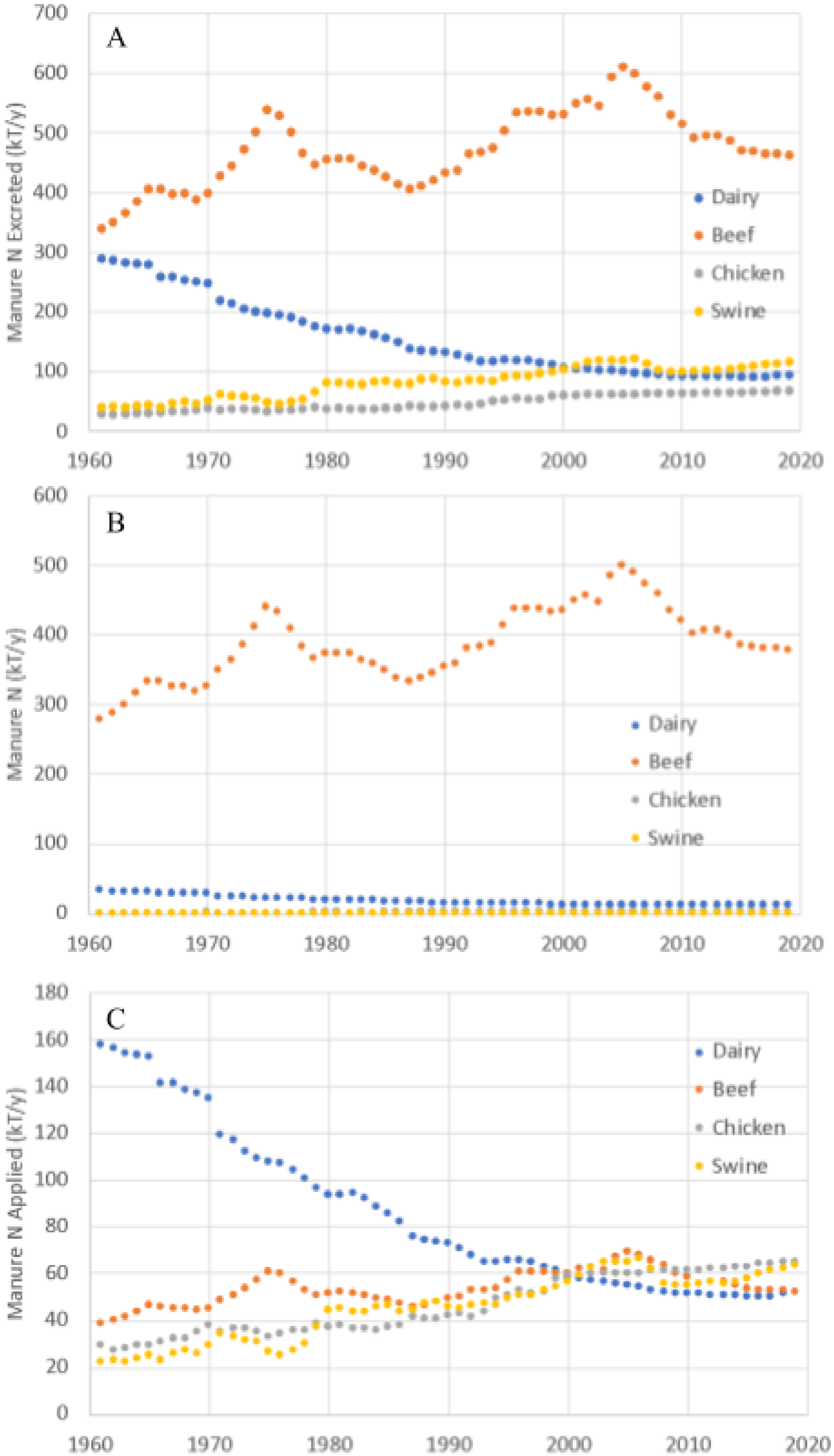


*Figure SEQ Figure * ARABIC 2: The amount of manure N (kT N/y) as excreted (A), deposited on pasture (B), and applied to soils (C)in Canada from 1961 to 2019 by animal species (FAOSTAT, 2022).*

In Canada's NIR, the emissions of CH_4 and N_2O from manure management are calculated based on animal numbers and a combination of Tier 1 and Tier 2 data based on animal species. Four animal species groups are identified: diary, beef (non-dairy cattle), poultry and swine (Fig. 3).

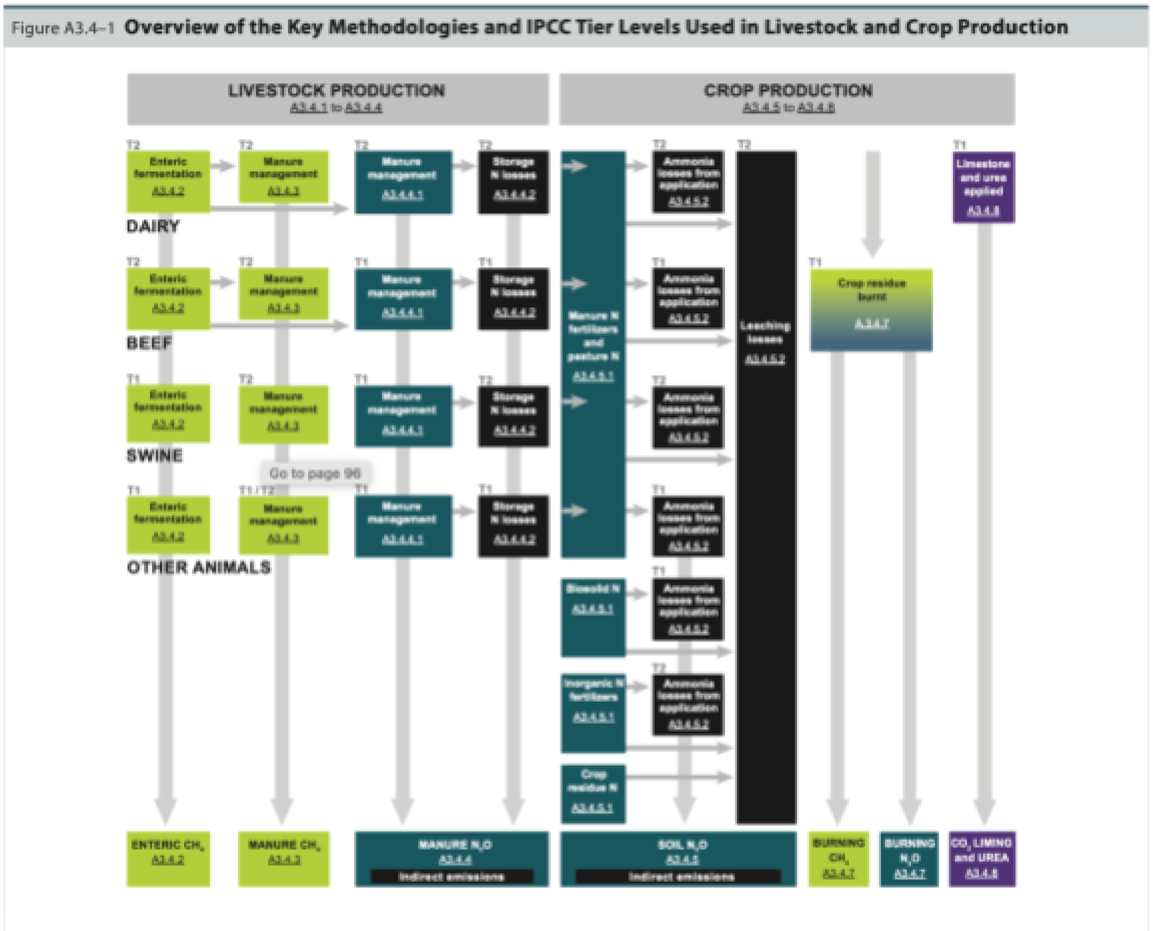


*Figure SEQ Figure * ARABIC 3: Process by which greenhouse gas emissions associated with animal production are reported in Canada's National Inventory Report (NIR, 2019).*

Quantities of emissions under consideration

N_2O emissions from Manure Management (AWMS)

The N_2O emissions from the management of animal manure is estimated using the IPCC Tier 1 method that considers: i) N excretion rates for various animal categories and subcategories, ii) types of animal waste management system (AWMS), and iii) emission factors associated with manure management systems.

Nitrous oxide emissions from manure management have declined by 20% over from a high of 4.9 Mt CO_2e/y in 2005 to 4.0 Mt CO_2e/y in 2019 (NIR, 2020; Fig. 4). Emissions are greatest where animal numbers are greatest (Fig. 5). The provinces of

Alberta, Ontario, Saskatchewan, Quebec, Manitoba, and British Columbia account for 90% of the N_2O emissions from manure management (Fig. 5). Emissions from animal manure applied to soil (0.5 Mt CO_2e/y) and deposited directly on pasture, range, and paddock (0.1 Mt CO_2e/y) are significantly less than those associated with manure storage and handling (Fig. 5).

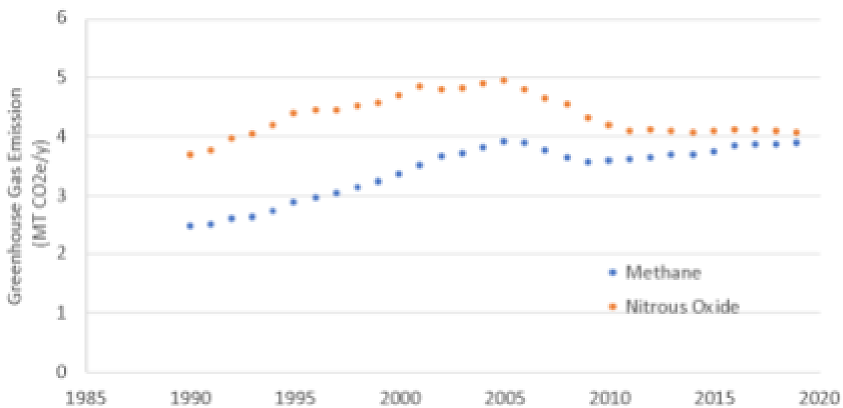


*Figure SEQ Figure * ARABIC 4: Nitrous oxide and methane emissions from animal manure management in Canada from 1990 to 2019 (NIR, 2020).*

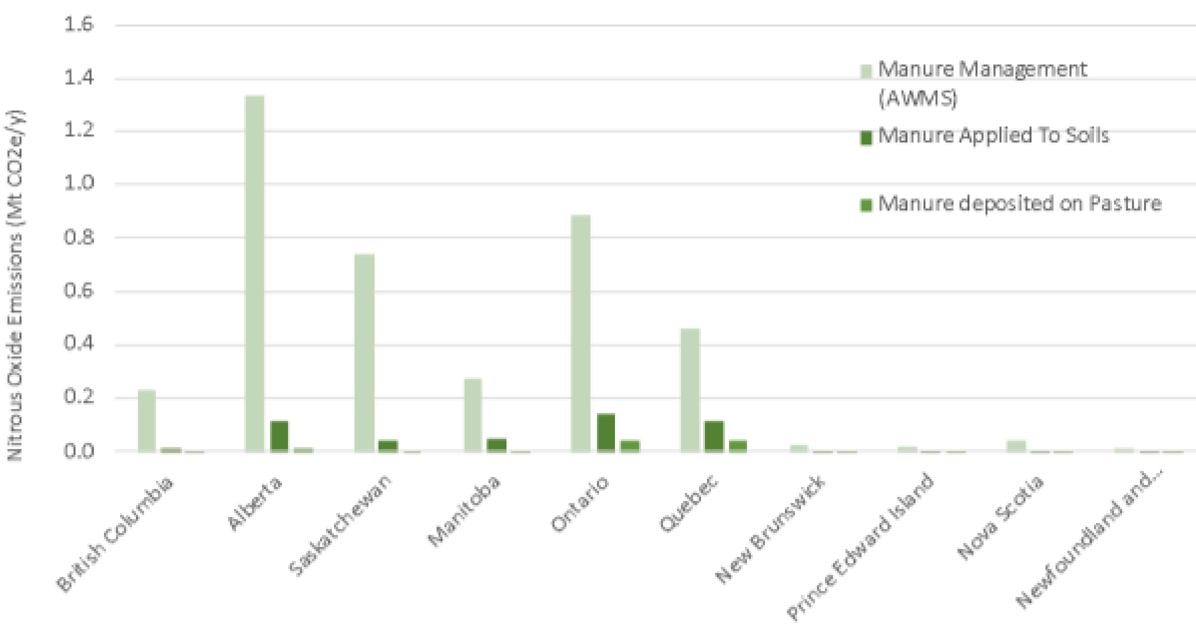


*Figure SEQ Figure * ARABIC 5: Nitrous oxide emissions (Mt CO2e/y) from manure management (AWMS), manure applied to soil, and manure deposited on pasture, range, and paddocks in 2019 (NIR, 2019)*

Nitrous Oxide Emissions from Agricultural Soils (NIR 530)

Description of emissions sources and mechanisms

The addition of nitrogen to soil results in direct and indirect production of N_2O. Direct emissions of N_2O from agricultural soils result from the production of N_2O in the soil during the processes of nitrification and denitrification that are induced by the addition of inorganic N fertilizers, organic N fertilizers (composts, manures, biosolids), urine and dung deposited on pasture, range and paddock (PRP) by grazing animals, crop residues decomposition, mineralization of nitrogen associated with loss of soil organic matter and cultivation of organic soils.

A total of 3.5 Mt N was added to agricultural soils in Canada in 2019 (NIR, 2020). Synthetic N fertilizers represented the majority (76%) of the nitrogen added to agricultural soils in 2019 (NIR, 2020). Other sources included animal manure applied to soil (12%), animal manure deposited on pasture, range, and paddocks (8%) and the mineralization of nitrogen in agricultural soils (5%). The addition of nitrogen to agricultural soils varied by province (Fig. 6), with the Prairie provinces accounting for 77% of the total N added to agricultural soils and Ontario/Quebec accounting for an additional 19%.

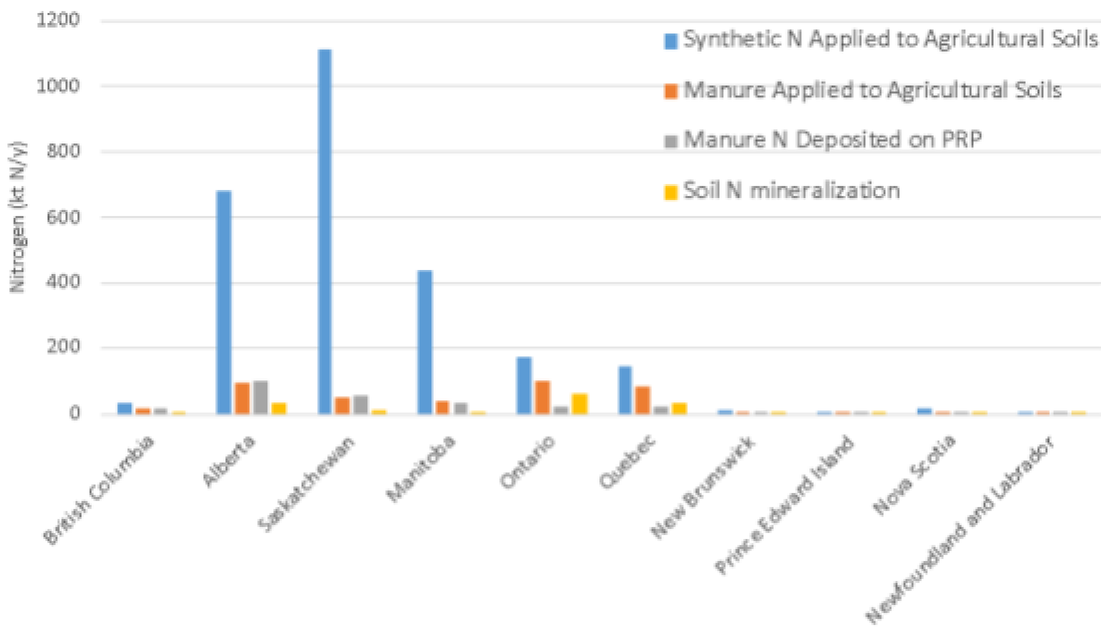


*Figure SEQ Figure * ARABIC 6: Nitrogen applied to agricultural soils and pasture, range, and paddocks in Canada in 2019 (NIR, 2020)*

How N_2O emissions from agricultural soils are measured and reported in the National Inventory Report

In our national inventory, N_2O emissions are not measured directly but rather are estimated based on N inputs. The basic calculation procedures are a Tier 2 method (Rochette *et al.*, 2008) as implemented in Canada's National Inventory Report (Environment and Climate Change Canada, 2020). The method accounts for effect of regional differences in the climate to alter direct emission factors with further modifications based on soil texture, tillage, topography, and fallow. In the most recent (2022) NIR the Tier 2 method has been updated according to the procedures described in Liang et al. 2020. The revised approach still considers soil texture, topography and climate but is based on a revised quantitative empirical relationship between synthetic N-induced soil N_2O emission factor and growing season precipitation (P). The revised approach also differentiates soil N_2O emission factors based on management factors. Specifically, empirical ratio factors are applied for sources of N of 1.0, 0.84, and 0.28 for synthetic N, animal manure N and crop residue N, respectively. Crop type ratio factors where soil N_2O EFs from applied manure- and synthetic-N on perennial crops were substantially reduced (19%) relative to those of annual crops.

This update has resulted in a recalculation of N_2O emissions primarily because of reduced emission factors for mainly dry areas of the prairies and for the application of nitrogen to perennial lands (-5,840 kt CO_2e/y). Smaller changes also resulted from (i) the implementation of the climate-specific emission factor from the 2019 Refinement to the 2006 IPCC Guidelines (-89 kt CO_2e/y); (ii) a correction to inorganic N fertilizer activity data for 2019 (-361 kt CO_2e/y); (iii) a correction to the IPCC default nitrogen excretion rates for swine (-12 kt CO_2e/y); and (iv) the implementation of methodologies to estimate changes in SOC impacted by crop productivity changes and manure application (-195 kt CO_2e/y). The inventory does not currently capture emissions reductions resulting from differences in N management, but the 2022 NIR states that "Canada plans to develop more robust ratio factors to account for these mitigation measures in the medium-term of three to 5 years as research results and activity data become available".

Indirect emissions of N_2O result from nitrogen compounds that are transported from agricultural soils, most notably through NH_3 volatilization and NO_3^- leaching. The production of N_2O occurs in the environment in which the N is deposited such as an

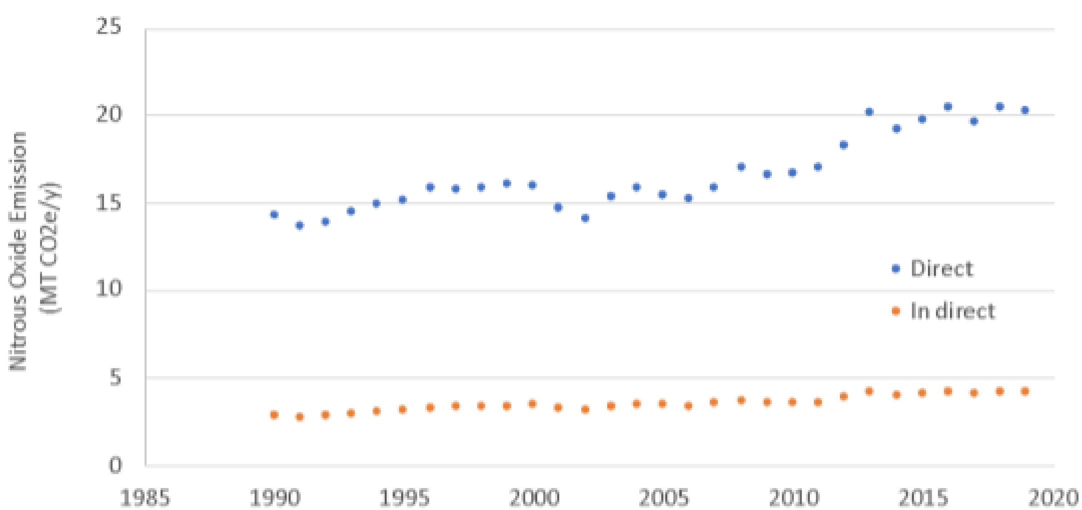


*Figure SEQ Figure * ARABIC 7: Nitrous oxide emissions from soil in Canada from 1990 to 2019 as a result of direct (inorganic N fertilizers, organic N fertilizers, urine and dung deposited on pasture, range and paddock by grazing animals, crop residues, mineralization associated with loss of soil organic matter and cultivation of organic soils) and indirect (volatilization and redeposition of inorganic and organic N) emissions (NIR, 2020).*

adjacent field or waterbody. Indirect N_2O emissions are estimated using at Tier 1 process, by first estimating the magnitude of N lost via volatilization or leaching, reflecting regional differences in these values, followed by the application of a regionally specific emission factor. Indirect emissions of N_2O are typically ~20% of direct emissions (Fig 7).

For this analysis, N input data was provided at the ecodistrict level for 2019 as reported prepared for the 2019 National Inventory Report as provided by Environment and Climate Change Canada. These estimates of N_2O emissions assume a linear relationship between N fertilizer application rate and N_2O emissions. At higher rates of N fertilization this approach may underestimate N_2O emissions as it is well documented that the N_2O emission factor increases as the rate of N fertilizer application exceeds plant N demand (Eagle et al., 2017; Van Groenigen et al., 2010).

Quantities of emissions under consideration

Synthetic Nitrogen Fertilizer

The majority (85%) of synthetic N fertilizer use occurs on the Prairies with an addition 12% being used in Ontario and Quebec (Figs. 8 & 9). The use of synthetic N fertilizer has increased linearly since 1980 and if this trend continues will be 3.5 Mt N /y in 2025 and 3.8 Mt N/y in 2030 (Fig. 8). The use of synthetic N fertilizer resulted in direct N_2O emissions equivalent to 10.56 Mt CO_2e/y in 2019. Of this total 7.65 Mt CO_2e/y (72%) occurred in the Prairie provinces and 2.48 Mt CO_2e/y (24%) occurred in Ontario and Quebec.

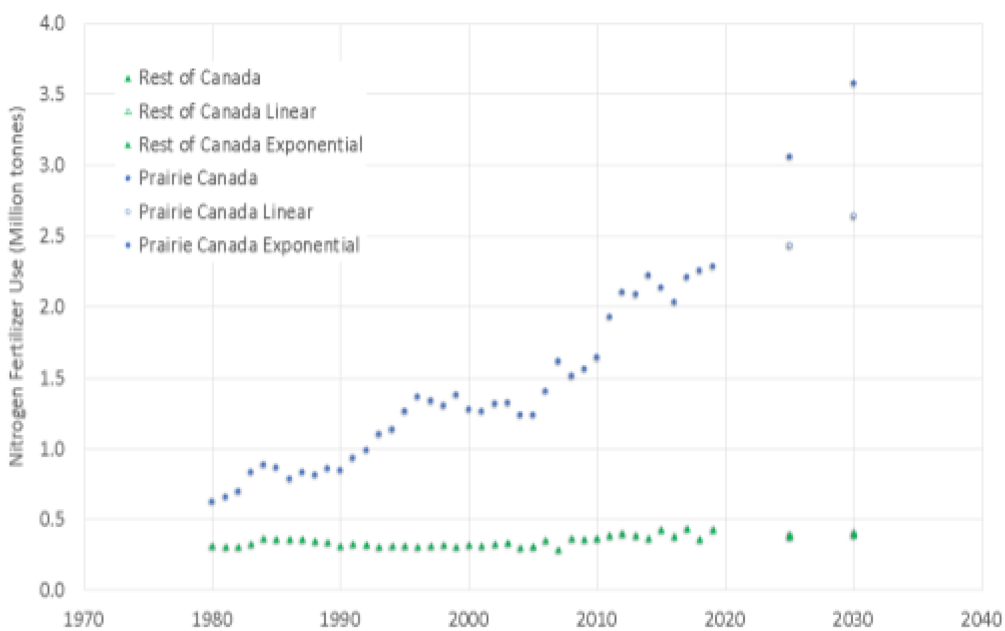


*Figure SEQ Figure * ARABIC 8: Historical trends in N fertilizer use in Prairie Canada and the rest of Canada in millions of tonnes. Prediction of N fertilizer use in each region according to a linear (open symbols) and exponential (closed symbols) curve fit.*

The use of synthetic nitrogen (N) fertilizer also results in indirect emissions of 2.6 Mt CO_2e/y associated with N lost from the site and contain embedded emissions associated with the production and transport of nitrogen fertilizer. Here we assume an emissions coefficient of 2.64 tonnes of CO_2e per tonne of N (Qualman and NFU, 2022) to estimate embedded emissions of 7.0 Mt CO_2e/y (Fig. 9). The total GHG emissions associated with the application of synthetic N fertilizer to agricultural land were 20 Mt CO_2e/y, with the prairie provinces accounting for 75% of those emissions.

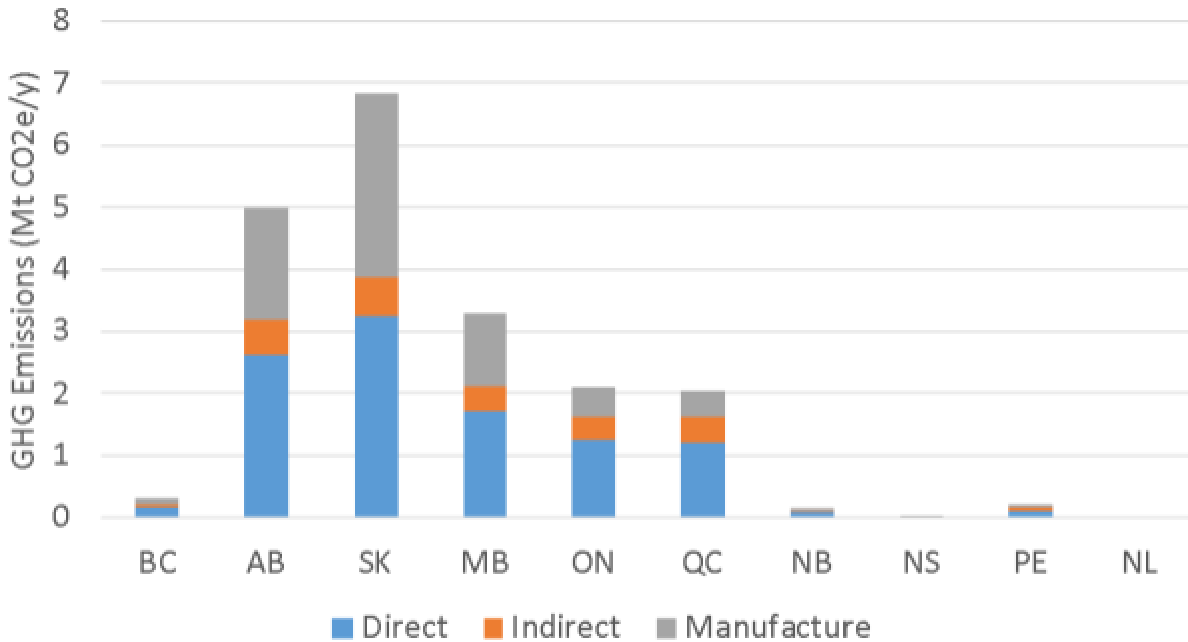


*Figure SEQ Figure * ARABIC 9: Greenhouse gas emissions (Mt CO_2e/y) associated with synthetic N fertilizer use in Canada, by province, in 2019. Emissions include direct emissions from soil, indirect emissions associated with ammonia volatilization and nitrate leaching and greenhouse gas emissions associated with the manufacture of N fertilizer.*

Manure applied to agricultural soils

Direct and indirect N_2O emissions are estimated for manure applied to land from a manure storage (AWMS) as well as manure deposited on pasture, range, or paddock (PRP). Manure from beef production is the dominant (97%) form of manure deposited on PRP (Fig. 2B) and represents the primary (81%) of beef manure. In terms of AWMS manure, that is manure not deposited on PRP and is thus applied to agricultural soils, the four major animal species make equal contributions at approximately 25% (Fig. 2C).

In the NIR the fate of nitrogen in land applied manure is tracked throughout the process of crop production and ammonia losses after application to croplands using Tier 2 level calculations for manure nitrogen from dairy and swine, with Tier 1 IPCC default loss factors being used for all other animals). Indirect emissions of N_2O from nitrogen that is lost from the agriculture system are estimated using Tier 1 IPCC 2006 emission factors. The rate manure application to land is estimated as a function of the total manure N produced in an ecodistrict and the crop requirements of that ecodistrict, which are then adjusted to provincial fertilizer sales as outlined in section (NIR, 2019).

Canada reported[1] 406 kt of AWMS manure N applied to agricultural land in 2019 in the NIR, with 70% being produced in the provinces of Ontario, Alberta, and Quebec (Fig. 10A). In Alberta much of the manure applied to soil is handled as a solid, in Quebec manure is handled primarily as a liquid, in Ontario liquid and solid manure handling systems are in equal proportion (Fig. 10B).

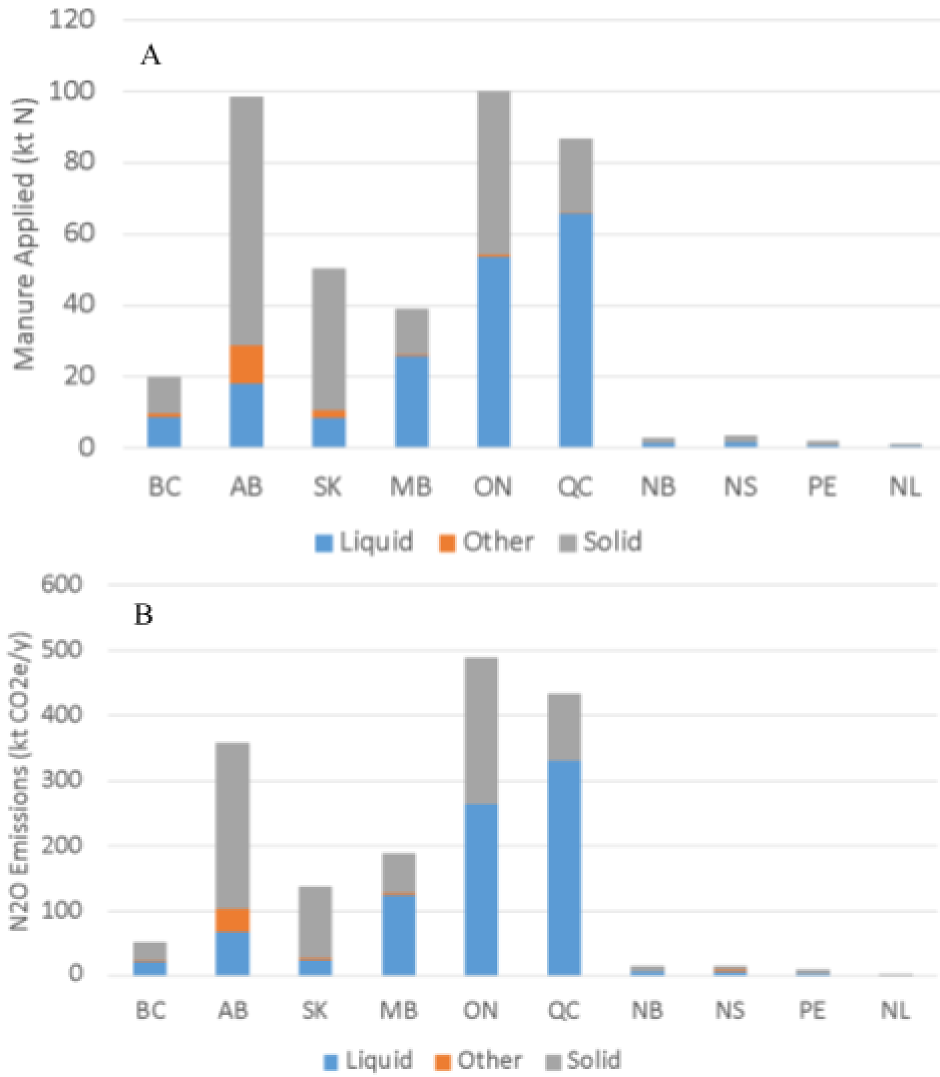


*Figure SEQ Figure * ARABIC 10: Manure N reported to be applied to soil (A) and N_2O emissions resulting from that manure application (B) in Canada in 2019 broken down by province and manure handling system (NIR, 2019).*

[1] Note this number is based on animal numbers and assumed rates of manure N production. Statistics as to the actual amount of manure produced are not collected. Note also that the amount of manure N applied to agricultural land in Canada for 2019 is reported to be 231 kt N by the UN FAO. The reason for this discrepancy is not apparent but may reflect differences in method of calculation.

When examining AWMS manure N applied to soil by animal species (Fig. 11), beef (non-dairy cattle) predominates with the manure being mainly handled as a solid, while dairy and swine are also major contributors, but their manure is primarily handled as liquid manure.

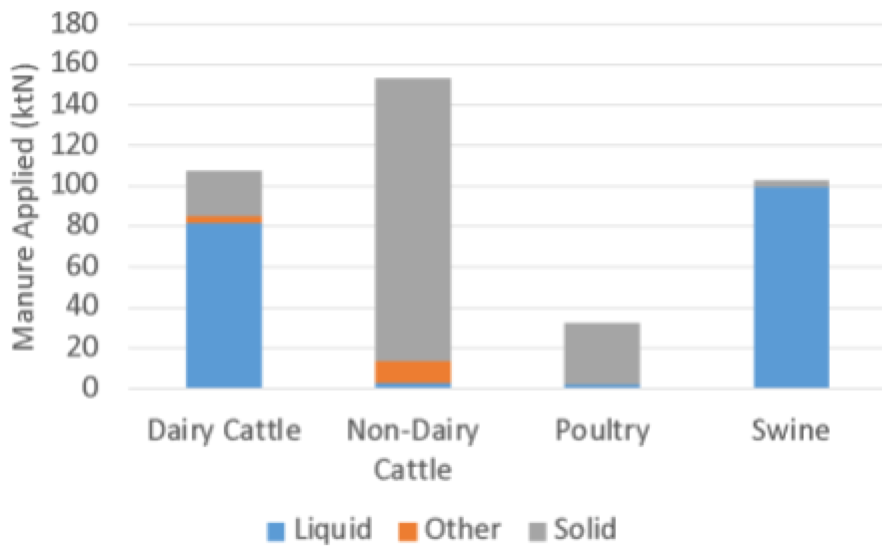


*Figure SEQ Figure * ARABIC 11: Manure applied to Canadian agricultural land in 2019 broken down by species and manure management system (NIR, 2020)*

A total of 268 kt N of manure N was deposited on pasture, range, and paddock in 2019. The majority (72%) of the N was generated by beef cattle in Alberta, Saskatchewan, and Manitoba (Fig. 12A) and deposition on PRP is the predominant (97%) fate of beef cattle manure (Fig. 11). Canada's NIR reports that in 2019 manure was collected and managed for 82% dairy, 50% of non-dairy cattle, 99% of poultry and 100% of swine. Canada's NIR notes a shift from solid manure storage to liquid, an increase in the number of covered manure storage systems and, in the case of the dairy sector, a shift in time in pasture. These trends have resulted in a decrease in the proportion of total N lost to the environment over time and this is reflected in the inventory.

N_2O emissions from AWMS manure applied to and manure deposited on pasture, range, and paddock

In 2019, direct N_2O emissions of 1700 kg CO_2e/y were associated with AWMS manure management. Greatest emissions occurred in the five central provinces (Fig. 10B). Direct N_2O emissions from manure on pasture, range and paddock systems were substantially less than those from AWMS manure totally 120 kt CO_2e/y (Fig. 12B) with the majority emissions occurring in Ontario and Quebec associated with the higher emission factors reflecting the more humid climatic conditions.

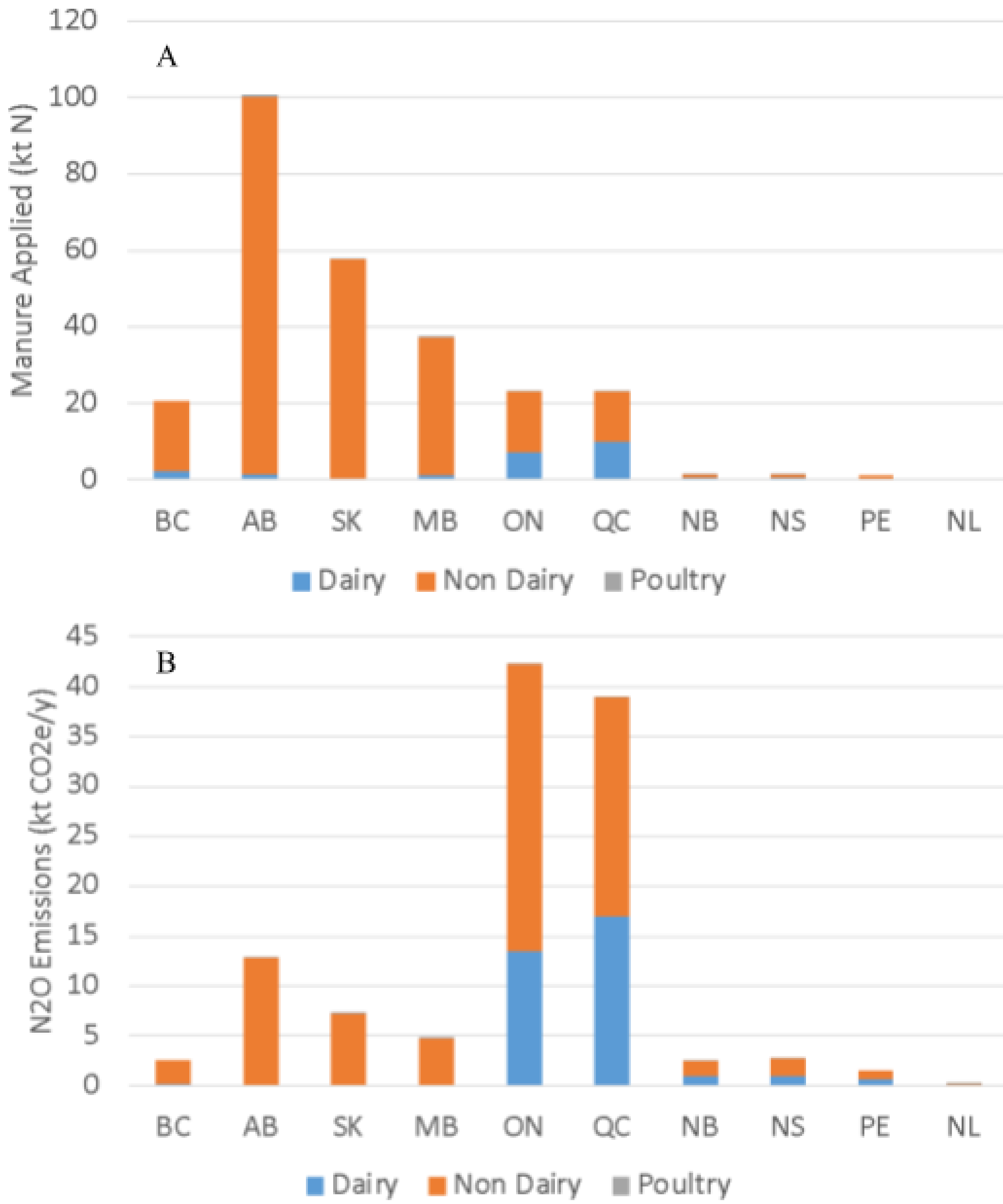


*Figure SEQ Figure * ARABIC 12: Deposition of manure N on pastures, range, and paddocks in Canada in 2019 broken down by province and animal species (A) and estimated N_2O emissions from manure deposited on pasture, range, and paddocks broken down by province and species (B). (NIR, 2020*

3. Details of Proposed emission-reduction measures

Manure Management

Description of proposed reduction measures including mechanisms of reduction action

Beneficial Management Practice 1 – Conserving the N content of manure

Several manure handling and storage practices can be implemented to reduce N losses, principally the loss of ammonia (NH_3) from manure and thereby maximize the retention of N in the manure. These practices differ according to the type of waste management system, primarily whether manure is handled as a solid, as a result of the addition of bedding, or as a liquid as a result of the addition of water.

Solid Manure - When manure is handled as solid, greater N retention can be achieved using bedding materials with wide C:N ratios. Carbon rich bedding materials such as straw or wood chips physically absorb urine and the wide C:N ratio of the material results in microbial action immobilizing the N in organic forms. As a result, the ammonium content of solid manure often 25% or less of the total N content. The use of bedding also increases the volume of manure that must be handled and results in a more heterogenous material that is more difficult to characterize in terms of its N content.

Liquid Manure – Storage of manure as a liquid provides a number of opportunities to retain manure N content and to produce a more homogenous and predictable plant N source upon land application. The anaerobic nature of liquid manure storage prevents nitrification, the conversion of ammonium to nitrate, and therefore limits the potential for N_2O production. Because there is less carbon-rich bedding added the majority of N (~75%) is in the form of ammonium. While ammonium is more plant available, it also has a much greater potential to be lost, particularly in alkaline environments. The loss of ammonia from manure can also result in indirect N_2O emissions in the adjacent environments where the ammonia is deposited.

Frequency of Transfer of Manure to Storage - Some practices are simply changes to manure handling such as the rapid transfer of manure from the barn floor to manure storage. Allowing urine to remain on the barn floor undilute results in ammonium from urea hydrolysis in an alkaline environment, also a result of urea hydrolysis and therefore high rates of ammonia (NH_3) loss. Hourly scraping of the barn floor or slotted barn floors result in 50% less ammonia loss than systems where the manure is allowed to accumulate on the barn floor undiluted and only transferred to storage on a daily basis (Table 1; Burton and Beauchamp, 1986).

*Table SEQ Table * ARABIC 1: Comparison of ammonia loss in various systems as determined from the nitrogen budget and direct measurement (Burton and Beauchamp, 1986).*

Barn	*Manure collection*	*Retention*	*Direct* (% of N excreted)	*Budget* (% of N excreted)
1	Daily scraping to pit	~1 day	19	21
2	Gravity directly to pit	~1 hour	9	5(25)[a]
3	Gravity to incline eventually to pit	~1 week	—	27

[a] Estimate in parentheses was derived from several restricted manure samples in the barn which resulted in a 20% underestimation of manure N content.

Manure Storage Acidification - There are also storage management options that can reduce NH_3 loss and therefore increase the nitrogen content of the manure. These include the acidification of storages, which is also promoted as a means of reducing CH_4 loss. Lowering the pH shifts the equilibrium from NH_3 to NH_4^+ and thus reduced the potential for NH_3 loss. The magnitude of NH_3 loss can be significant and therefore the retention of N can be substantial.

Manure Storage Covers – The addition of cover on a manure storage reduces air exchange at the manure surface and therefore the loss of NH_3. Covers have the added advantage that they prevent the dilution of manure with rainwater and therefore result in manure with a higher solids and nitrogen content, reducing the volume of manure that must be spread on the land.

Composting of Manure – Composting of manure with a high C:N ratio bedding material results in the immobilization of N as organic N compounds. Care must be taken to ensure both that there is sufficient carbon to immobilize the ammonium and that the pH of the compost is not allowed to become alkaline. Either situation can result in large amounts of ammonia loss.

Nitrogen Applied to Agricultural Soils

Synthetic Nitrogen Fertilizer - Increased adoption of 4R nitrogen management

In response to concerns relating to the potential for fertilizers to impact the environment, the fertilizer industry developed the 4R Nutrient Stewardship Program to promote improved fertilizer management. The 4R nutrient stewardship program refers to four key practices in nutrient management: 1) right source – choose plant-available nutrient forms that provide needed nutrients with the timing of release matched to crop demand, 2) right rate – ensure adequate, but not excessive, amounts of all limiting nutrients are applied to meet plant requirements in relation to yield and quality goals, 3) right time – time nutrient applications considering the interactions of crop uptake, soil supply, environmental risks, and field operation logistics, and 4) right place – place nutrients to take advantage of the root-soil dynamics, spatial variability within the field, and potential to minimize nutrient losses from the field (Reetz *et al.*, 2015). Here we propose further incentivization of practices that utilize and refine 4R principles to improve nitrogen management.

Beneficial Management Practice 2 – Quantitative determination of Right Rate

Right Rate is perhaps the most important of the 4Rs. It is critical that N fertilizer be applied at the *Right Rate* to realize efficiencies and N_2O emission reduction potential that the remaining three "Rs" offer. If the rate of N application exceeds crop demand, N accumulates in the soil and the efficiencies of the other three R's will not be realized and increased N will remain in the soil at harvest and often be lost during the non-growing period. In Canada, under current agronomic practices, N is often not the primary factor that is limiting crop yield production (Delaporte et al. 2020; STRATUS 2019; Morris et al., 2018). This is particularly true as climatic variability increasingly limits crop productivity as seen in Prairie Canada in 2021 (Climate Atlas of Canada: Agriculture). *Right Rate* is also the most contentious of the "Rs" in that it represents the tension between the desire for increased yield and the need to constrain N fertilizer use to meet environmental objectives.

There are multiple factors that should be considered in selecting the *Right Rate*. These include a consideration of the availability of nutrients from all sources (e.g., livestock manures, commercial fertilizers and atmospheric nitrogen fixed by legumes), performing of annual soil testing, apply nutrients to meet crop requirements while accounting for the nutrients already in the soil and the calibration of application equipment to deliver target rates (Fertilizer Canada). These approaches can be broadly characterized as those based on a yield goal, those that seek to optimize the profit function such as maximum return to nitrogen (MRTN), and more advance modelling approaches that consider soil and climatic factors such as ADAPT-N (Morris et al., 2018). These approaches balance reliance on more practical methods, relying on more immediately available tools and more amenable to reporting of outcomes of improved N management, with more sophisticated approaches attempting to realize maximum efficiency of N use. Here we advocate for the adoption of some of these practices both to improve N management and as a means of reflecting those improvements in reporting processes such as the National Inventory Report.

Nitrogen Balance – In the 4R framework determination of the *Right Rate* is dependent upon the assessment of all N sources and using this information in compiling a nitrogen balance (Fertilizer Canada). Quantifying on all N sources and completing a N balance on a field-by-field basis should be a component of an improved nitrogen management plan. This is also a common element in a number of the reporting conventions being used or proposed such as the N-Visible approach proposed by the Environmental Defense Fund (EDF). The N-Visible framework builds on evidence compiled by a group of expert scientists (including AAFC scientists) convened by the EDF and published in the peer-review press (Eagle et al., 2020). This group identified a relationship between a simple calculated N balance and N_2O emissions for a wide range of cropping systems (Fig. 13). The N-visible framework simplifies the N balance calculation by only considering the measurable N inputs (N fertilizer, manure, legume N fixation) less measurable outputs (crop yield, residues removed from the field), increasing the potential for this framework to be adopted broadly to support supply chain needs to document improved sustainability.

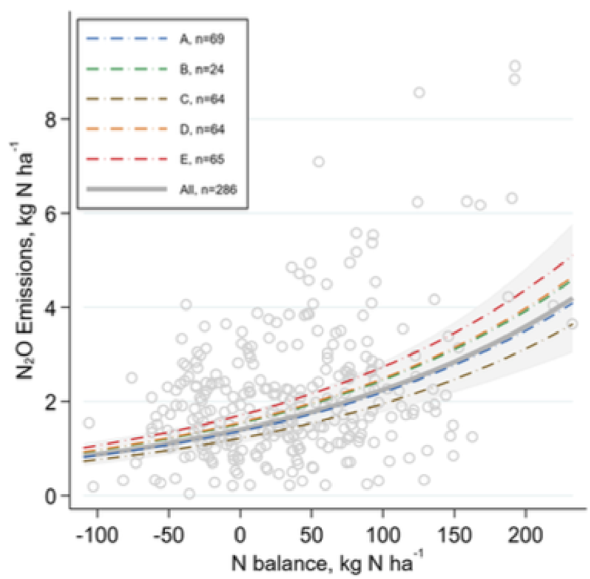


Figure 13: Relationship between calculated nitrogen balance and N2O emissions from a range of cropping systems (A..) from Eagle et al. (2020)

In the N-visible framework the balance between N inputs and N outputs cannot exceed a prescribed "safe zone" (Fig. 14) of between 28 - 84 kg N ha^{-1} (25 and 75 lbs N $acre^{-1}$). Staying within this safe zone will result in N losses between 7 - 22 kg N ha^{-1} (7 and 20 lbs N $acre^{-1}$). Note that a positive N balance of up to 28 kg N ha^{-1} is recommended to maintain soil health (build soil organic matter), balances should be less 84 kg N ha^{-1} to minimize the potential for N loss. An annual N loss of 22 kg N ha^{-1} is predicted at the upper limit of this safe zone. The pathways of N loss are not specified in this estimate.

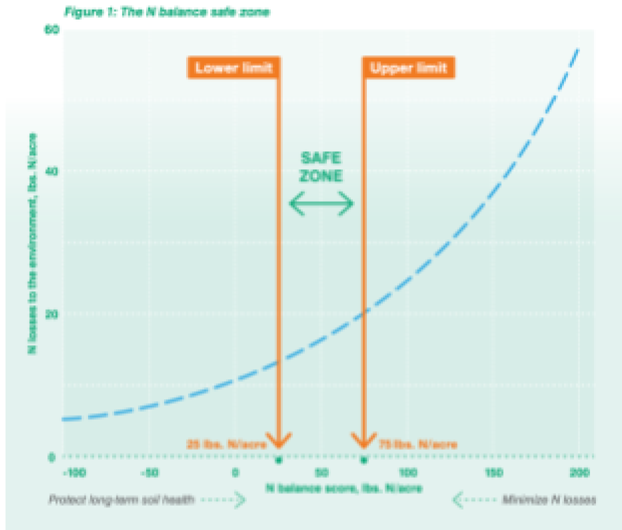


*Figure SEQ Figure * ARABIC 14: N-Visible "safe zone" where N inputs in excess of N outputs fall within an acceptable rage of N loss to the environment.*

An N balance approach is also at the heart of AAFC's Residual Soil Nitrogen Agri-Environmental Indictor (Fig. 15; Drury et al., 2016; Clearwater et al., 2016). It is important to note however that in this case the N balance calculation considers a broader range of N inputs (including atmospheric deposition) and estimates various N losses (N leaching, denitrification, ammonia loss) as N outputs. As a result, the value of Residual Soil Nitrogen is not directly comparable to the N balance calculated in the N-Visible framework.

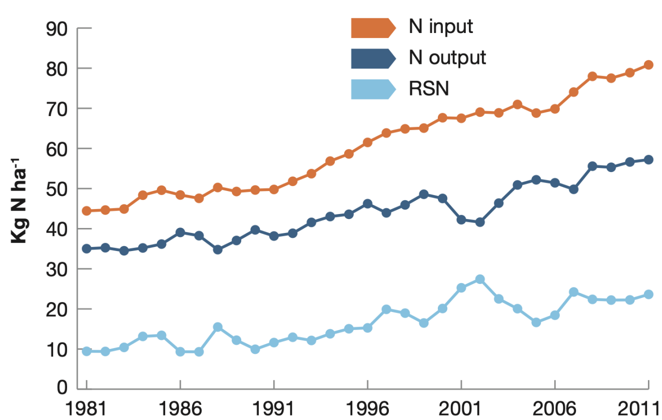
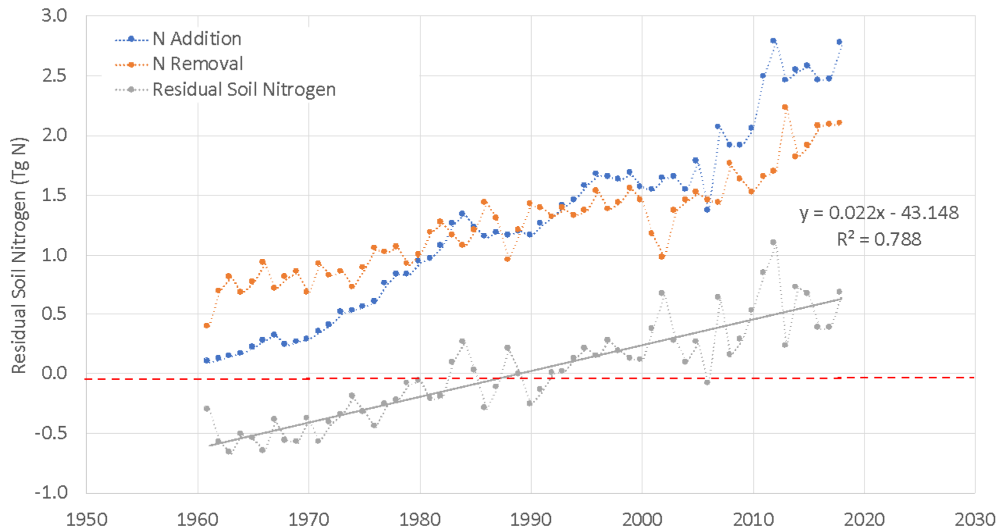


Figure 15: Estimated N input (fertilizer, manure, legume N fixation) and output (harvest, residue export, denitrification, nitrate leaching and ammonia loss) and calculated difference (residual soil N) in Canadian soils from1981 to 2011 (Clearwater et al., 2016).



Selecting Economically Optimum Rates of Nitrogen – The choice of *Right Rate* should be based on the determination of the most economic rate of N application, the rate where the last increment of addition N results in a yield gain of equivalent value to the cost of the N added. It is currently more common for provincial governments to promote fixed "recommended" regional rates of N fertilizer that have remained constant over decades as varieties and management practices have evolved. Where a yield goal is considered in determining a recommended N rate, it is often based on "target yields" that reflect a desired yield rather than the actual yields being obtained.

Numerous studies have identified the opportunity to reduce N fertilizer rates to reflect the increased efficiency of N fertilizer use. Venterea *et al.* (2016), in Minnesota, found that implementing 4R practices maintained corn yield with a 15% reduction in N fertilizer use. In a modelling study, Banger *et al.* (2020) estimated that with 4R practices, N fertilizer use in the corn growing area of Ontario could be reduced by up to 33% (Fig. 16). Utilizing basic 4R implementation with lower N fertilizer rates result in corn yield was predicted to up to 10% higher yield than the use high rates of N fertilizer without 4R practices. There was no further increase in yield with more implementation of 4R beyond the basic level when the same fertilizer rate was used.

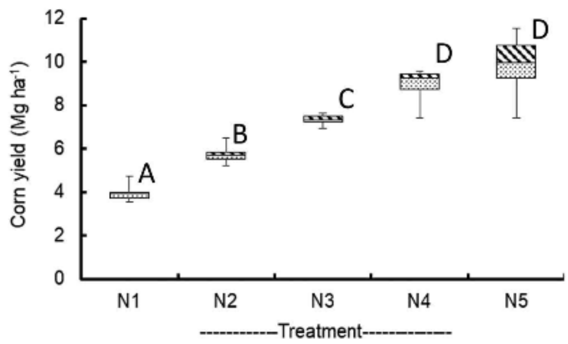


FIGURE 1 | Crop yield in the five nitrogen rate treatments during 2009–2018. In all the treatments, 30 kg N ha^{-1} was applied at planting and rest was sidedressed in the growing season. Total N rates in five treatments were as follows: N1: 30 kg N ha^{-1}; N2: 57 kg N ha^{-1}; N3: 87 kg N ha^{-1}; N4: 145 kg N ha^{-1}; N5: 218 kg N ha^{-1}. Treatments indicated by letters are significantly different from each other with a *p*-value of 0.05 based on a Tukey HSD (Honestly Significant Difference).

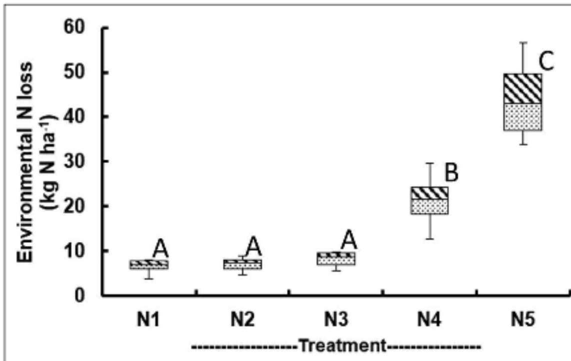


FIGURE 2 | DNDC estimated environmental nitrogen loss (kg N ha^{-1}) from corn production. In the treatments, 30 kg N ha^{-1} was applied at planting and rest was sidedressed in the growing season. Total N rates in five treatments were as follows: N1: 30 kg N ha^{-1}; N2: 57 kg N ha^{-1}; N3: 87 kg N ha^{-1}; N4: 145 kg N ha^{-1}; N5: 218 kg N ha^{-1}. Treatments indicated by letters are significantly different from each other with a *p*-value of 0.05 based on a Tukey HSD (Honestly Significant Difference).

Figure 17: Relationship between N application rate and A) corn grain yield (Mg/ha) and B) environmental N loss (kg N/ha) from Banger et al 2021.

It is important to note that, currently, farmer knowledge of 4R does not necessarily result in reduced fertilizer N or reduced N_2O emissions. In 2019, Ontario corn producers who indicated they were very familiar or somewhat familiar with 4R applied 28% higher rates of N fertilizer than those that were not familiar 4R practices (Stratus 2019). These higher rates of N fertilizer application were not offset by higher yields and therefore resulted in lower nitrogen response measured as the kg grain per kg N fertilizer added (Fig 17). Thus, familiarity with 4R practices alone did not translate into improved N management as expected, but in this case resulted in poorer N use efficiency. This is also consistent with earlier surveys by Stratus (Stratus, 2015, 2016), which reported that producers who believed that they had good familiarity with 4R practices used a higher average rate of N fertilizer application than those who professed no familiarity with 4R practices. This underscores that knowledge of 4R practices alone is not sufficient to lead to improved N management. Selection of the right rate of N fertilizer application must be based on realistic yields that consider other factors that limit yield (i.e., weather, disease).

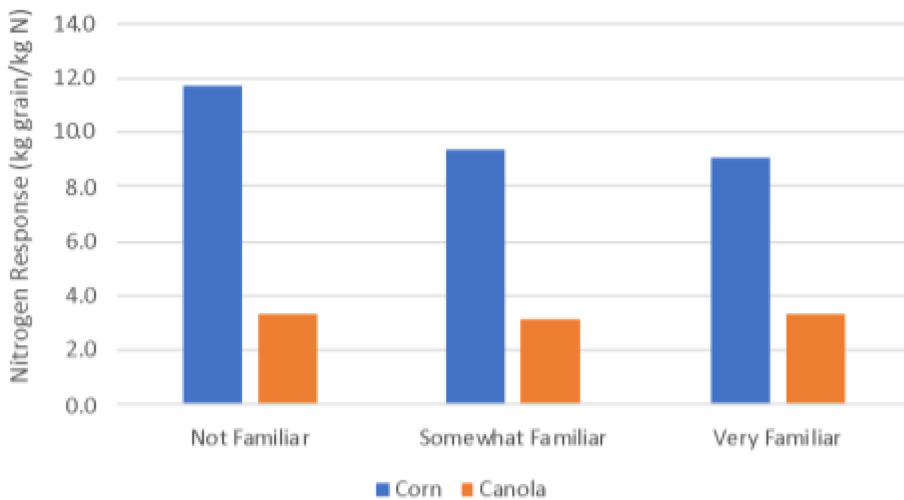


*Figure SEQ Figure * ARABIC 18: Nitrogen response (kg grain/kg N fertilizer) for corn production in Ontario and canola production in Prairie Canada in 2019 (data from Stratus, 2019).*

Yanni et al. (2020) assumed that a 20 kg N/ha reduction from (170 to 150 kg N/ha) resulted in no yield loss on corn. De Laporte et al. (2020) shows that an average reduction in N rate from 176 kg N/ha to 124 kg N/ha results in an average corn yield loss of only about 1.1% across the province of Ontario over 30 years of weather with some other practice adaptations.
The benefits of the increased efficiency of 4R will be greatest when they are coupled with a reduction in the rate of N fertilizer application to reflect the increase in efficiency (Zebarth et al., 2012).

Determination of N Right Rate should be based on actual yields (5-year yield averages +5%) rather than "target yields". Where crop yield is considered in determining N fertilizer requirements, often producers will "target" a yield that is above historical yields realized. Together with the common practice of adding "insurance" nitrogen to the rate recommendations, these practices result in inefficient nitrogen use and environmental impact. We recommend that N rate recommendations be based on crop N demand using 5-year historical yields for the field with a +5% yield increase as the basis for fertilizer N rate recommendations. We estimate this could result in a 10% reduction in synthetic N fertilizer use without a statistically significant decrease in yield.

Quantify soil N supply – One of the basic tenants of determining right rate is to "consider the availability of nutrients from all sources" and that annual soil testing should be conducted in support of determining the *Right Rate*. Unfortunately, there are not currently robust and broadly accepted means of quantifying the nitrogen supplying potential of the soil. In Prairie Canada fall nitrate testing if often employed, but this fails to account for the potential for growing season N mineralization. In the more humid regions of the country fall nitrate testing is ineffective and the use of spring or early season nitrate tests such as the Pre-Side dress Nitrogen Test (PSNT) are not widely used. The nitrogen supplying capacity of the soil varies significantly between regions, within a field, and is impacted by cropping system and management practices. Over the past several decades multiple researchers across the country have been working on the methods to measure nitrogen supplying capacity (Zhang et al., 2002; Selles, et al., 1999; Sharifi et al., 2010; Dessureault-Rompré et al., 2011; Niyraneza et al., 2012; St Luce et al., 2012), but these methods have not found their way into common practice. There is a need to adopt site-specific measures of soil N supplying capacity as part of routine soil testing and use these results in determining the need for supplemental N additions. This requires investment that translates our current scientific understanding of the measurement and prediction of N mineralization into commercially available soil testing processes and mapping products. This is not an issue unique to Canada. Several approaches to quantitively determine *Right Rate* and assessing the appropriateness of those rates have emerged (Morris et al., 2018).

Use of in-field calibration strips – Nitrogen rate recommendations are often the result of regional N response trials, in some cases decades old. With advent of precision fertilizer applicators and yield monitoring combines the opportunity exists for producers to establish calibration strips within their fields on a routine basis. Here we recommend an extension program that would have producers establish a 10% N rate reduction strip on each of their fields each year to confirm that their current rate of N use is appropriate. The program could compensate for yield loss should it occur. Over time this would provide the producer with a field-specific calibration of the response of their crop to the N rate being applied.

Measurement of nitrate remaining in the soil after harvest – One of the challenges producers face is assessing how well they have done in balancing N inputs with removals. Is not practical for producers to measure N_2O emissions or NO_3^- leaching on their fields. One relatively straightforward tool to assess the crop N balance is to measure the amount of nitrate remaining in the soil profile after harvest. In Prairie Canada this measure also has value in predicting the amount of N that is likely to be carried over to the next crop year. In the rest of Canada this measure reflects the amount of N that is likely to be lost during the non-growing period.

This concept also forms the basis of one of Agriculture and Agri-Food Canada's Agri-environmental Indicators residual soil nitrogen (RSN) (Clearwater et al., 2016). RSN is calculated as the difference between N inputs (N fertilizer, manure, crop residue) and N outputs (harvest N). Note that in Canadian agricultural soils the majority of mineral N remaining in the soil in the fall would be in the form of nitrate (NO_3^-) as in these soils as nitrification would go to completion, converting ammonium (NH_4^+) to NO_3^-. RSN is an indicator of the potential for environmental impact on water, primarily as NO_3^- leaching, and air as a result of N_2O emissions. An examination of RSN over time in Canadian agriculture indicates a trend from negative values for RSN prior to 1985, indicating net removal of N from Canadian agroecosystems, primarily originating from soil N mineralization to a condition of positive values for RSN, indicating N additions in excess of N removals and therefore an increase in the potential for N loss (Fig. 16). This is imbalance not only directly drives increased N_2O emissions and nitrate leaching, but it is also an indicator of the potential for N loss from Canadian agriculture.

Right Place

Variable Rate/Precision Nitrogen Management – there are emerging technologies that will facilitate the precision N management, varying the rates of N fertilizer application based on the productivity areas of the field and sensor-based techniques for in-crop N application based on weather and crop status. Together these techniques provide the opportunity for more precise N management in space and time. Unfortunately, there are relatively few studies in the literature that quantify the opportunity for precision N management to reduce N_2O emissions. A European study by Sehy et al. (2003) found that reducing N fertilizer application by 17% in low yielding portions of the landscape resulted in 34% reduction in N_2O fluxes with no impact on corn yield (Sehy et al., 2003). Glenn et al. (2021) found that variable rate N application resulted in reduced N_2O emissions in Canola production in Manitoba, but this effect varied considerably from year to year (Fig. 19).

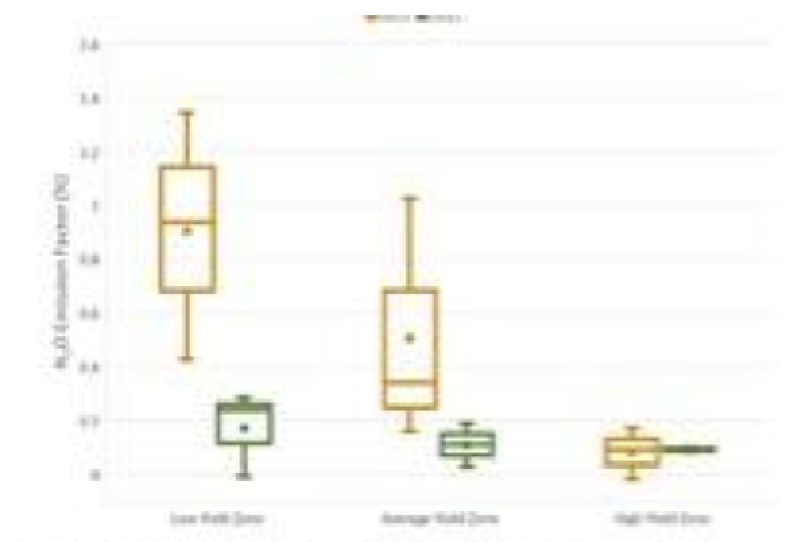


*Figure SEQ Figure * ARABIC 19: N2O emission factor as a function of yield zone in canola production in Manitoba (Glenn et al., 2021).*

Beneficial Management Practice 3 – Increased Adoption of Precision Nitrogen Management

This approach requires the development of means of generating variable rate N application maps based on variable estimated yield potential using an N balance approach and landscape-based soil N supply models. These tools exist to a limited degree and the industry is currently exploring their use and these approaches are being adopted by a limited number of more advanced producers. Variable rate N application can be very technology intensive, requiring precision N supply maps, equipment capable of variable rate N application, and GPS enabled yield monitors. There are a number of firms that currently offer both field mapping services and custom variable rate N application services. There also less technical approaches that are based on the identification of management zones within a field and different, but uniform, N management within those zones. Both approaches have the potential to increase N use efficiency and reduce N_2O emissions. The barriers to the adoption of management zone approaches are less than for that for approaches that are dependent on more advanced technologies. Which approach is most appropriate to a particular farmer is a function of the nature of variation in yield potential within the field, the value of the crop, and the cost of N fertilizer.

There is considerable opportunity to improve the tools and approaches used in developing precision N strategies including the generation of predictive soil N maps, use of in-field sensors of crops N status to inform side-dress and foliar N applications and improved models of the impact of climate variability on landscape-based yield potential. Fortunately, this is an area of active research in Canada and advances can be anticipated in the near future.

Right Time

Beneficial Management Practice 4 – Elimination of fall N application

The application of N fertilizer in the fall previous to canola production is a relatively common practice. In 2019, 23% of Canola growers applied N fertilizer in the fall of the previous year, 43% of those producers applying N in the fall were from Manitoba and many (35.7%) claimed to be very familiar with 4R (STRATUS, 2020). Studies have shown fall applications resulted in greater N_2O emissions (Kryzanowski, 2018), failed to detect a difference in spring vs fall applications (Abalos et al., 2016a), or found fall N applications to result in less N_2O loss than spring applications (Tenuta et al., 2016; Kryzanowski et al., 2018), particularly if N application occurs late in the fall after the soil has dropped below 5 °C (Tenuta et al., 2016). In a study of N fertilization of barley across soil zones in Alberta, Kryzanowski (2018) found spring application of nitrogen fertilizer was the most effective means of reducing total emissions. Switching from fall applied nitrogen fertilizer to spring application resulted in 17% to 25% reduction in N_2O emissions.

Right Source

There is extensive research supporting the value of enhanced efficiency fertilizers in reduction N_2O emissions. This research demonstrates that these reductions are greatest and most consistent when the increased efficiency of the N source is reflected in a reduced rate of N application. Only in situations where the rate of unenhanced fertilizer is insufficient to meet plant nitrogen demand will an enhanced efficiency product result in greater yield. Reflecting enhanced efficiency N sources in rate will also help pay for the additional cost associated with the enhanced efficiency fertilizer product.

Beneficial Management Practice 5 – Increased use of Enhance Efficiency Nitrogen Fertilizer

Enhanced efficiency fertilizers (EFFs) have been shown to result in relatively consistent reductions in N_2O emissions (Drury et al. 2012; Decock 2014; Thapa et al., 2016; Vyn et al. 2016; Drury 201; Snyder, 2017; Eagle et al., 2017). In a meta-analysis of 113 data sets from 35 studies globally, Akiyama et al. (2010) report an average reduction in N_2O emissions of 38% and 35% from inhibitor treated products and polymer coated products, respectively. Similarly, in their global meta-analysis, Thapa et al., (2016) reported mean reductions in N_2O emissions as a result of the use of nitrification inhibitors of 38%, urease in combination with nitrification inhibitors of 30%, and controlled-release N fertilizers of 19%. Thapa et al. (2016) found nitrification inhibitors and controlled release products gave relatively consistent reduction of 25% to 50%, whereas urease was more variable resulting in 0 to 50% reductions. Urease inhibitors alone are less effective in controlling N_2O emissions (Abalos et al., 2016b; Akiyama et al., 2010).

Thapa et al. (2016) examined the influence of crop type, pH, texture, mode of application, tillage, and irrigation on the effectiveness of nitrification inhibitors, urease inhibitors, a combination of urease and nitrification inhibitors and controlled release products. Nitrification inhibitors were most effective in corn-based systems and when used in banded fertilizer applications (Thapa et al., 2016). The use of nitrification inhibitors alone can result in increased loss of NH_3 (Drury et al., 2017; Snyder, 2017) and are generally less effective at reducing N_2O loss from alkaline soils as compared to neutral or acidic soils (Thapa et al. 2016).

The use of urease inhibitors in combination with nitrification inhibitors have been reported to result in greater reductions in N_2O emissions (Decock, 2014; Abalos et al. 2016b; Drury et al., 2017; Snyder, 2017), particularly in alkaline soils, when used in banded systems, coarse-textured soils or in irrigated systems (Thapa et al. 2016). Drury et al. (2017) noted that when ammonia volatilization was reduced by adding a urease inhibitor, N_2O emissions were increased by 30% when comparted to a nitrification inhibitor alone. They noted that by reducing pollution swapping (increased NH_3 loss to reduce N_2O loss), corn grain yields increased by 5% to 7%. The combination of a urease and a nitrification inhibitor resulted in increased yields of 19% compared with urea without the inhibitors. Wagner-Riddle (2017), in examining corn production systems in Ontario, observed significant reduction in N_2O emissions and NO_3^- loss when UAN+EEF was applied as side-dress in a wet year when emissions were large. Tenuta (2017) examined the potential for enhanced efficiency fertilizers in combination with fall N application to reduce N_2O emissions. They observed treatments with highest cumulative N_2O emissions were LIMUS (urease inhibitors), urea, ESN (coated urea) and anhydrous ammonia N-serve (nitrification inhibitor), with eNtrench (nitrification inhibitor) and SuperU (urease + nitrification inhibitor) resulting in lower emissions. Snyder (2017) reported more consistent reductions in direct N_2O emissions as a result of the use of dual urease and nitrification inhibitor combinations, falling in the range of 17% to 46% reduction in N_2O emissions.

Manure applied to Agricultural Soils

Beneficial Management Practice 6 – 4R management of manure

Much like synthetic N fertilizer attention to place, time, and rate of manure application can result in improved N retention. The potential to reduce N_2O emissions is less clear. The addition of a labile N source such as ammonium in combination of available organic carbon compounds and water create an environment conducive the soil microbial activity and as a result can increase N_2O emissions. Where there are clear opportunities to reduce N_2O emissions is in ensuring the N added as manure is appropriately credited with a corresponding reduction in amounts of synthetic N fertilizer added. Thus, measures to conserve nitrogen upon land application indirectly create opportunities to reduce N_2O emissions. There are also opportunities to use nitrification inhibitors to conserve the N content of the manure and reduced N_2O emissions. The addition of DCD to pig manure resulted in a 60% reduction in N_2O emissions in a corn-wheat system in Brazil (Aita et al., 2015).

Liquid Manure

Right place

The injection of liquid manure N into soil is a desired practice to significantly reduced NH_3 loss and odours, but this practice has also been shown to increase N_2O loss (Zhou et al., 2017; Cambereri et al., 2017). The loss of ammonia upon surface application can be sizable, as much as 50%, and thus injection of manure can result in sizable increase in the N supplied by manure application. Proper crediting of the ammonium conserved, an immediately available source of N to the plant, by reducing the rate of synthetic N fertilizer application could offset these increased emissions. Failure to do so would result in increased emissions (Charles et al., 2017).

Right time

Manure is commonly applied in the fall, often in the early fall when soil temperatures are still warm enough to allow nitrification to convert ammonium to nitrate. Nitrate accumulation in the fall increases the potential for overwinter N_2O emissions. Often the non-growing season, particularly spring thaw is the period of greatest N_2O emissions. From a nitrogen management perspective, it would be beneficial to delay the application of manure, particularly liquid manure, until the spring. This is not always possible as a result of insufficient storage capacity to store manure for a full year or limited time available during the spring available to apply the manure prior to planting.

There are two management strategies that can be used to limit nitrification of fall applied manure N. The first is to delay application until soil temperature drops below 5 °C. The nitrification process slows considerably at these temperatures and most of the N in the manure would be retained as ammonium until the soil warms in the spring. The second approach is to add a nitrification inhibitor to the manure. This chemically inhibits nitrification and would similarly retain most of the N in the manure until the soil warms in the spring. Both practices apply to primarily to liquid manure where much of the N is contained in the ammonium form and therefore has the greatest potential to be nitrified to nitrate. In solid manure more of the N is in an organic form and takes longer to mineralized before nitrification can occur and therefore accumulation of nitrate in the soil in following fall application is less of an issue. These practices are also most relevant in the more humid regions of the country where overwinter N losses are greatest. The elimination of fall application of manure or delayed fall application until the soil has cooled to below 5 °C would reduce N_2O emissions by 17% (Vanderzaag et al., 2011).

Solid Manure

Despite solid manure being the greater source of N_2O emissions, the opportunities to manage solid manure applied to land are more limited. The most significant of which is the rapid incorporation of the manure following land application. Like, liquid manure, the rapid integration of the manure into the soil decreases the potential for NH_3 loss and, if properly credited, the opportunity to reduce synthetic N application (Cambereri et al., 2017).

While the ammonium content of manure can be considered to be plant available at the time of application, the organic N contained in manure must be mineralized by microbial action to become plant available following land application. The time required for N mineralization is a consideration in terms of the utilization of solid manure as a plant N source. In essence solid manure acts as a slow-release N source, much like enhanced efficiency fertilizers. However, unlike enhanced efficiency fertilizers, the N_2O emissions from solid manure tend to be greater than from either liquid manure or inorganic N sources (Fig. 20; Decock, 2014). This is a result of the carbon content of the manure stimulating microbial activity resulting in greater oxygen consumption and a greater frequency of denitrifying conditions that contribute to N_2O production.

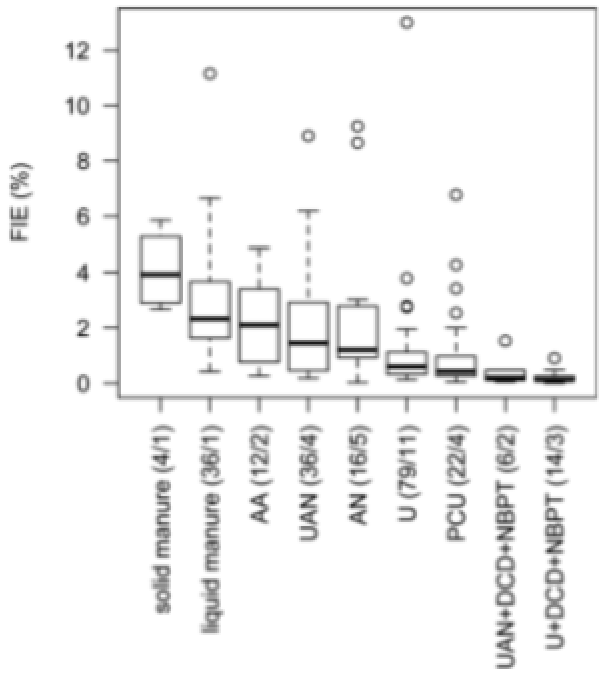


*Figure SEQ Figure * ARABIC 20: Influence of N source on fertilizer induced emissions (FIE) of N_2O. AA = anhydrous ammonia; UAN = urea ammonium nitrate; AN = ammonium nitrate; U = urea; PCU = polymer coated urea; DCD and NBPT are nitrification and urease inhibitors respectively. Numbers in parentheses indicate the number of observations on which the analysis was based, and the number of different field site from which observations originated (Decock, 2014).*

Beneficial Management Practice 7 – Improved crediting of organic N sources

The nitrogen content of manure is often discounted in nutrient management due to uncertainty in the composition of manure, the uniformity of its application in the landscape, and the timing of the availability of the N contained in manure to the crop. The uncertainty is greatest for solid manures. Liquid manure tends to be more uniform in storage, can be applied with greater certainty, and the high percentage of ammonium makes determination of plant available N more predictable. Thus, the trend to increased handling of manure as a liquid helps reduce the uncertainty associated with N supply to crops. To address difficulties in sampling and timely analysis of the N content of the manure producers often rely on "book values" of manure N content and conservative estimates of the N value of the manure. Despite having a strong suite of tools to inform manure management such as Ontario's AgriSuite, a relatively small percentage of Ontario farmers indicated that they use these tools and a majority either do not test their manure or test it less than every three years.

4. Quantification of emissions

Table 2: Summary of N_2O emission reductions associated with manure management and synthetic N fertilizer management.

Beneficial Management Practice	Emissions Reduction (kt CO_2e/y)
Manure Management	***250***
BMP 1 - Conserving the N content of manure in storage	250
Synthetic N Fertilizer Management	***3,485***
BMP 2 - Quantitative determination of Right Rate	1,115
BMP 3 - Increased adoption of precision nitrogen management	360
BMP 4 - Increased use of enhance efficiency nitrogen fertilizer	1,860
BMP 5 - Elimination of fall N application	200
Land Application of Manure Management	***535***
BMP 6 - 4R management of manure	165
BMP 7 - Improved crediting of organic N sources	370
Total	***4,270***

The above represent a total reduction of 3.5 Mt CO_2e/y in N_2O emissions. When expressed a percentage of the 10.6 Mt CO_2e/y of the direct N_2O emissions associated with N fertilizer use this represents a 33% reduction in emissions.

Manure Management

Beneficial Management Practice 1 – Conserving the N content of manure

Improving the handling and storage of manure could reasonably result in an increase in the ammonium content of the manure by 50%. For liquid manure that would result in an approximate increase in the N value of the manure upon land application by 40% (ammonium content is typically 80% of total N) for solid manure the increase would be more like 20% as the ammonium content of solid manure is less (~ 40% or less).

In 2019 there were 187 kt of N of liquid manure N and 205 kt N of solid manure applied to soil in Canada (NIR, 2020) and thus these conservation measures would result in an additional 75 kt N in liquid manure and 41 kt N in solid manure being applied to Canadian soils. If a 50% credit were applied to this additional N it could replace approximately 58 kt N of synthetic fertiliser N. While we do not anticipate there would be any less N_2O generated from the manure N itself, there would be less emission as a result of the avoided emissions of the additional synthetic N fertilizer and its manufacture and transport. The avoided use of synthetic N fertilizer would represent an emissions reduction of 250 kt CO_2e/y.

Nitrogen Applied to Agricultural Soils

Beneficial Management Practice 2 – Quantitative determination of Right Rate

Analysis of 5 years of canola N response data (Holzapfel and May, 2017; Holzapfel, 2018; Holzapfel, 2019) indicates that the maximum economic rate of N application to canola is approximately 150 kg N/ha. The STRATUS Survey indicates that 23% of canola acres used rates greater than this amount, some significantly higher (> 250 kg N/ha). We propose that more systematic adoption Right Rate determination based on most economic rates of N application and soil N testing could result in a 30% (~50 kg N/ha) reduction in N fertilizer application on 25% (2.3 Mha) of the land in canola. This equates to a 415 kt CO_2e/y.

Long term trials examining corn response in Ontario suggest an economically optimum N rate for corn production is approximately 170 kg N/ha. The STRATUS Survey indicates that 54% of corn acres used rates greater than this amount, some significantly higher (> 300 kg N/ha). We propose that more systematic adoption Right Rate determination based on most economic rates of N application and soil N testing could result in a 50 kg N/ha reduction in N fertilizer application on 50% (0.7 Mha) of the land in corn. This equates to a 360 kt CO_2e/y reduction.

For wheat propose that more systematic adoption Right Rate determination based on most economic rates of N application and soil N testing could result in a 25 kg N/ha reduction in N fertilizer application on 50% (3.58 Mha) of the land in wheat. This equates to a 340 kt CO_2e/y reduction.

Beneficial Management Practice 3 – Increased adoption of precision nitrogen management

This approach required the development of means of generating variable rate N application maps based on variable estimated yield potential using an N balance approach and landscape-based soil N supply models. The industry is currently exploring these tools and they are being adopted by more advanced producers. For this analysis we estimate the N_2O emission reduction potential for the implementation of variable rate N management, whether based on advanced technology or through the implementation of management zones for the major crops in Canada.

For canola production in the Prairies increase precision N management might double from the 2019 value of ~15% to 30% by 2030. If we assume that adoption of precision N management could result in a 10% reduction in N_2O emissions and a 10%

reduction in N fertilizer rate (15 kg N/ha), over 1.7 Mha (20% of canola acreage) this could result in 200 kT CO_2e/y in direct N_2O emissions in 2030.

For corn production in the rest of Canada increase precision N management might triple from the 2019 value of ~12.6% to 35% by 2030. If we assume that adoption of precision N management could result in a 10% reduction in N_2O emissions and a 10% reduction in the rate of N fertilizer application, over 0.4 Mha (20% of corn acreage) this could result in 85 kT CO_2e/y in direct N_2O emissions.

For wheat if we assume that adoption of precision N management could result in a 10% reduction in N_2O emissions and a 10% reduction in N fertilizer rate (7 kg N/ha), over 1.4 Mha (20% of wheat) this could result in 75 kT CO_2e/y in direct N_2O emissions in 2030.

Beneficial Management Practice 4 – Increased use of enhance efficiency nitrogen fertilizer

Given the extensive amount of research on the use of EEFs I believe there is sufficient published research to support the 15-35% reduction estimated by AAFC.

There is relatively low adoption of the use of enhanced efficiency products in canola (~ 10% of N fertilizer applied) and therefore there is considerable opportunity for further adoption. This particularly true for EEFs as the major barrier to adoption is cost. The increasing cost of N fertilizer will make EEFs more financially attractive, and this is a BMP that could easily be supported by a financial incentive. Thus, it would not be unreasonable to suggest that as much as 50% of the N fertilizer applied to canola could be EEF products and that the rate of N fertilizer addition could be reduced by 10% when using these products. The literature indicates that the use of an EEF N source results in a 35% reduction in N_2O emissions.
This would translate into 1120 kt CO_2e/y in direct emissions reductions in 2030.

There is also relatively low adoption of the use of enhanced efficiency products in corn (~ 5-10% of N fertilizer applied) and therefore there is considerable opportunity for further adoption. As with canola, we assume that as much as 40% of the N fertilizer applied to corn could be EEF products and that the rate of N fertilizer addition could be reduced by 10% when using these products. This would translate into 440 kt CO_2e/y in direct emissions reductions in 2030.

For wheat we assume that as much as 25% of the N fertilizer applied to wheat could be EEF products and that the rate of N fertilizer addition could be reduced by 10% when using these products. This would translate into 300 kt CO_2e/y in direct emissions reductions in 2030.

Together the estimated emission reductions associated with the use of enhanced efficiency N fertilizers in canola, corn, and wheat result in a 1,970 kt CO_2e/y less than the 2,350 kt CO_2e/y proposed by AAFC.

Beneficial Management Practice 5 – Elimination of fall N application

For canola, 20% of the N applied was applied in the fall. We estimate that switching from fall N application to spring application on 15% of the land in canola (1.3 Mha) could result in a 10% reduction in N_2O emissions and would allow a 10% reduction in N application rate to reflect increased efficiency. This would result in a 200 kt CO_2e/y reduction in N_2O emissions, a bit larger than AAFC's estimate of 120 kt CO_2e/y.

Beneficial Management Practice 6 – 4R management of manure

Right time

Retaining fall applied liquid manure N as ammonium in the more humid regions of Canada by either of these management approaches would increase the nitrogen retention by approximately 25% and would reduce N_2O emissions associated with manure application by 50%. The improved N retention would result in the retention of an additional 33 kt N/y, if fully credited in a reduced N fertilizer application would result in a reduction of 55 kt CO_2e/y. The reduced N_2O emissions occur overwinter period would result in a 110 kt CO_2e/y year reduction in N_2O emissions. The primary benefit of these practices would occur in Ontario in Quebec where the greatest amounts of fall application of liquid manure occur.

Beneficial Management Practice 7 – Improved crediting of organic N sources

More accurately crediting the N contained in that manure could result a replacement of 25 kg N/ha of fertilizer following application of liquid manure (25% of the manure N application rate of ~100 kg N/ha) and 10 kg N/ha of fertilizer following application of solid manure (20% of the manure N application rate of ~50 kg N/ha). If applied to the total manure N applied as a liquid (187 kt N) and as a solid (205 ktN) according to 2019 NIR data provided by ECCC, this would result in an N_2O emissions reduction of 210 kt CO_2e/y for liquid and 160 kt CO_2e/y for solid manure.

5. Current adoption and potential increase by 2030

A lack of activity data associated with on farm N management has severely constrained this analysis and will constrain efforts to reflect improved N management in the NIR. Here we have relied extensively on a series of surveys commission by Fertilizer Canada and conducted by STRATUS Ag Research. We primarily use the resulted reported for their 2019 survey as it corresponds to the 2019 data we have been provided by ECCC. The STRATUS survey is restricted to the major crops canola, corn and soybeans. Canola and corn are also the dominant N demanding crops grown in Canada and thus the survey is particularly useful in assessing those crops. The lack of activity for other crops severely limits our ability to describe baseline activity or develop meaningful estimates of potential adoption by 2030.

Nitrogen Applied to Agricultural Soils

Synthetic N fertilizer

Beneficial Management Practice 2 – Quantitative determination of Right Rate

Soil Nitrogen Testing

The 2019 STRATUS Survey reports that 17.1% of producers soil test for N annually, 9.8% every second year, 24.1% every 3 years, 16.5% every 4 or 5 years, 13.6% less than 4 or 5 years and 16.2% never test their soil for nitrogen. In general, the frequency of soil N testing has decreased from 2015 to 2019. The provinces where soil N testing is performed most frequently are Alberta and Manitoba where annual soil N testing is carried out by 41.7% and 41.0% of respondents. In Manitoba 21.2% of producers test their soil annually, while in Ontario on 3.6% of producers test their soil annually. Large farms, run by young to middle aged producers who are very familiar with 4R practices are the most frequent users of soil N tests. The primary reasons for not performing soil N tests are cost (39.9%), reliance on other systems (20.2%), don't think soil tests are useful (19.9%), "for some crops it does not matter if I soil test the year before" (15.7%), not advocated by advisor (15%), do not trust the results (10.2%), service not available (3%), and reliance on experience, historical results (2.2%).

There is a need to increase participation in soil N testing, particularly in Ontario. It is not unreasonable to suggest a doubling or tripling of annual soil nitrogen testing from 17% annual testing to at 34% to 50%. This could be achieved by adopting a test that was more informative to producers, ensuring the benefits of the test are communicated to agronomists and producers, and decreasing the cost of testing (both sample collection and analysis). A 50% subsidy of soil sampling and nitrogen testing is recommended. There is also a need to provide improved extension with respect to the interpretation of these tests to generate confidence in the recommendations that result. Improved training of Agrologists and Provincial and Federal extension/technical communication staff would achieve this objective.

Right Rate - Canola (Prairies)

Broken down by province the average N application rates for canola production were Manitoba 142.8 lbs N/ac (160 kg N/ha), Saskatchewan 127.3 lbs N/ac (143 kg N/ha), and Alberta 127.2 lbs N/ac (143 kg N/ha). A total of 14.1% of the N fertilizer used in canola in 2019 was applied at an application rate in excess of 220 lbs/ac (247 kg N/ha). The highest rates of N application (136.7 lbs/ac; 153 kg N/ha) were applied by those who indicated they were very familiar with 4R principles. (STRATUS, 2019).

In 2019, 44.7 of canola producers indicated they used a nitrogen soil test to decide fertilizer rates, 30.5% based on nutrient balance, 53.7% based on past experience, and 19.2% use a third-party consultant. Only 2.6% relied on government recommendations and 3.0% relied on CCA recommendations.

From 2015 to 2019, actual yields were 94%, 97%, 90%, and 90% yields. In 2019, producers who indicated they were very familiar with 4R practices reported actual yields were 90% of target yields, a yield gap of 5 bushels/ac (STATUS, 2019). In 2019, 13.6% of canola growers indicated that weather caused them to deviate from their fertility program, 46.4% was as a result of drought conditions with 33.3% reporting a decrease in fertilizer application. This further suggests there is an opportunity to reduce recommended N rates by 5% across the board without any yield impact.

Right Rate - Corn (Ontario)

Average N application rates for corn in Ontario have been increasing each year from 2015 to 2019. In 2015 the average rate was were148 lbs N/ac (166 kg N/ha), 2016 148.3 lbs N/ac (166 kg N/ha), 2018 157.3 lbs N/ac (176 kg N/ha), and in 2019 170.5 lbs N/ac (191 kg N/ha). The most frequent rate of N application was 180 to 190 lbs N/ac (200 - 213 kg N/ha), but 6.1% of corn acres receive a rate in excess of 300 lbs/ac (247 kg N/ha). In 2019, the highest rates of N application were applied by those who indicated they were either very familiar with 4R principles (174.3 lbs/ac; 195 kg N/ha) or somewhat familiar (176.7 lbs/ac; 198 kg N/ha). In 2019, 12.3% of the N fertilizer used was applied at a rate in excess of 300 kg N/ha. (STRATUS, 2019).

In 2019, 37.5 of corn producers in Ontario indicated they used a nitrogen soil test to decide fertilizer rates, 38.8% based on nutrient balance, 40.4% based on past experience, and 23.0% use a third-party consultant. A total of 9.8% of producers relied on government recommendations and 11.9% relied on CCA recommendations.

From 2015 to 2019, actual yields were 98%, 94%, 100%, and 90% yields. In 2019, producers who indicated they were very familiar with 4R practices reported actual yields were 87% of target yields, a yield gap of 25.6 bushels/ac (STATUS, 2019). In 2019, 21.3% of corn growers indicated that weather caused them to deviate from their fertility program, 30.6% reporting they responded by decreasing fertilizer application.

The decrease in N rate proposed here is before any decrease in N rate associated with the use of enhanced efficiency fertilizers or precision N management described below and therefore the use of enhanced efficiency fertilizers would be additive and could result in further reductions in N fertilizer rate.

Beneficial Management Practice 3 – Increased adoption of precision nitrogen management

Canola (Prairies)

In 2019, 68.9% of canola producers indicated that they use the same fertilizer program for all of their fields with 11.6% indicating they used variable rate application on all of their fields and an additional 3.2% indicating they used variable rate on some of their fields. A total of 16.3% indicated that their fertility program was "tailored field by field" (STRATUS, 2019). The use of variable rate was similar in the three Prairie provinces (MB 16.7%, SK, 14.7%, AB 13.9%). A larger percentage (26.0%) of large farms (> 4000 ac) or very familiar with 4R practice (27.0%) were utilizing a variable rate fertilizer program on some or all of their fields (STRATUS, 2019).

Corn (Ontario)

In 2019, 55.4% of corn producers in Ontario indicated that they use the same fertilizer program for all of their fields with 7.3% indicating they used variable rate application on all of their fields and an additional 5.3% indicating they used variable rate on some of their fields. A total of 32.1% indicated that their fertility program was "tailored field by field" (STRATUS, 2019). A larger percentage (22.7%) of large farms (> 1000 ac) or very familiar with 4R practice (20.3%) were utilizing a variable rate fertilizer program on some or all of their fields (STRATUS, 2019).

Beneficial Management Practice 4 – Elimination of fall N application

Canola (Prairies)

In 2019, 22.5% of canola growers applied N fertilizer in the fall of the previous year, 43% of those producers applying N in the fall were from Manitoba and many (35.7%) claimed to be very familiar with 4R (STRATUS, 2019). In Manitoba 44.8% of canola acres received synthetic N in the fall. In Saskatchewan 20% and in Alberta 15.1% of acres received synthetic N fertilizer in the fall (STRATUS, 2019).

In 2019, a total 19.8% of the total amount of synthetic N fertilizer applied to canola was applied the previous fall. In Manitoba 36.3%, Saskatchewan 18.1%, and Alberta 12.8% of the total N applied to canola was applied in the fall. Of the N applied in the fall, 4.9% was applied as either SuperU or ESN. The majority (56.9%) of N applied in the fall was anhydrous ammonia. The average rate of fall N application was 96.6 lbs N/ac (108 kg N/ha), the highest rate of all timings. Note that this refers to rate of N applied at that time and does not preclude additional N being applied at other times.

In 2019, the STRATUS survey estimated a total of 534,246 lbs (242,330 kg) of the total of 2,697,283 lbs (1,223,467 kg) of N was applied in the fall to canola. This estimate was based on area of land receiving N fertilizer in the fall multiplied by the application rate. In 2019, the average N application rate was 129.7 lbs N/acre (145 kg N/ha).

Corn (Ontario)

In 2019, 2.5% of Corn growers in Ontario applied N fertilizer in the fall of the previous year representing 4% of the N fertilizer used (STRATUS, 2019). Much of the fall N applied was listed as either "other sources" (59.7%) or anhydrous ammonia (24.5%). SuperU or ESN represented only 6.4% of the N applied in the fall.

Beneficial Management Practice 5 – Increased use of enhance efficiency nitrogen fertilizer

Canola (Prairies)

In 2019, 13.5% of canola growers used either ESN or Super U, representing 11.9% of canola acres, and 7.3% of the N fertilizer applied. Note that 19.7% of the N was listed as coming from "other sources", which may have included other enhanced efficiency products. When asked about their use of N stabilizers (other than SuperU or ESN), 6.9% of acres treated with anhydrous ammonia, 8.9% of acres receiving urea, and 7.8% of acres receiving UAN included a stabilizer. Stabilizer was most often (22.4% of acres) used when the N was applied after planting, 12.0% of acres where N was applied in the previous fall receive stabilizer, 6.5% and 6.0% of acres where the N was applied in the spring before planting or at planting respectively (STRATUS, 2019). The percentage of acres treated with N stabilizers was greatest in Saskatchewan (10.7%), followed by Alberta (7.1%), with the lowest use in Manitoba (2.2%). Large farms (10.5% of acres), young producers (10.4% of acres), familiar with 4R practices (10.6% of acres) are the most frequent users of stabilizers.

Corn (Ontario)

In 2019, 9.3% of corn growers in Ontario used either ESN or Super U, representing 13.5% of corn acres, and 5.5% of the N fertilizer applied. The most common timing of SuperU/ESN application was at planting and represented 21.7% of the N applied at that time.

When asked about their use of N stabilizers (other than SuperU or ESN), 29.4% of acres treated with ammonium nitrate, 5.4% of acres receiving anhydrous ammonia, 18.1% of acres receiving urea, and 49.1% of acres receiving UAN included a stabilizer. Stabilizer was most often (19.8% of acres) used when the N was applied after planting, 3.6% of acres where N was applied in the previous fall receive stabilizer, 17.9% when applied at planting, and 8.7% of acres when applied in the spring before planting (STRATUS, 2019). Medium sized (500 – 1000 acres) farms (26.1% of acres), middle aged producers (20.7% of acres), and those only somewhat familiar with 4R practices (26.2% of acres) are the most frequent users of stabilizers.

Animal Manure

Beneficial Management Practice 6 – 4R management of manure

Fall manure application

In 2021, the Ontario Soil and Crop Improvement Association surveyed producers on the management of their manure (Timing Matters, 2021). The survey indicated that while 95% of those surveyed felt that manure was "a valuable resource that should be closely managed", 55% did not consult a professional agronomist, certified crop advisor, nutrient management consultant or nutrient management calculator in making their manure management decisions. The majority (65%) of manure was applied in the early to late fall after crop harvest, with 46% being applied in the early fall.

Canola (Prairies)

In 2019, 22.5% of Canola growers applied manure, 83% of them applied manure in the fall. Many (52%) were large farms, young (43%), and not familiar (43%) with 4R practices.

Corn (Ontario)

In 2019, 45% of Corn growers in Ontario applied manure, 75% of them applied manure in the fall. This was an increase from previous years. Many (40.9%) were large farms and young farmers (48.1%). Farmers familiar with 4R practices were as likely as those that were not to apply manure in the fall.

Beneficial Management Practice 7 – Improved crediting of organic N sources

Uncertainty is one of the primary limitations to the crediting of the N content of manure. This uncertainty relates both the composition of manure as well as the plant availability of the N contained in the manure. Often manure management tools such as OMAFRA's AgriSuite use estimated or "book values" of manure composition. These values do not consider site-specific conditions that influence manure composition and often may not even reflect regional differences in animal management, storage conditions or climate impact on the composition of the manure. For this reason, it is recommended that producers test the composition of the manure on an annual basis. Surveys of producers in Ontario suggest this does not occur. Therefore, we propose the cost of manure analysis be subsidized. Further we propose that summary statistics on the results of manure testing for various animal species and manure management be published annually to allow for more contemporary, regionally appropriate data be available for producers who have not tested their manure and to contextualize the results of the test they have performed in terms of regional averages. These statistics could also be used to update the databases that tools such as AgriSuite draw upon. A 50% subsidy on the cost of manure analysis, which typically manure analysis costs ~$50/sample, would result in increased testing and enhance our understanding of the composition of manure being applied.

A 2018 STRATUS survey of Ontario corn grain producers revealed that 26% of corn acres receive manure, mostly in the fall and usually from the farmer's own farm. Those applying manure treat 73% of their land for liquid manure and 61% of land for solid manure. Liquid manure was slightly more common than solid manure. Most of the manure was surface applied and incorporated within 1-2 days of application. On average growers were applying 102 kg N/ha (91 lbs of N/acre) when using liquid manure and 48 kg N/ha (43 lbs N/acre) when applying solid manure. The same survey examined the application of manure to perennial forages in Ontario, revealing that 49% of perennial forage growers apply manure, mostly in the fall or after first cut. Liquid manure users treat 100% of their perennial forage acres, solid manure users treat 66% of their perennial forage acres. Most manure comes from the farmer's own farm. As with corn, there were equal numbers of solid and liquid handling systems. Most of the manure was surface applied and not incorporated. On average growers were applying 84 kg N/ha (75 lbs of N/acre) when using liquid manure and 33 kg N/ha (29 lbs N/acre) when applying solid manure.

In terms of manure management, 35% of manure users in Ontario do not have a manure management plan, only 12% of users use NMNAN software. Almost all users adjust their fertilizer rates based on manure application, but the magnitude of the adjustment was not specified. Manure is most commonly applied once in three years rotating fields to which the manure is applied. Over 60% of users consider soil N levels in deciding where to apply manure. For liquid manure management, 17% of producers tested manure every year, 13% every second year, 11% every third year and 30% every fourth year or less frequently with an additional 30% indicating they did not test their manure. Thus 60% of producers either do not test their manure or do only once in four years. For solid manure management, 5% of producers tested manure every year, 6% every second year, 13% every third year and 32% every fourth year or less frequently with an additional 44% indicating they did not test their manure. Thus 76% of producers either do not test their manure or do only once in four years. Improved soil organic matter was most often mentioned as the primary benefit of manure application, reducing N fertilizer was the most common secondary benefit. Better manure content testing capacity would address the first issue. Better modelling of the mineralization or organic N would address the second.

6. Barriers to adoption

Manure Management

One of the challenges of animal manure management is the cost of transportation. Manure, particularly liquid manure, has a high water content (>90%) and therefore is costly to transport. As a result, to paraphrase a producer I work with, "if you have manure, you have manure; if you don't, you don't". His point is that the difficulty and cost of transportation often limits the opportunity to distribute manure to one's own farm or farm in close proximity. Thus, for those that generate manure, this limits their opportunity to export – they have manure and must manage it. For those that do not generate manure it is often difficult to locate an economically viable manure source – they don't have manure, even if they would like to apply it. Here we do not include the cost of transport in costing and therefore that would have to be considered on a case-by-case basis. The processing of manure through solid separation and composting can help to reduce the costs of transportation and other organic waste sources such as biosolids must be transported providing the opportunity for producers that do not generate manure to add an organic nitrogen source to their land. In this analysis, I focus primarily on the improved efficiency of manure N management within existing operations using manure and not on the use of manure on operations that do not currently apply manure. One of the primary impacts of improved manure N management and crediting is a reduction in the requirement for supplemental inorganic N fertilizer. Improved conservation of manure N and tools to create greater confidence in the N supplying potential of manure will allow producers to reduce their reliance on supplemental N, reducing production costs and environmental impacts.

Nitrogen Applied to Agricultural Soils

Nutrient management is already fundamental aspect of crop production so adoption of 4R principles is not a profound change to the cropping system management. Given the widespread availability of contract services and custom fertilizer applicators to deliver 4R implementation, there would relatively a relatively modest technological or expertise barrier to adoption. There may be a need for additional training and certification of agronomists to deliver these services but 4R management is rapidly becoming standard curriculum for certification processes for nutrient management planners and certified crop advisors. The financial resources for producers to engage independent agronomists may present an economic barrier.

Beneficial Management Practice 2 – Quantitative determination of Right Rate

There are additional measurements specific to nitrogen management that will add cost to 4R implementation. These include the measurement of the nitrogen supplying capacity of the soil and the amount of nitrate remaining in the soil following harvest. These practices are necessary for a measurement-based determination of the right rate of N fertilizer use in intermediate and advanced implementation of 4R N practices and the validation of the success of 4R implementation in reducing the potential for N loss. Soil nitrate measurements are not currently routinely done in all regions of Canada and the commercial laboratory capacity to conduct these measurements may need to be improved to ensure the success of the program. There is a need to develop standardized approaches and the commercial laboratory capacity to conduct these measurements.

Beneficial Management Practice 3 – Increased adoption of precision nitrogen management

One of the major barriers to adoption is the soil sampling required for the development of prescriptions maps and the equipment needed to apply variable rates of N according to geographic position. A less sophisticated implementation of precision N management based on management zones can help to overcome some of these limitations but may also not result in as great an improvement in N management and/or N_2O emissions reduction. Uncertainty as to the magnitude of N_2O emissions reduction associated with the implementation of precision N management and the variation in these reductions from season to season also represent a barrier to adoption.

Beneficial Management Practice 4 – Elimination of fall N application

Often synthetic nitrogen sold at a reduced price in the fall encouraging producers to apply it at that time. In addition, spring can be a very busy time of year encouraging producers to shift as much field activity to the fall as possible. While there is ample evidence of the inefficiencies associated will fall N application, it is tempting to simply apply more N in the fall to offset those efficiencies. The discounting of fertilizer N represents a barrier to increasing the efficiency of N use in agriculture.

Beneficial Management Practice 5 – Increased use of enhance efficiency nitrogen fertilizer

The greatest barrier to use of enhanced efficiency fertilizers is the increased cost of the product. While it is anticipated these costs will come down as the market grows and become smaller as the cost of N increases, this still represents a barrier. Reducing the rate of N recommended when using enhanced efficiency N sources, reflecting their increased efficiency, would help to offset the increased cost of these N sources.

7. Changes needed in emissions reporting

In developing and implementing BMPs for improved nitrogen management it is important to consider how or if these measures will be reflected in our National Inventory Report (NIR) on GHG emissions. While actions that are not captured in the NIR would none-the-less reduce GHG emissions, they would not advance our goal of achieving a reduction in reported GHG emissions. Thus, one of the critical points in the development of these BMPs is a consideration of how these activities can be captured and reported such that they appear in the NIR. There is hope that progress can be made on this front as the 2022 NIR states that "Canada plans to develop more robust ratio factors to account for these [nitrogen management] mitigation measures in the medium-term of three to 5 years as research results and activity data become available". The availability of activity data may be the most critical issue in this regard. There is clearly a need for a joint industry-government initiative to collect activity data with respect to N management that can be used to better characterize the efforts to improve N management. Other sectors have succeeded in introducing "traceability" measures to document the integrity of products as they go to market. These models should be explored in an effort to document the sectors actions to improve N management and their resultant impact on greenhouse gas emissions.

8. Co-benefits

It is important to recognize that improved N management practices also result in significant co-benefits related to reduce N losses. This includes reduced NH_3 volatilization, which can have adverse impacts on surrounding ecosystems, reduced NO_3^- leaching to groundwater which is a major concern in a number of provinces and increases in soil carbon storage and soil health. These are all issues that current face the agriculture sector in Canada and so efforts to improve N management will provide multiple co-benefits and may help to motivate producers to adopt these practices.

Ammonia volatilization is primarily the result of ammonia-based N fertilizers and manures being left on the soil surface, exposed to the atmosphere. 4R practices that delay the rate of ammonia formation (urease inhibitors) or place the N source in the soil (right place) can reduce these emissions. Practices which delay the conversion of NH_3 to NO_3^- can increase the potential for NH_3 emissions and raise the potential "pollution swapping" (Drury et al., 2017). The distribution of NH_3 emissions from fertilizer (Fig. 21) reflects the use of higher rates of ammonium-base N fertilizers and animal manures in agriculture, with greatest emissions in Eastern Canada.

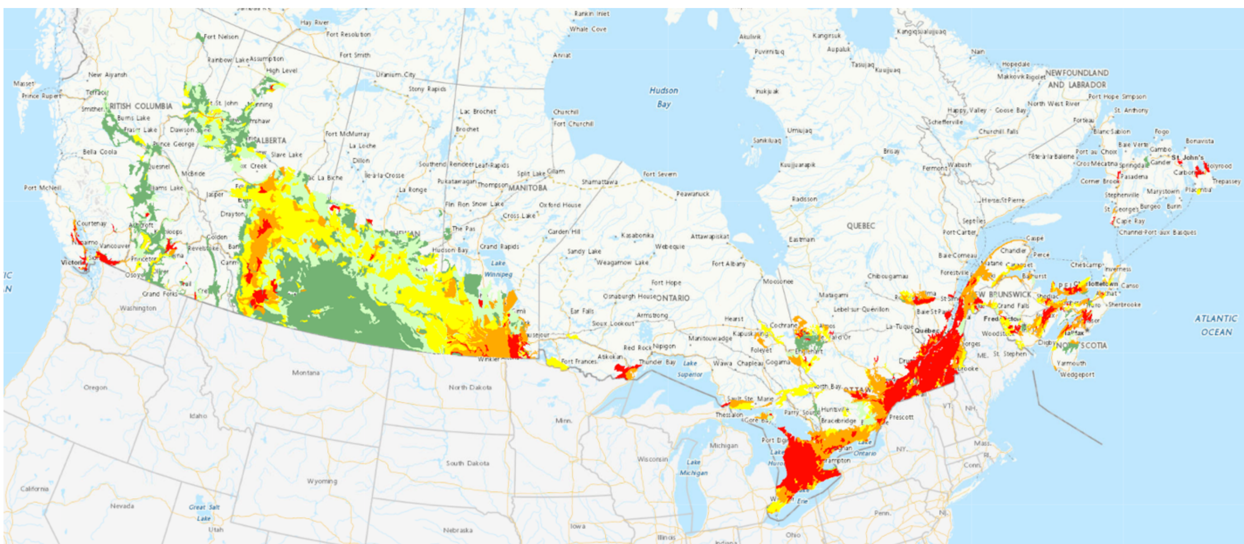


Figure 21: Ammonia emissions from agriculture[2].

Another important impact of N use in agriculture is the increased potential for NO_3^- leaching to groundwater. In more humid regions of Canada, where annual precipitation exceeds annual evapotranspiration there is annual recharge of groundwater sources, primarily during the non-growing season. This recharge of groundwater also has the potential to carry contaminants to the groundwater should they be allowed to accumulate in the soil prior to periods of recharge. Thus, the potential for NO_3^- contamination of groundwater is a product of the timing of the

[2] Source : https://open.canada.ca/data/en/fgpv_vpgf/cc0aadbf-f5e6-41f2-8877-84469bb76076

recharge of groundwater and the timing and magnitude of NO_3^- accumulation in the soil. The Agri-Environmental Indicator of the Risk of Water Contamination by Nitrogen (Fig. 22) represents the potential for excess nitrogen to impact water. Its calculation is based, in part, on an estimation of Residual Soil Nitrogen, the difference between estimated N inputs and N outputs.

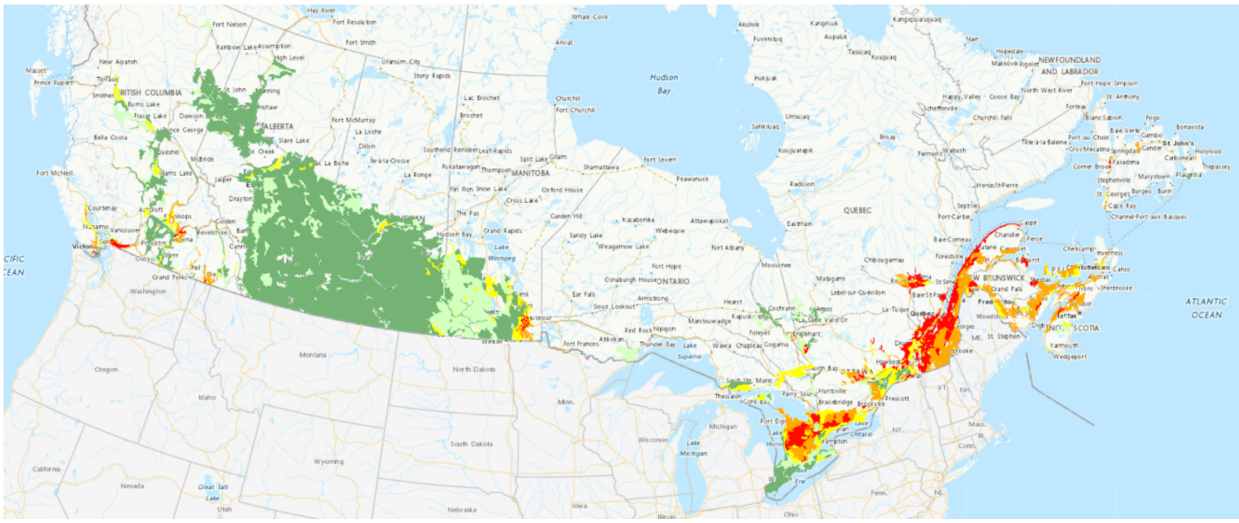


Figure 22: Risk of water contamination by nitrogen[3].

Increasingly Canadians are becoming aware of the importance of the carbon content of our soils and the opportunity to improve the health of our soils by increasing soil organic matter content. The decline in soil carbon content is particularly severe in Eastern Canada (Fig. 23), but soil organic carbon content in Prairie Canada is no longer increasing at the rate it was in 2011. Soil health and soil resiliency are areas of increasing interest/concern for Canadian farmers as climate variability challenges our production systems. Effective N management can help to sustain increases in soil organic matter and results in increased soil health and resiliency in the face of climate change. It is difficult to assign a value to this co-benefit.

[3] Source : https://open.canada.ca/data/en/fgpv_vpgf/8f96099a-cb27-45fb-986b-5fdb5f3b1828

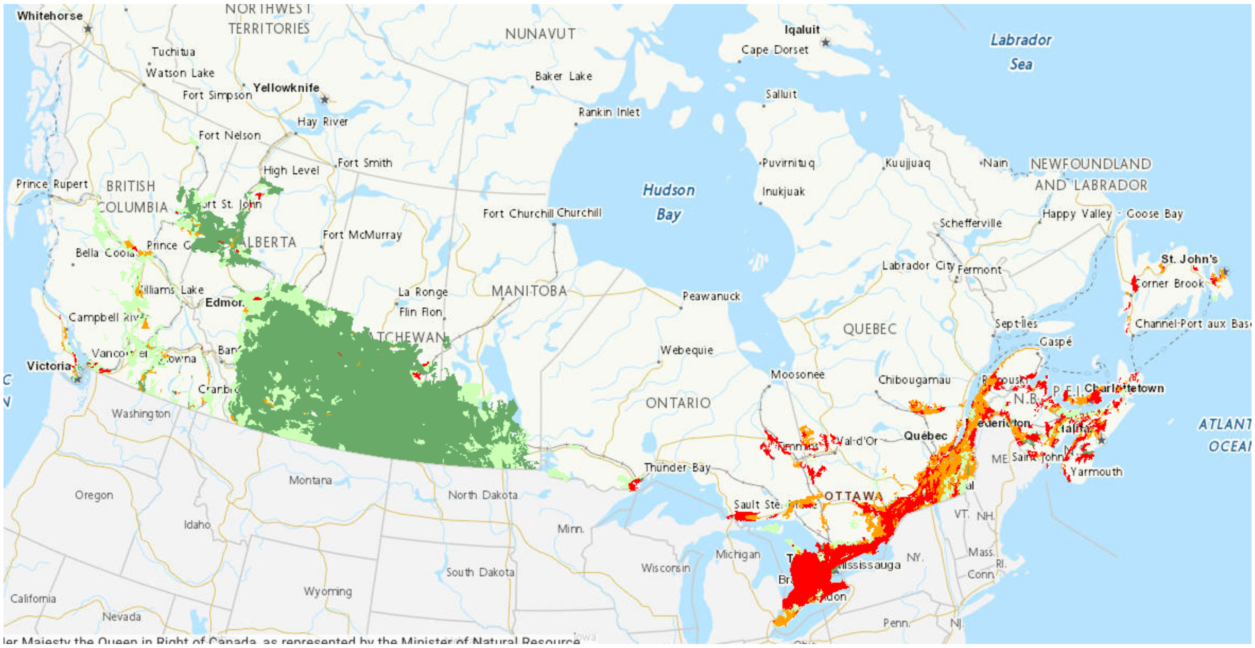


Figure 23: Change in soil organic carbon content of Canadian agricultural soils in 2011 as reflected in Canadian Agri-Environmental Indicators (Clearwater et al., 2016)[4].

[4] Source: https://search.open.canada.ca/openmap/3da8d43c-83a4-436f-abbd-d370a953b187

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Greenhouse Gas Mitigation Potential for Manure Management Systems in Canada

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1. Description of emissions source and mechanisms

Storage of livestock manure is necessary to enable the application of manure at an appropriate time to supply nutrients to crops. In Canada, livestock producers typically use liquid slurry storages or solid manure storages and drylot for storing manure until land application. Manure excreted on pastures and paddocks by grazing animals are not managed. Both CH_4 and N_2O are emitted during the handling and storage of livestock manure and the magnitude of emissions depends on the quantity and the duration of manure stored, manure characteristics which in turn are related to livestock type and nutrition, and the type of manure management system (MMS) used. Liquid slurry manure systems are less aerated; thus, anaerobic conditions in these systems generate high amounts of CH_4 emissions but relatively low N_2O emissions. In contrast, solid manure systems are relatively well-aerated and generate high amounts of N_2O emissions but relatively low CH_4 emissions. The proportion of manure stored using different management systems vary regionally and by animal category (Fig S1). In Canada, dairy and swine producers store large proportion of their manure using liquid systems while beef and poultry producers mainly use solid manure systems, whereas low-density pasturing systems such as beef cow-calf systems result in widely dispersed manure on pasture and paddocks. Other livestock, such as sheep, goats, bison, horses, llamas/alpacas, deer and elk, wild boars, and mules and asses, are generally raised in pasture and/or medium-density production facilities producing mainly solid manure.

For the Canadian National Greenhouse Gas emissions inventory, CH_4 and N_2O emissions from manure management are estimated for each animal category/subcategory by multiplying its population by the corresponding emission factor (amount of each GHG emitted per animal per year). The animal population data are collected from annual and /or biennial surveys corrected using the Census of Agriculture every five years. Annual emission factors are estimated using the IPCC Tier 2 methodology (IPCC, 2006) using country specific livestock production parameters derived from Canadian research and expert consultations.

2. Quantities of emissions under consideration and trendlines

In 2019, the livestock sector in Canada contributed about 7.9 Mt CO_2eq emitted from manure management systems representing 14% of the national agricultural greenhouse gas (GHG) emissions (ECCC 2021). Of these emissions, 49% was CH_4 and 42% was direct N_2O emissions while the remaining 9% was indirect N_2O emissions resulting from ammonia (NH_3) volatilization and leaching/runoff losses from manure handling and storage systems.

The management of beef and poultry manure contributed mainly to N_2O emissions, whereas the management of swine manure contributed mainly CH_4 (Fig 1). Emissions from dairy manure have shifted from mainly N_2O to mainly CH_4 due to changes in manure storage practices such as switching from solid systems to liquid systems accompanied by dairy farm consolidation (ECCC 2021). When the GHG contributions from different livestock types are considered, the four major livestock types (dairy cattle, beef cattle, swine, and poultry) contributed 98% of the total GHG emissions from manure management systems with all other livestock types contributing the remaining 2% (Fig 2). Therefore, mitigation potential estimated in this report primarily focus on the four major livestock categories mentioned above while acknowledging that other livestock types make important contributions to Canadian economy. Mitigation strategies applicable to beef solid manure systems may also be suitable for other livestock whose manure is largely managed as solid manure.

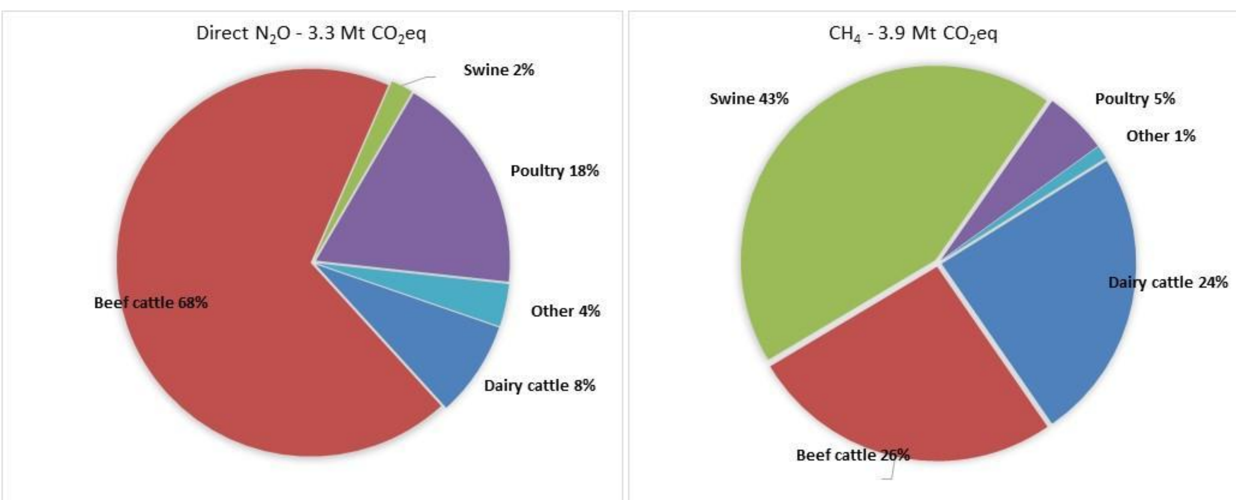

Fig 1. Proportional contribution of CH_4 and N_2O emissions from manure management by different livestock types in Canada in 2019. About 0.7 Mt CO_2eq of indirect N_2O emissions

resulting from ammonia emissions and N leaching/runoff from manure handling and storage systems are not included in the figure (Data source: National GHG Inventory Report 2021).

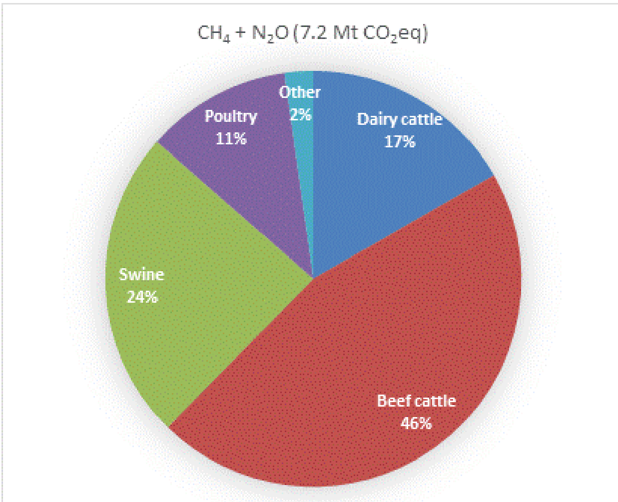

Fig 2. Contribution form different livestock groups to total GHG emissions from manure management systems in Canada in 2019 (excluding 0.7 Mt CO_2eq of indirect N_2O emissions). (Data source: National GHG Inventory Report 2021).

The long-term trend in CH_4 emissions from manure management closely follows the changes in the animal population and manure management practices in the swine and dairy sectors, increasing from 2.4 to 3.8 Mt or a 59% increase from 1990 to 2019 (Fig 3). In contrast, N_2O emissions from manure management corresponds to the trend in the beef cattle population, increasing from 2.9 Mt in 1990 to 3.7 Mt (32% increase) in 2005 and subsequently declining to 3.2 Mt (overall 9% increase from 1990 to 2019) (Fig 3).

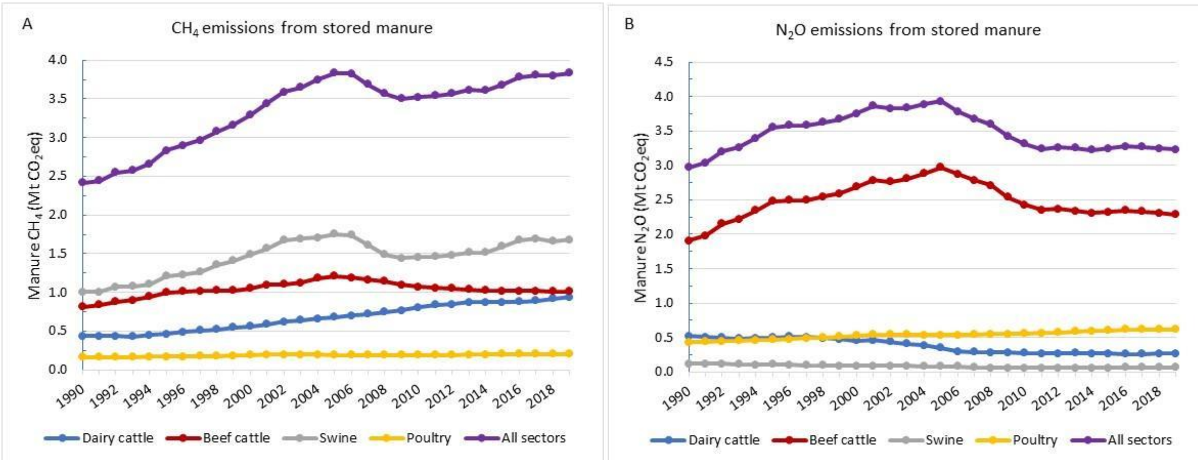

Fig 2. Long-term trends in CH_4 and N_2O emissions from manure management systems in four major livestock groups in Canada. (Data source: National GHG Inventory Report 2021).

3. Provincial level breakdown of GHG emissions from manure management systems

Greenhouse gas emissions from manure management systems in the four major livestock types at the Provincial level are presented in Tables 1 to 4. The provincial level disaggregated emissions data show that Quebec and Ontario contributed the largest share (69% %) of the GHG emissions from dairy manure management (Table 1) whereas the three prairie provinces (Alberta, Saskatchewan, and Manitoba) contributed the largest share (77%) of the GHG emissions from beef manure management (Table 2). In the swine sector, 80% of the GHG emissions from manure management were contributed from Quebec, Ontario, and Manitoba (Table 3), while 83% of the emissions from poultry manure management originated from Ontario, Quebec, British Columbia, and Alberta (Table 4).

The proportional contributions of GHG emissions from different manure management systems in different provinces varied appreciably, however, general trends were similar (Figures 3 to 6). These disaggregated data indicate the following key points which would assist in identifying appropriate mitigation measures for a particular region, livestock sector, or what type of gas needs to be targeted for mitigation.

- 72% of the total GHG emissions from dairy manure management was CH_4 (Table 1), of which >95% was emitted from liquid slurry storages (Fig 3).

- 65% of the total GHG emissions from beef manure management was N_2O (Table 2), of which 97% was emitted from solid storage and drylots (Fig 4).
- 91% of the total GHG emissions from swine manure management was CH_4 (Table 2), of which 99% was emitted from liquid storages (Fig 5).
- 63% of the total GHG emissions from poultry manure management was N_2O (Table 4), of which 99% was emitted from solid manure storages (Fig 6).

Table 1. Provincial level GHG emissions from dairy manure management systems (MMS) in Canada in 2019. (Data source: National GHG Inventory Report 2021).

Province	Dairy cattle Population[1] (1000 heads)	CH_4 (kt CO_2eq/yr.)	direct N_2O (kt CO_2eq/yr.)	Indirect N_2O (kt CO_2eq/yr.)	Total GHG from MMS (kt CO_2eq/yr.)	% of total	Average Emission Factor (kg CO_2eq/ head/ yr.)
Alberta	121.4	98.6	36.6	5.8	141.0	9%	1162
British Columbia	125.5	113.9	24.4	6.3	144.7	9%	1153
Manitoba	60.2	40.3	16.2	2.6	59.1	4%	982
New Brunswick	26.8	18.5	5.5	1.1	25.1	2%	937
Newfoundland	8.0	8.9	2.4	0.4	11.7	1%	1469
Nova Scotia	30.4	23.0	6.1	1.1	30.3	2%	995
Ontario	472.4	363.7	160.8	27.0	551.5	35%	1167
Prince Edward Island	20.8	10.9	5.3	1.0	17.2	1%	831
Quebec	504.1	417.4	91.1	19.1	527.7	34%	1047
Saskatchewan	39.1	29.2	15.1	2.3	46.7	3%	1194
National	1408.5	1124.4	363.6	66.8	1554.8		1104

[1]Dairy cows and dairy heifers.

Table 2. Provincial level GHG emissions from beef manure management systems (MMS) in Canada in 2019. (Data source: National GHG Inventory Report 2021).

Province	Beef cattle Population[1] (1000 heads)	CH_4 (kt CO_2eq/yr.)	direct N_2O (kt CO_2eq/yr.)	Indirect N_2O (kt CO_2eq/yr.)	Total GHG from MMS (kt CO_2eq/yr.)	% of total	Average Emission Factor (kg CO_2eq/ head/ yr.)
Alberta	4728.7	345.1	1022.2	169.1	1536.3	46%	325
British Columbia	522.0	39.1	70.8	10.9	120.8	4%	231
Manitoba	992.4	64.3	144.5	21.6	230.5	7%	232
New Brunswick	39.5	3.9	7.2	1.2	12.3	0%	311
Newfoundland	3.3	0.5	0.4	0.1	1.0	0%	299
Nova Scotia	46.1	5.0	7.9	1.4	14.3	0%	310
Ontario	1132.8	96.5	274.2	43.0	413.7	12%	365
Prince Edward Island	40.0	2.3	8.3	1.2	11.9	0%	298
Quebec	630.9	104.4	92.1	17.0	213.6	6%	338
Saskatchewan	2355.9	164.9	563.7	86.9	815.5	24%	346
National	10491.5	826.0	2191.3	352.5	3369.8		321

[1]Beef cows, bulls, beef replacement heifers, Heifers for Slaughter, steers, and calves

Table 3. Provincial level GHG emissions from swine manure management systems (MMS) in Canada in 2019. (Data source: National GHG Inventory Report 2021).

Province	Swine Population (1000 heads)	CH_4 (kt CO_2eq/yr.)	direct N_2O (kt CO_2eq/yr.)	Indirect N_2O (kt CO_2eq/yr.)	Total GHG from MMS (kt CO_2eq/yr.)	% of total	Average Emission Factor (kg CO_2eq/ head/ yr.)
Alberta	1505.0	192.2	4.0	11.6	207.8	11%	138
British Columbia	86.5	9.2	1.1	0.6	11.0	1%	127
Manitoba	3385.0	328.0	15.6	30.7	374.4	20%	111
New Brunswick	29.3	2.7	0.3	0.2	3.2	0%	111
Newfoundland	0.8	0.1	0.0	0.0	0.1	0%	115
Nova Scotia	15.2	1.5	0.3	0.1	1.9	0%	123
Ontario	3621.0	453.5	13.6	29.5	496.6	27%	137
Prince Edward Island	37.3	3.9	0.5	0.3	4.7	0%	127
Quebec	4342.5	568.4	18.1	32.1	618.6	33%	142
Saskatchewan	955.0	118.6	8.3	7.1	134.1	7%	140
National	13977.5	1678.2	62.0	112.2	1852.4		133

Table 4. Provincial level breakdown of GHG emissions from poultry manure management in Canada in 2019. (Data source: National GHG Inventory Report 2021).

Province	Poultry Population [1] (1000 heads)	CH_4 (kt CO_2eq /yr.)	direct N_2O (kt CO_2eq / yr.)	Indirect N_2O (kt CO_2eq/yr.)	Total GHG from MMS (kt CO_2eq/yr.)	% of total	Average Emission Factor (kg CO_2eq/ head/ yr.)
Alberta	14893.9	38.4	46.6	12.8	97.7	10%	6.6
British Columbia	21484.2	16.7	87.2	21.2	125.2	13%	5.8
Manitoba	9000.2	31.0	31.7	9.0	71.6	7%	8.0
New Brunswick	2936.9	3.6	10.7	2.7	17.0	2%	5.8
Newfoundland	1890.0	2.1	6.2	1.5	9.8	1%	5.2
Nova Scotia	4918.5	8.3	17.4	4.6	30.3	3%	6.2
Ontario	54532.1	44.5	246.4	59.2	350.2	36%	6.4
Prince Edward Island	552.4	0.4	2.2	0.5	3.0	0%	5.5
Quebec	37639.1	53.9	142.4	35.6	231.9	24%	6.2
Saskatchewan	6096.1	5.7	22.3	5.4	33.5	3%	5.5
National	153943.5	204.5	613.1	152.7	970.3		6.3

[1]Include: turkeys, Hens, Pullets, Broilers

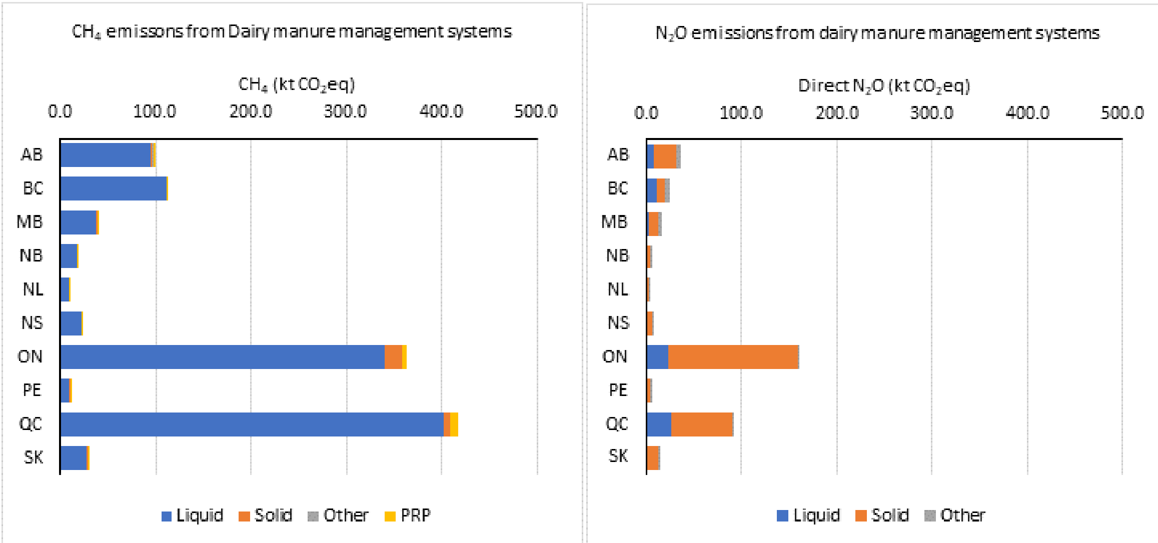

Fig 3. GHG emissions from different manure management systems in the dairy cattle sector in different Provinces of Canada in 2019. (Data source: National GHG Inventory Report 2021).

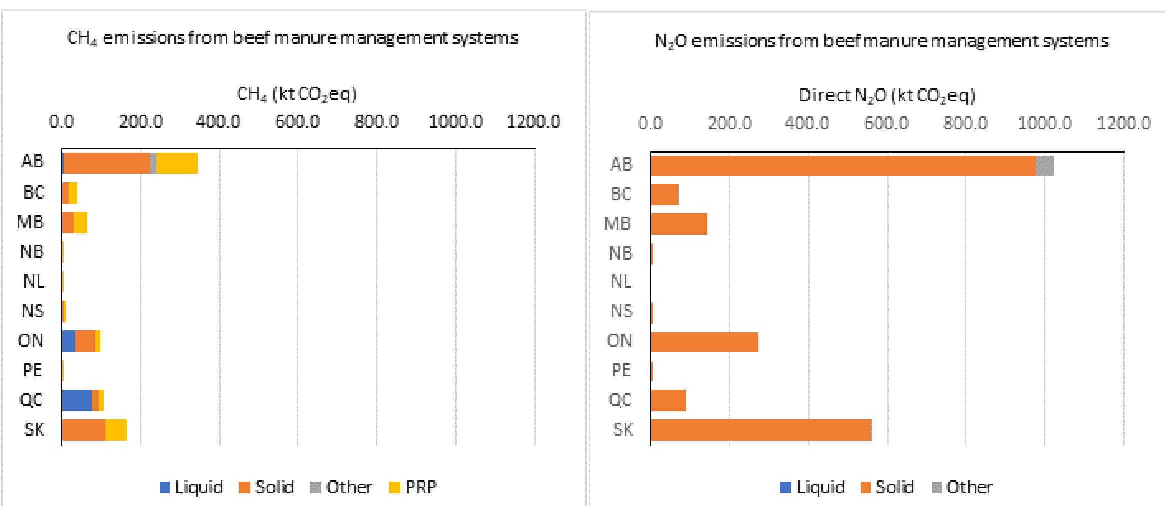

Fig 4. GHG emissions from different manure management systems in the beef cattle sector in different Provinces of Canada in 2019. (Data source: National GHG Inventory Report 2021).

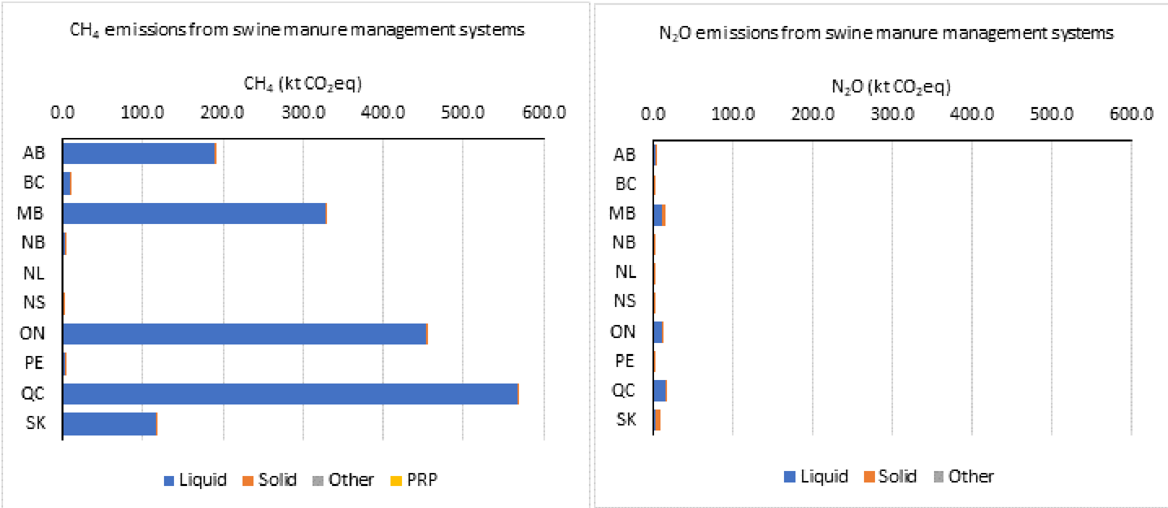

Fig 5. GHG emissions from different manure management systems in the swine sector in different Provinces of Canada in 2019. (Data source: National GHG Inventory Report 2021).

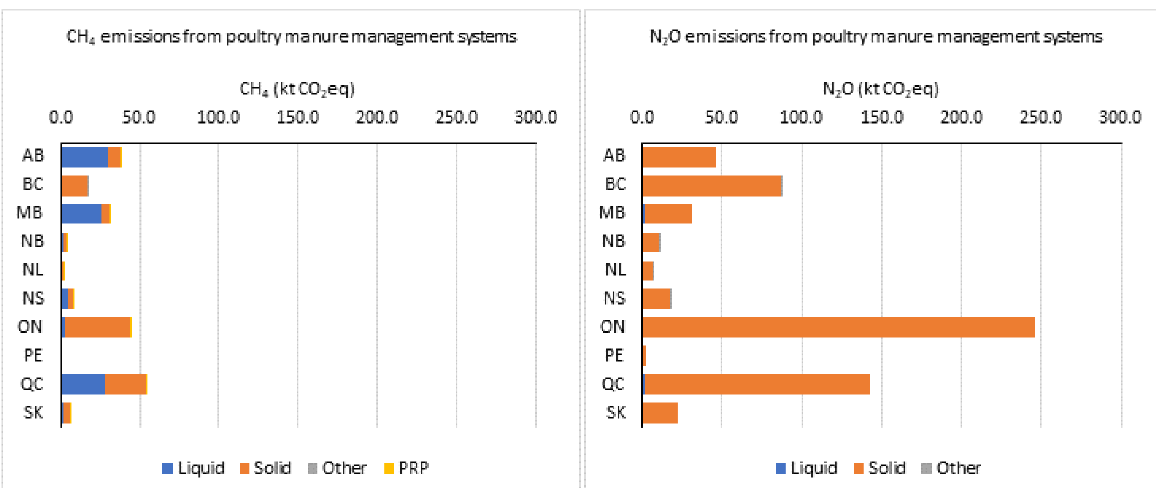

Fig 6. GHG emissions from different manure management systems in the poultry sector in different Provinces of Canada in 2019. (Data source: National GHG Inventory Report 2021).

4. Strategies for reducing CH_4 and N_2O emissions from Manure Management Systems

Based on the existing knowledge and practices and new and emerging research conducted in Canada, seven specific management options are considered in this report for reducing CH_4 and N_2O emissions from manure management systems. These options are: (1) Solid-Liquid Separation, (2) Synthetic Impermeable Floating Covers, (3) Complete Emptying of liquid manure tanks prior to the onset of summer warm season (hereafter referred to as 'Complete Emptying', (4) Anaerobic Digestion, (5) Slurry Acidification, (6) Composting with Turning and Passive Aeration, and (7) Covering solid manure stack with synthetic cover. Of these seven options, the first five options focus on reducing emissions from liquid manure systems in the dairy and swine sectors whereas the last two mitigation measures (composting with turning and passive aeration and covering manure stack with synthetic cover) focus on reducing emissions from solid manure, especially in the beef cattle sector. A description of the proposed management options including mechanisms of GHG reduction actions and GHG emission reduction factors for these mitigation measures are explained below, followed by quantification of possible emission reductions for the four major livestock groups in Canada. Recognizing that attempts to abate emissions of one gas sometimes may lead to increases in emissions of another gas, the impact of a given mitigation measure on the emissions of CH_4, N_2O, and NH_3 emissions were considered when estimating the GHG mitigation potential using a particular strategy. Although NH_3 is not a greenhouse gas, volatilization of NH_3 and re-deposition can contribute to indirect N_2O emissions (IPCC 2006). The net change in total GHG emissions (CH_4, Direct N_2O, and indirect N_2O) resulting from the implementation of a given mitigation measure was expressed in CO_2 equivalents using the 100-yr global warming potential factors of 25 kg of CO_2eq kg^{-1} of CH_4 and 298 kg of CO_2eq kg^{-1} of N_2O (IPCC, 2007).

4.1 Solid-Liquid Separation:

The process of methane generation under liquid manure storage conditions requires the carbon contained in manure solids as substrate under anaerobic conditions. By separating a portion of these solids from liquid slurry, a solid separator system can reduce methane emissions from the existing liquid manure storage system. Solid-Liquid separation results in a solid fraction rich in dry matter (DM) and nutrients, and a liquid fraction relatively low in both DM and nutrients. Calculation of emissions change should include emissions from both fractions to obtain an overall impact of solid separation on gaseous emissions. Solid-liquid separation has additional benefits to a farm's nutrient management plan. For example, (a) the lower volume and weight of 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 over-loading on crop lands within the farm, and (b) solid separation could potentially increase the storage capacity for the liquid fraction (Møller et al. 2000).

Methane and N_2O emissions from the separated solid and liquid fractions during storage can behave differently. For example, total N_2O emissions during the storage of separated solid and liquid fractions could increase significantly compared with N_2O emissions from the untreated slurry, largely due to higher N_2O emissions from the separated solid fraction due to the existence of aerobic/anaerobic zones in the stored solids, similar to conditions existing in solid manure piles (Fangueiro et al. 2008). In contrast, the total CH_4 emissions from the two fractions (separated solids and liquids) could be reduced by about 32 - 39% relative to the emissions of CH_4 from the untreated slurry (Fangueiro et al. 2008, Kupper et al 2020). Since CH_4 constitutes more than 90% of the total CO_2 equivalent emissions from untreated slurry, a significant net decrease of the sum of two gases (N_2O + CH_4) can be achieved by solid-liquid separation when compared with emissions from untreated slurry. Although a significant increase in NH_3 emissions from the stored solid fraction has been observed relative to emissions from untreated 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. 2008a).

4.2 Synthetic Impermeable Floating Covers

This strategy involves creating a barrier over the free surface of liquid slurry by covering it either with fixed structures or floating material. Early recommendations for the use of manure storage covers have been mainly for mitigating odors and NH_3 emissions from manure (Vanderzaag et al. 2008). Average NH_3 emission reductions due to different type of covers range from 50-90% for stored cattle and swine slurry (Kupper et al 2020).

Two types of covers can be used in manure storages: (a) natural covers, and (b) artificial covers. Natural covers can be materials such as straw or wood chips purposely added to the surface of storage to encourage a crust formation whereas artificial covers can be impermeable or permeable (e.g., plastic sheets and plastic fabrics, Vanderzaag et al. 2008). 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 un-covered 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. The effectiveness of permeable natural covers in mitigating GHG emissions from liquid manure is not always consistent due to 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 (Kupper et al 2020).

In contrast, impermeable covers can predictably prevent GHG emissions from 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). A review by Kupper et al. (2020) evaluating the effect of various types of covers on CH_4 and N_2O emissions from dairy and swine slurry reported up to 68% reductions of N_2O and up to 62% reductions of CH_4 with the use of synthetic floating covers such as plastic fabrics or plastic sheets.

4.3 Complete Emptying of Tanks

When manure is removed for application to crops, and if some of the old manure is left in the storage, this old manure can act as an inoculum for the rapid re-establishment of the methanogen population and CH_4 emissions from newly added manure (Sommer et al. 2007; VanderZaag et al. 2011b; Wood et al. 2014). Previous studies with pilot scale outdoor manure tanks have noted > 2month long lag phases of low rates of CH_4 emissions before the onset of higher CH_4 fluxes when manure was loaded into empty storage structures (VanderZaag et al. 2008, 2009, 2010b; Wood et al. 2012). Masse´ et al. (2008) used modeling to extend the results from pilot scale studies and recommended more frequent and complete emptying of manure storages as an effective management practice to reduce CH_4 emissions. Using pilot scale concrete outdoor 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 newly added dairy manure over a 155-d long summer storage period in Nova Scotia, Canada. The absence of an inoculum (complete emptying) significantly reduced CH_4 emissions by 56 % compared to partially emptied (inoculated) tanks, while there was no significant difference in N_2O emissions. Although NH_3 emissions were significantly increased from un-inoculated manure storage due to slower crust formation, which would need additional NH_3 abatement measures, a significant 49 % reduction in total greenhouse gas (GHG) emissions was observed because the overall GHG budget (as CO_2-eq) was dominated by CH_4.

4.4 Anaerobic Digestion

Anaerobic digestion (AD) is a process that promotes degradation of organic matter in biological wastes in the absence of oxygen using microbial consortia composed of hydrolytic and fermentative bacteria (Massé et al. 2011). This process stabilizes 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. GHG mitigation in this context refers to the reduction of total GHG emissions by transitioning from liquid manure management system to a digester system and capturing biogas and combusting it for energy generation which convert CH_4 into CO_2 during the combustion. When designed and operated properly, ensuring against gas leakages from the digester system and emissions from the digested slurry, AD offers one of the most effective methods for reducing CH_4 emissions from manure management (Frear et al. 2011). Additionally, AD offers other benefits such as reducing pathogens and odor in manure. By life cycle analysis (LCA) method with data derived from pilot scale digesters, several studies have demonstrated up to 80% reductions in CH_4 emissions relative to conventional liquid manure management after implementation of AD systems in dairy farms. (Kaparaju and Rintala 2011; Maranon et al. 2011; Battini et al. 2014).

The effectiveness of an AD system for GHG mitigation is influenced by two major factors: uncontrolled leakages of CH_4 from the anaerobic digester (fugitive CH_4 emissions) and CH_4 emissions from the stored digestate (Moller et al. 2009). Fugitive CH_4 emissions can occur (a) at times when the reactor is opened for maintenance, (b) leaking from valves and fittings, (c) intentional release of biogas through safety valves due to over-pressure, (d) emissions associated with the use of biogas (burning in the engine or upgrading), and (e) inefficiency in the flare operation (Moller et al. 2009). In a study involving 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. The other critical factor influencing the effectiveness of a farm digester for GHG mitigation is the magnitude of emissions from stored digestate. When Hydraulic retention time (HRT) during the AD process is not optimized, volatile solids (VS) in manure may not be properly degraded, so that only part of the CH_4 generating potential of the 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. Therefore, optimizing digester performance is important for improving the effectiveness of AD system for GHG mitigation. Furthermore, chemical changes during the VS digestion process such as increased NH_4+ concentration and pH in the digestate relative to raw manure can increase NH_3 emissions during the digestate storage. Additional measures, such as covering the digestate storage tank may be useful to avoid digestate becoming a significant source of GHG in a AD system (Amon et al. 2006; Clemens et al. 2006).

Kariyapperuma et al. (2018) conducted a life cycle analysis of an AD system in an Ontario dairy farm with 190 dairy cows and 41 dairy replacement heifers using the measured GHG emissions data and digester performance parameters collected over 4-yr period. Before the installation of the digester, the farm managed its manure according to conventional liquid dairy manure management practices generally used in Ontario. Total measured GHG emissions from this conventional liquid manure management system over one year were compared with measured GHG emissions after operating the digester for two subsequent one-year cycles of co-digestion using two ratios of dairy manure and industrial food waste: scenario 1: 55% dairy manure: 45% industrial food waste, scenario 2: 68% dairy manure: 32% industrial food waste. The sum of CH_4 emissions from the digester (fugitive emissions) and the storage of digestate were reduced by 71% with scenario 2 compared with emissions from the conventional manure management (raw manure storage), while CH_4 emissions with scenario 1 were significantly higher than the emissions from conventional manure management, mainly due to higher fugitive CH_4 emissions from the digester (DeBruyn et al 2020). Direct N_2O emissions were not significantly different between the conventional manure management and either co-digestion scenarios, however, NH_3 emissions from the digestate storage increased by about 50%. Total GHG emissions under co-digestion scenario 2 (68% dairy manure: 32% industrial food waste) were 70% lower than those under conventional liquid dairy manure management, because indirect N_2O emissions due to NH_3 volatilization was less than 2% of the total GHG budget. These results highlight the importance of optimizing digester operating conditions, including the optimal ratios of substrate (dairy manure: industrial food waste) when anaerobic co-digestion is practiced in dairy farms for realizing the GHG mitigation potential of using AD systems.

Although AD could be an effective way to reduce GHG emissions while also generating an additional income for the farm 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 energy sources used for heat and power generation (Frear et al. 2011).

4.5 Slurry Acidification

Acidification of manure slurry with concentrated sulfuric acid (H_2SO_4), a strategy developed for ammonia (NH_3) mitigation in Western Europe, is also known to dramatically reduce CH_4 emissions from stored manure (Peterson et al. 2012; Fangueiro et al., 2015). Slurry acidification decreases NH_3 emissions by shifting the equilibrium between NH_3 and NH_4^+, with low pH favoring NH_4^+ dominance (Peterson et al 2012), whereas CH_4 generation is disrupted following acidification by making the environmental conditions in manure less favorable for methanogens (Peterson et al 2014; Habtewold et al. 2018). Although other acidifying agents (e.g., hydrochloric acid) have been evaluated for slurry acidification, sulfuric acid is the preferred acid, because it causes additional methanogenesis inhibition from sulfur transformations (Peterson et al 2012; Sokolov et al 2020). A recent meta-analysis of the acidification of dairy and swine manure from 37 studies, with 192 measurement events, showed that acidification treatment was very effective in mitigating gaseous emissions from stored manure slurry without pollution swapping: reducing CH_4 emissions by 86%, N_2O emissions by 21%, and NH_3 emissions by 77%, relative to emissions from untreated slurry (Emmerling et al. 2020).

Most manure acidification research has been conducted in Europe and more specifically in Denmark, where 2011 legislation banned the surface application of livestock manure, unless acidified below a pH of 6.4 (Nyord et al., 2013). Commercial technologies for acidification of slurry have targeted slurry in pits below housed animals, but acidification of slurry in storage tanks prior to field application is also practiced. Procedures have been developed for the safe handling of concentrated H_2SO_4, but current applications of the method involve high running costs since often >5 kg tonne^{-1} slurry is needed to achieve the target pH (Leegaard Riis, 2014). Addition of sulfate or S-containing amino acid methionine under experimental conditions, without manipulation of pH, have also reduced CH_4 emissions by more than 50% (Peterson et al 2012), indicating that the inhibition of methanogenesis is at least partly linked to sulphur transformations and not only a function of pH reduction. Therefore, a reduced dose of H_2SO_4 could potentially achieve a substantial reduction of CH_4 emissions. Research relating to slurry acidification in Canada has focused on evaluating lower doses of H_2SO_4 acid for reducing CH_4 emissions from dairy manure to make the slurry acidification strategy cost-effective. For example. Sokolov et al. (2019), using pilot scale outdoor manure storage tanks in Nova Scotia, investigated the impact of two rates of acidification: 1.4 L of 70% H_2SO_4 m^{-3} manure (medium pH) and 2.4 L of 70% H_2SO_4 m^{-3} manure (low pH), compared with untreated manure (no acidification) on GHG emissions from stored dairy manure over 160 days in the warm season. The medium pH manure treatment had on average a pH of 6.5, the low pH manure treatment had on average a pH of 6, and the untreated manure had on average a pH of 7.4. The cumulative CH_4 emissions were reduced by 85%; N_2O emissions were reduced by 75% while NH_3 emissions were reduced by 41% relative to untreated dairy slurry, indicating that substantial reductions in CH_4, N_2O and NH_3 emissions from stored dairy manure were possible with the low dose treatment of H_2SO_4 acid. In a follow-up study, Sokolov et al. (2020) demonstrated that further decreases in the use of H_2SO_4 acids for manure acidification may be possible by acidifying the residual manure (old manure left in slurry storages after removing manure for application to crops) before re-filling manure tanks with fresh manure. These authors measured GHG emissions from outdoor manure storages filled with 20% inoculum (1-year-old manure) and 80% fresh manure with the inoculum treated in three ways: untreated inoculum (control); previously acidified inoculum (1-year prior); and newly acidified inoculum using 70% H_2SO_4 at a rate of 1.1 L m^{-3} manure. Methane emissions were reduced by 77% using newly acidified inoculum and 38% using previously acidified inoculum, compared with the control with untreated inoculum. Ammonia and N_2O emissions were also reduced by 33 and 73%, using newly acidified inoculum and 23 and 50% using previously acidified inoculum, respectively, compared with the control. These results suggest that lower acid rates and acidifying less frequently may still have good GHG mitigation effects while minimizing the use and cost of H_2SO_4 acid.

Initial results with dairy manure indicate that acidification of slurry did not impact N_2O emissions from soil after field application of acidified manure, while offering the benefit of greater plant available N due to reduced NH_3 losses in storage and after field application (Fangueiro et al., 2015b; Fangueiro et al., 2016).

4.6 Composting with Turning and Passive Aeration

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). It is a most frequently used treatment method for solid organic waste management (Pardo et al. 2015). As 55 to 96% of managed beef manure (excluding manure excreted on pasture by grazing cattle) is stored as solid manure among different provinces of Canada, composting may be regarded as a suitable GHG mitigation strategy for the beef cattle producing farms. In fact, the majority of research assessing composting methods and GHG emissions from composting in Canada have focused on solid beef manure in the prairie provinces (e.g. Hao et al. 2001; 2004; 2011). 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). Optimum adjustment of the structure and C/N ratio by adding bulking materials (e.g., straw) together with close monitoring of the process parameters (e.g., moisture, temperature) have potential to reduce GHG emissions while at the same time minimizing N losses to obtain a final product with higher N retention (Pardo et al. 2015).

Greenhouse gases (CH_4 and N_2O) can be generated during composting because of microbial breakdown of organic matter. Oxygen can be depleted quickly within the composting pile creating heterogeneous anaerobic zones near the centre of the pile and progressively increasing aerobic zones towards 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 centre 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 aeration (Gilroyed et al. 2011). A recent meta-analysis of research involving solid livestock manure composting (predominantly dairy and beef cattle manure with few studies involving solid pig manure and poultry manure) indicated 70% reductions in CH_4 emissions and 49% reductions in N_2O emissions by windrow composting with mechanical turning and passive aeration compared with corresponding emissions from stockpiled solid manure (Pardo et al. 2015). However, there was significant increase (~54% increase) in NH_3 emissions with composting which has been attributed to the important role temperature plays on the NH_4^+ - NH_3 (gas) equilibrium (Pardo et al. 2015). Although total GHG emissions (in CO_2 eq. emissions) were reduced by 30% with composting relative to total emissions from stockpiled manure, additional measures to mitigate NH_3 emissions may be necessary to enhance the environmental benefits of composting and to improve the quality of compost by retaining more N in the final product.

Amending livestock manure with certain additives has shown to be effective in reducing NH_3 emissions during composting. For example, pilot-scale composting at Lethbridge Research Centre, Alberta, have demonstrated that mixing Phosphogypsum (PG), a by-product of phosphorus fertilizer production, during feedlot cattle manure composting retained significantly higher levels of N in the final compost than in the cattle manure composted alone, due to lower NH_3 losses (Hao et al. 2005). Similarly, laboratory-scale composting trials in British Columbia have shown that amending poultry manure with additives like zeolites or magnesium salt ($MgCl_2$) and Phosphate salt (KH_2PO_4) reduced NH_3 losses by 40 to 84% % (Kithome and Paul 1999; Zhang and Lau 2007). A meta-analysis involving 105 studies with 303 paired comparisons to quantify the impact of different additives on NH_3 and GHG emissions during composting has demonstrated that on average, additives (e.g., biochar, zeolite, vermiculite, perlite, Mg^{2+} and PO_4^{3-} salts) reduced NH_3 emissions by 44%, while also reducing N_2O emissions by 45% and CH_4 emissions by 68% (Cao et al. 2019).

5. Greenhouse gas mitigation potential with selected mitigation measures

For estimating the possible reductions in GHG emissions from manure management under different rates of adoption of selected mitigation measures for each livestock sector, we first estimated per animal average GHG mitigation potential (kg CO_2eq head^{-1} year^{-1}) for each mitigation measure for each livestock type using the following steps:

(1). Adjustment factors for change in emissions of CH_4, direct N_2O, and indirect N_2O (due to NH_3 volatilization) relative to the emissions from untreated manure resulting from the implementation of a specific mitigation measure were calculated based on the reviewed research explained in the Section 4 above. The derived adjustment factors and key references are presented in Table 5.

(2). These adjustment factors were then applied to the emissions from target manure management system in the target livestock sector to determine the changes in emissions of CH_4, direct N_2O, and indirect N_2O resulting from the change of practice.

(3). The resulting total GHG emissions under the new practice (new scenario) were estimated and the difference in total GHG between the baseline (or 'business as usual') scenario and the new scenario were calculated. This value is the total unconstrained GHG mitigation possible for a given mitigation strategy.

(4) The amount of total unconstrained GHG reduction was then divided by livestock population number in that sector to derive an estimated average per head GHG mitigation potential per year (kg CO_2eq head^{-1} year^{-1}).

The average per animal GHG mitigation potential with selected mitigation measures for the four major livestock groups are presented in Table 6 to Table 9. For both dairy and swine sectors, the largest estimated average per animal GHG reductions were with the use of slurry acidification (714 kg CO_2eq head^{-1} year^{-1} for dairy and 119 kg CO_2eq head^{-1} year^{-1} for swine at the national level), mainly due to large reductions in CH_4 emissions while also reducing NH_3 emissions (therefore, indirect N_2O emissions) substantially (Table 6 and Table 8). This is about 65% reduction relative to the average baseline emission factor for dairy cattle (which is 1104 kg CO_2eq head^{-1} year^{-1}, Table 1) and about 89% reductions relative to the average baseline emission factor for swine (which is 133 kg CO_2eq head^{-1} year^{-1}, Table 3). The average per animal GHG reduction was lowest with solid-liquid separation, since the decreases in CH_4 (32-39% relative untreated manure, Table 5) were partially nullified by increases in N_2O and NH_3 emissions. The average per animal GHG reductions with anaerobic digestion and synthetic impermeable covers were intermediate: 534 and 492 kg

CO_2eq head^{-1} year^{-1} for dairy cattle, representing 48% and 45% reductions relative to the baseline emission factor and 110 and 80 kg CO_2eq head^{-1} year^{-1} for swine, representing 82% and 60% reductions relative to the baseline emission factor at the national level (Table 6 and Table 8).

For the beef cattle sector where most of the emissions are N_2O from solid manure, composting with turning and covering manure stacks with synthetic covers could result in generally similar average per animal emission reductions of 113 and 108 kg CO_2eq head^{-1} year^{-1}, respectively (Table 7), or about 35% reductions relative to the baseline emissions factor (which is 321 kg CO_2eq head^{-1} year^{-1}, Table 2). Similarly, for the poultry sector, composting with turning and covering manure stacks with synthetic covers could result in generally similar average per animal emission reductions of 1.9 and 2.3 kg CO_2eq head^{-1} year^{-1}, respectively at the national level (Table 9). This represented 30 to 35% reduction in GHG emissions relative to the baseline average GHG emissions factor for poultry (which is 6.3 kg CO_2eq head^{-1} year^{-1}, Table 4).

The estimated average GHG reduction per animal for each specific mitigation measure in combination with the number of animals potentially under that specific mitigation measure for a given adoption scenario over a specific period (e.g., 2023 to 2030 period) was used to estimate the possible GHG mitigation achievable. This was done for the four major livestock groups to estimate the possible GHG mitigation for manure management systems in different Provinces of Canada as explained in Section 7.

Table 5. Percentage emission change due to different manure treatment strategies relative to untreated manure (baseline = BL or reference manure storage). Negative values indicate emission reduction, and positive values indicate an increase in emissions due to treatment.

Dairy Slurry

Manure Treatment Method	Reference System (baseline)	**Emission reduction (% from BL)[1]**		
		CH_4	N_2O	NH_3
Solid-Liquid Separation	Untreated liquid manure	-32%	-43%	23%
Impermeable Synthetic Covers	Open Liquid manure	-62%	0%	-77%
Anaerobic Digestion	Untreated liquid manure	-71%	-11%	50%
Slurry Acidification	Untreated liquid manure	-87%	-75%	-41%
Complete emptying	Partial Emptying	-53%	6%	35%

Swine Slurry

Manure Treatment Method	Reference System (baseline)	Emission reduction (% from BL)[1] CH4	N2O	NH3
		CH_4	N_2O	NH_3
Solid-Liquid Separation	Untreated liquid manure	-39%	258%	1%
Impermeable Synthetic Covers	Open Liquid manure	-62%	0%	-77%
Anaerobic Digestion	Untreated liquid manure	-99%	363%	-45%
Slurry Acidification	Untreated liquid manure	-96%	39%	-77%
Complete emptying	Partial Emptying	-53%	6%	35%

Solid Manure

Manure Treatment Method	Reference System (baseline)	Emission reduction (% from BL)[1]		
		CH_4	N_2O	NH_3
Composting with turning and passive aeration	Stockpiled solid manure	-71%	-49%	54%
Covering manure stack with plastic sheet	Open manure stack	-24%	-39%	-60%

[1]References for the emission reduction factors: Solid-Liquid separation for dairy and swine slurries: Kupper et al. (2020); Impermeable synthetic covers: average adjustment factors for dairy and swine slurries from Kupper et al. (2020); Anaerobic digestion for dairy slurry co-digestion: Kariyapperuma et al. (2018); Anaerobic digestion for swine slurry: Kupper et al. (2020); Slurry acidification for dairy slurry: Sokolov et al. (2020); Slurry acidification for swine slurry: Kupper et al. (2020); Complete emptying: (Wood et al. 2014); Composting with turning and passive aeration: (Pardo et al. 2015); Covering manure stack with plastic cover: (Pardo et al. 2015).

Table 6. Estimated average per animal GHG reduction potential for selected mitigation measures for reducing GHG emissions from dairy manure management systems in Canada.

Province	**Solid-Liquid Separation** total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Storage Covers** total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Complete emptying** Total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Anaerobic Digestion** Total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Slurry Acidification** total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr
Alberta	33.3	274	60.9	502	48.7	401	66.8	550	89.9	741
British Columbia	39.1	312	72.0	573	56.6	451	77.8	620	106.2	846
Manitoba	13.6	225	24.5	408	19.7	327	27.0	449	36.4	606
New Brunswick	5.9	220	11.3	421	9.0	335	12.2	455	16.2	604
Newfoundland	3.0	378	5.6	699	4.4	557	6.1	763	8.2	1026
Nova Scotia	7.6	251	14.3	470	11.3	373	15.5	510	20.8	685
Ontario	116.0	246	220.9	468	175.1	371	238.5	505	318.6	674
Prince Edward Island	3.2	156	6.3	303	5.0	241	6.8	327	9.0	432
Quebec	137.0	272	259.2	514	206.7	410	281.6	559	374.9	744
Saskatchewan	9.5	244	17.6	450	14.1	362	19.3	495	25.8	661
National	**368.3**	**261**	**692.5**	**492**	**550.8**	**391**	**751.5**	**534**	**1006.0**	**714**

Table 7. Estimated average per animal GHG reduction potential for selected mitigation measures for reducing GHG emissions from beef manure management systems in Canada

	Composting		**Manure Stack Covers**	
Province	Total GHG mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	total GHG mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr
Alberta	554.1	117	522.0	110
British Columbia	40.0	77	37.5	72
Manitoba	79.3	80	76.2	77
New Brunswick	4.0	100	3.8	95
Newfoundland	0.2	73	0.2	66
Nova Scotia	4.3	94	4.1	90
Ontario	147.5	130	142.9	126
Prince Edward Island	4.4	110	4.3	108
Quebec	49.5	79	47.3	75
Saskatchewan	304.4	129	293.2	124
National	**1187.8**	**113**	**1131.4**	**108**

Table 8. Estimated average per animal GHG reduction potential for selected mitigation measures for reducing GHG emissions from Swine manure management systems in Canada

Province	**Solid-Liquid Separation** total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Storage Covers** total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Complete emptying** Total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Anaerobic Digestion** Total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr	**Slurry Acidification** total mitigation potential (kt CO_2eq/yr)	Per head mitigation potential (kg CO_2eq/head /yr
Alberta	55.8	37	125.5	83	97.1	65	185.6	123	189.3	126
British Columbia	2.1	24	6.0	69	4.6	53	8.0	93	8.9	103
Manitoba	75.2	22	225.7	67	162.3	48	296.1	87	332.5	98
New Brunswick	0.6	20	1.8	60	1.4	46	2.4	80	2.6	90
Newfoundland	0.0	19	0.0	56	0.0	43	0.1	74	0.1	84
Nova Scotia	0.3	21	1.0	63	0.7	49	1.3	83	1.4	94
Ontario	114.6	32	303.0	84	229.2	63	419.1	116	452.4	125
Prince Edward Island	0.9	23	2.5	68	2.0	52	3.4	91	3.8	101
Quebec	142.3	33	375.9	87	288.8	67	521.0	120	562.9	130
Saskatchewan	28.0	29	77.6	81	59.9	63	105.6	111	116.1	122
National	**419.7**	**30**	**1118.9**	**80**	**845.9**	**61**	**1542.6**	**110**	**1670.0**	**119**

Table 9. Estimated average per animal GHG reduction potential for selected mitigation measures for reducing GHG emissions from poultry manure management systems in Canada.

Province	Composting Total GHG mitigation potential (kt CO2eq/yr)	Per head mitigation potential (kg CO2eq/head /yr	Manure Stack Covers total GHG mitigation potential (kt CO2eq/yr)	Per head mitigation potential (kg CO2eq/head /yr
Alberta	22.5	1.5	26.5	1.8
British Columbia	42.1	2.0	50.0	2.3
Manitoba	14.6	1.6	17.6	2.0
New Brunswick	5.1	1.7	6.1	2.1
Newfoundland	3.0	1.6	3.6	1.9
Nova Scotia	8.1	1.7	9.7	2.0
Ontario	118.5	2.2	141.5	2.6
Prince Edward Island	1.0	1.9	1.2	2.2
Quebec	68.9	1.8	81.6	2.2
Saskatchewan	10.8	1.8	12.8	2.1
National	**294.8**	**1.9**	**350.6**	**2.3**

6. Current adoption levels of the selected GHG mitigation measures

Statistics Canada in collaboration with Agriculture and Agri-food Canada collect the farm practices related to manure handling systems and manure treatment methods used in Canadian farms through the Farm Environmental Management Survey (FEMS). The most recent publicly available FEMS report (AAFC 2016) provides the current level of adoption for some manure treatment methods discussed in this analysis. Tables 10 to 15 below present the relevant data on the level of adoption of different manure treatment methods used in Canadian farms as reported in the FEMS report (AAFC 2016). Key observations from this data are explained below.

6.1 Liquid Manure treatment methods

Across Canada, about 50% of farms that store liquid manure did not report any treatments (Table 10), while the most common treatment reported was the aeration or agitation (45%), which is usually done to ensure consistent nutrient content in manure prior to land application, not for GHG mitigation. Thus, 95% of farms that manage liquid manure did not use manure treatment method aiming for GHG mitigation. Of the specific measures for GHG mitigation considered in this analysis, solid-liquid separation was reported by about 4% of the farms across Canada, while anaerobic digestion and CH_4 capture were reported by about 2% of farms. However, anaerobic digestion and CH_4 capture were reported under 'other' practices category which also included 'filtered through marsh or constructed wetland' (Table 10). When the percentage of farms reporting different treatment methods for liquid manure by the livestock sector was considered, solid-liquid separation was reported by about 4.9% of farms in the dairy sector whereas anaerobic digestion and CH_4 capture were reported alone with 'filtered through marsh or constructed wetland' by 1.6% of farms each in the dairy and swine sectors, and 9% of farms in the beef cattle sector (Table 11). It was not possible to discern any regional differences in these practices as much of the regional data were suppressed, perhaps due to limited number of farms reporting these practiced regionally (Table 10 and Table 11). There was no information related to current level of adoption for slurry acidification. We assume this mitigation measure is not practiced in Canada as it is relatively recently developed strategy for GHG mitigation for stored manure.

Table 10. Percentage of farms practicing various liquid manure treatment practices by Province.

Province	Aerated or agitated (%)	Mixed with additives to modify odour, pH or nutrient content (%)	Mixed or turned to accelerate composting [1] (%)	Processed to separate liquid from solid (%)	Other Practices[2] (%)	None (%)
Atlantic	52.7	x	x	x[3]	x	47.0
Quebec	41.6	2.8	3.9	6.1	x	52.5
Ontario	46.2	9.6	3.9	x	2.1	51.1
Prairie	49.0	12.7	11.7	x	9.3	49.1
BC	68.3	9.1	x	x	x	28.8
Canada	**45.4**	**6.6**	**5.3**	**3.8**	**2.3**	**50.3**

[1]This practice is possibly only used for thicker/semi-solid manure that is two thick for agitation.

[2] Other practices include: (a) Filtered through marsh or constructed wetland, (b) digested in an anaerobic system and methane capture.

[3]data suppressed.

Table 11. Percentage of farms utilizing various liquid manure treatment practices by Livestock sector.

Livestock Sector	Aerated or agitated (%)	Mixed with additives to modify odour, pH or nutrient content (%)	Mixed or turned to accelerate compostin g[1] (%)	Processed to separate liquid from solid (%)	Other Practices[2] (%)	None (%)
Dairy	47.4	5.7	5.5	4.9	1.6	49.2
Beef	44.5	x	16.0	X[3]	9.4	46.8
Pork	39.5	9.5	x	x	2.0	55.8
Poultry	55.0	x	x	x	x	32.8

[1]This practice is possibly only used for thicker/semi-solid manure that is two thick for agitation; [2] Other practices include: (a) Filtered through marsh or constructed wetland, (b) digested in an anaerobic system and methane capture; [3]data suppressed.

6.2 Solid Manure treatment methods

Of the farms that managed solid manure, the majority did not practice any treatment for solid manure (73% of farms at the national level, Table 12). The most common treatment practice reported was mixing or turning to accelerate composting, about 24% of farms at the national level. There were appreciable regional differences in percentage of farms practicing composting, ranging from about 10% of the farms in Ontario and Quebec to about 44% of farms in Manitoba. Other practices such as mixing additives to modify odour, anaerobic digestion, and combinations of various treatments occurred on very few farms (3%). When the percentage of farms reporting different treatment methods for solid manure by the livestock sector was considered, the beef and poultry sectors reported 28% and 13% of the farms practicing composting (Table 13).

Table 12. Percentage of farms utilizing various solid manure treatment practices by province

Province	Mixed with additives to modify odour[1] (%)	Mixed or turned to accelerate composting (%)[1]	All other practices[2] (%)	None (%)
Atlantic	2.5	16.4	2.3	78.7
Quebec	2.6	9.9	1.4	86.0
Ontario	3.1	9.6	1.6	85.7
Manitoba	x	43.9	x	54.5
Saskatchewan	x	29.7	0.9	68.9
Alberta	1.0	35.4	1.2	62.4
BC	3.4	28.3	X	66.0
Canada	1.8	23.7	1.4	73.1

[1]values in this column represent farms that implemented only this practice; [2]all other practices include 'added to an anaerobic digestion system'; [3]data suppressed.

Table 13. Percentage of farms reporting various solid manure treatment practices by Livestock sector[1].

Livestock Sector	Mixed with additives to modify odour[2] (%)	Mixed or turned to accelerate composting[2] (%)	All other practices[3] (%)	None (%)
Dairy	2.3	11.4	1.7	84.7
Beef	1.3	28.0	1.0	69.7
Pork	X	37.8	X	54.0
Poultry	5.7	12.7	4.1	77.6

[1]values may add to more than 100% since farms were able to report more than one practice; [2]values in this column indicate farms that practice only this practice; [3] Other practices include: " added to an anaerobic digestion system" and various other practices specified by the farmer and combination of multiple practices.

6.3 Use of covers for liquid and solid manure storages

The Farm Environment Management Survey reported the use of a roof over the manure storage together with the use of storage covers. The primary purpose of a roof is most likely to keep out precipitation and may have little impact on gaseous emissions unless airflow is impeded (AAFC 2016).

Most of the farms that managed liquid manure did not use any type of cover (74% of farms at the national level, Table 14) while about 9% of farms used a structure with a roof. Of the remaining 17% of farms, 12% used a concrete cover, likely to keep out precipitation and 6% used straw, tarp or 'geomembrane' cover, especially in the dairy and swine sectors (Table 15)

Of the farms that managed solid manure, most farms (88%) did not use any roof or cover over their manure storage(s). Seven percent covered all their storages and five percent covered some of their storages; however, the type of cover was not specified (AAFC 2016).

Table 14: Percentage of farms having various types of roof or cover for their primary liquid manure storage system by Province

Province	No cover (%)	Concrete (%)	Structure with a roof[1]	Starw	All other[2]
Atlantic	81.9	x	x	x	x
Quebec	76.6	6.2	13.4	1.	2.3
Ontario	69.1	20.3	3.7	3.8	3.1
Prairie	76.1	9.7	x	10.4	2.8
BC	61.0	18.9	15.9	x	x
Canada	73.8	11.5	8.7	3.3	2.6

[1]possibly a roof on poles with no walls; [2] Other include 'tarp', 'lid', or geomembrane.

Table 15: Percentage of farms having various type of roof or cover for their primary liquid manure storage system by Province

Province	No cover (%)	Concrete (%)	Structure with a roof[1]	Starw	All other[2]
Dairy	76.8	7.9	10.4	2.8	2.1
Beef	70.5	13.2	X	X	X
Pork	66.8	21.1	4.4	4.7	2.9
Poultry	53.8	26.6	x	x	x

[1]possibly a roof on poles with no walls; [2] Other include 'tarp', 'lid', or geomembrane.

7. Adoption scenarios and possible GHG reduction from stored manure

Based on current level of adoption of different manure treatment methods (as discussed in Section 6 above), we assumed 95% of the farms in the dairy cattle sector and 98% farms in the swine sector do not use any treatment for liquid manure aiming for GHG mitigation at present. For solid manure in the beef cattle sector and poultry sector it was assumed that the only treatment method practiced was 'composting with turning and mixing' which was reported by 10-44% of farms that managed solid manure in different provinces in Canada. Excluding these farms, we assumed 57 to 90 % of the beef cattle farms and poultry farms in different regions do not use any treatment for stored solid manure for GHG mitigation at present. Therefore, GHG mitigation potential for manure management by implementing selected mitigation measures was estimated using assumed adoption scenario for the farms that did not use any treatment for stored manure.

We assumed the manure management infrastructure has a lifespan of approximately 25 years and 4% of producers in Canada would be replacing this infrastructure in any given year (Drever et al. 2021). Assuming an additional 1% of producers that would consider replacing the infrastructure for manure management, if there is incentive to do so, we assumed an adoption rate of 5% of farms that did not practice a specific GHG mitigation measure previously would start practicing that measure starting from 2023. We assumed this adoption scenario for each specific mitigation measure selected individually, for the target livestock sector.

For example, for the dairy cattle sector: taking each mitigation strategy at a time, we assumed 5% of farms that did not practice a specific mitigation measure will adopt that specific mitigation measure starting from 2023. Using the average herd size (number of dairy cattle per farm) in each Province, we next calculated the number of dairy cattle adopting the practice in each year during the 8-year period from 2023 to 2030. We estimated that by 2030, this would bring about 34% of the dairy cattle population adopting the selected mitigation measure. In the next step, using the per head GHG mitigation potential for a specific strategy we then calculated the total amount of GHG that could possibly be reduced for each year from 2023 to 2030.

This analysis was performed using five selected mitigation measures (solid-liquid separation, impermeable synthetic storage covers, complete emptying, anaerobic digestion, and slurry acidification) for liquid manure systems in the dairy and swine sectors and two selected mitigation measures (composting with turning and passive aeration and manure stack covers) for solid manure systems in the beef and poultry sectors.

The estimated GHG mitigation potential by 2030 for the four major livestock groups with selected mitigation measures are presented in Tables 16 to 19. These estimates indicate that for example, slurry acidification could result in a 21-25% reduction in GHG emissions from dairy manure management in different provinces by 2030, relative to 2019 emissions. With other four mitigation measures, the possible mitigation by 2030 ranged from 8% to 17% reduction by 2030 relative to 2019 emissions for different provinces (Table 16). For the swine sector, the possible GHG reduction by 2030 ranged from 7% with solid-liquid separation to 31% with slurry acidification, relative to 2019 emissions (Table 17). For the beef cattle sector, the possible GHG reduction by 2030 ranged from 8 to 11% with composting and 7 -12% with covering manure stack with synthetic cover (Table 18). For the poultry sector, the possible GHG reduction by 2030 ranged from 7 to 11% with composting and 8 -12% with covering manure stack with synthetic cover (Table 19).

As the assumed adoption rate would bring a maximum 34% of livestock population adopting a single specific mitigation measure, farms that did not adopt that specific measure would be able to adopt any other specific measure out of the selected measures within a particular livestock sector. Therefore, a policy program that would encourage a combination of 2 effective measures in the dairy and swine sectors could potentially achieve more than 40% GHG reduction by 2030 relative to 2019 emissions. For example, for the swine sector alone, a combine adoption of slurry acidification and synthetic impermeable covers could potentially achieve about 50% GHG reduction by 2030 relative to 2019 GHG emissions at the national level (Table 17). At the assumed 5% of farms per year adoption rate, a combined adoption of two measure would potentially reach a maximum of 68% of the swine population. A similar combined adoption of the two measures (slurry acidification and synthetic impermeable covers) in the dairy sector could potentially achieve a 37% GHG reduction by 2030 relative to 2019 GHG emissions (Table 16). For the beef and poultry sectors we considered two mitigation measures targeting GHG emissions from solid manure: composting with turning and passive aeration and covering manure stack with impermeable cover. The combined adoption of these two measures potentially could achieve about 22% GHG reduction by 2030 relative to 2019 emissions for these two livestock groups (Table 18 and Table 19).

Table 16. Estimated annual GHG mitigation potential for manure management systems with the implementation of each selected mitigation measure individually in the dairy cattle sector by 2030 in Canada's jurisdictions. (We assumed 5% of dairy farms per year that did not practice the specific measure in the previous year would start practicing the measure starting from 2023)

Province	Baseline Emissions in 2019 (kt CO_2eq $Year^{-1}$)[1]	Solid-Liquid Separation	Storage covers	Complete Emptying	Anaerobic Digestion	Slurry Acidification
		Annual Mitigation in 2030 (kt CO_2eq $Year^{-1}$)[2]				
Alberta	141.0	11.2	20.5	16.4	22.5	30.3
		8%	15%	12%	16%	21%
Atlantic Provinces	84.3	6.7	12.6	10.0	13.6	18.2
		8%	15%	12%	16%	22%
British Columbia	144.7	13.2	24.2	19.1	26.2	35.8
		9%	17%	13%	18%	25%
Manitoba	59.1	4.6	8.3	6.6	9.1	12.3
		8%	14%	11%	15%	21%
Ontario	551.5	39.0	74.3	58.9	80.3	107.2
		7%	13%	11%	15%	19%
Quebec	527.7	46.1	87.2	69.6	94.8	126.2
		9%	17%	13%	18%	24%

Saskatchewan	46.7	3.2	5.9	4.8	6.5	8.7
		7%	13%	10%	14%	19%
National	1554.8	123.9	233.1	185.4	253.0	338.6
		8%	15%	12%	16%	22%

[1]total GHG emissions (CH_4, direct N_2O, and indirect N_2O emissions in CO_2eq emissions) from stored dairy manure in 2019. This includes emissions from dairy cows and dairy heifers.

[2]Percentage reduction due to the specific mitigation measure in 2030 relative to emissions in 2019 is presented immediately below the annual reduction.

Table 17. Estimated annual GHG mitigation potential for manure management systems with the implementation of each selected mitigation measure individually in the Swine sector by 2030 in Canada's jurisdictions. (We assumed 5% of swine farms per year that did not practice the specific measure in the previous year would start practicing the measure starting from 2023)

Province	Baseline Emissions in 2019 (kt CO_2eq $Year^{-1}$)[1]	Solid-Liquid Separation	Storage covers	Complete Emptying	Anaerobic Digestion	Slurry Acidification
		Annual Mitigation in 2030 (kt CO_2eq $Year^{-1}$)[2]				
Alberta	207.8	18.8	42.4	32.8	62.8	64.0
		9%	20%	16%	30%	31%
Atlantic Provinces	9.9	0.6	1.8	1.4	2.4	2.7
		6%	18%	14%	24%	27%
British Columbia	11.0	0.7	2.0	1.5	2.7	3.0
		6%	18%	14%	24%	27%
Manitoba	374.4	25.2	75.6	54.4	99.2	111.4
		7%	20%	15%	27%	30%
Ontario	496.6	38.6	102.0	77.2	141.1	152.3
		8%	21%	16%	28%	31%
Quebec	618.6	47.9	126.5	97.2	175.4	189.5
		8%	20%	16%	28%	31%
Saskatchewan	134.1	9.4	26.1	20.2	35.6	39.1
		7%	19%	15%	27%	29%
National	1852.4	141.1	376.3	284.4	518.7	561.6
		8%	20%	15%	28%	30%

[1]total GHG emissions (CH_4, direct N_2O, and indirect N_2O emissions in CO_2eq emissions) from stored swine manure in 2019.

[2]Percentage reduction due to the specific mitigation measure in 2030 relative to emissions in 2019 is presented immediately below the annual reduction.

Table 18. Estimated annual GHG mitigation potential for manure management systems with the implementation of each selected mitigation measure individually in the Beef cattle sector by 2030 in Canada's jurisdictions. (We assumed 5% of beef farms per year that did not practice the specific measure in the previous year would start practicing the measure starting from 2023)

Province	Baseline Emissions in 2019 (kt CO_2eq $Year^{-1}$)[1]	Composting with turning & passive aeration	Storage covers
		Annual Mitigation in 2030 (kt CO_2eq $Year^{-1}$)[2]	
Alberta	1536.3	139.8	131.6
		9%	9%
Atlantic Provinces	39.5	4.2	4.1
		11%	10%
British Columbia	120.8	11.2	10.5
		9%	9%
Manitoba	230.5	17.4	16.7
		8%	7%
Ontario	413.7	52.1	50.4
		13%	12%
Quebec	213.6	17.4	16.6
		8%	8%
Saskatchewan	815.5	83.6	80.5
		10%	10%
National	3369.8	353.8	337.1
		11%	10%

[1]total GHG emissions (CH_4, direct N_2O, and indirect N_2O emissions in CO_2eq emissions) from stored beef cattle manure in 2019.

[2]Percentage reduction due to the specific mitigation measure in 2030 relative to emissions in 2019 is presented immediately below the annual reduction.

Table 19. Estimated annual GHG mitigation potential for manure management systems with the implementation of each selected mitigation measure individually in the Poultry sector by 2030 in Canada's jurisdictions. (We assumed that 5% of poultry farms per year that did not practice the specific measure in the previous year would start practicing the measure starting from 2023)

Province	Baseline Emissions in 2019 (kt CO_2eq Year^{-1})[1]	Composting with turning & passive aeration	Storage covers
		Annual Mitigation in 2030 (kt CO_2eq Year^{-1})[2]	
Alberta	97.7	7.6 8%	8.9 9%
Atlantic Provinces	60.2	5.8 10%	6.9 11%
British Columbia	125.2	14.2 11%	16.8 13%
Manitoba	71.6	4.9 7%	5.9 8%
Ontario	350.2	39.9 11%	47.6 14%
Quebec	231.9	23.2 10%	27.5 12%
Saskatchewan	33.5	3.7 11%	4.3 13%
National	970.3	99.2 10%	118.0 12%

[1]total GHG emissions (CH_4, direct N_2O, and indirect N_2O emissions in CO_2eq emissions) from stored poultry manure in 2019.

[2]Percentage reduction due to the specific mitigation measure in 2030 relative to emissions in 2019 is presented immediately below the annual reduction.

8. Barriers to adoption

- Capital equipment requirements specific to each mitigation measure can be high. Insufficient added revenue to offset increased costs associated with the implementation of mitigation measures is a major factor preventing the adoption of most of mitigation measures considered here.
- Anaerobic digesters:
  (a) Digester systems require capital investment and adjustment, which will vary depending on the existing collection and treatment systems.

(b) The cost-effectiveness of transitioning from an existing system to a digester system depends on the size of the farm, the ability to sell co-benefits, and the price of electricity.

(c) Regulatory, economic, and technical challenges may act as barriers to transitioning to anaerobic digesters.

(d) Digester systems can be complex, and a variety of factors must be monitored thus requiring continuous attention for proper operation; systems may thus be require additional personnel to maintain.

(e) In some instances, legal barriers may significantly lower adoption rates

- Solid-liquid separation: Technical and economic barriers (due to high capital investment) to the adoption of improved solids separators on farms. High operation and maintenance costs.
- Slurry acidification: Infrastructure costs associated with special storage tank for acid and controlled acid delivery system, administrative constraints as well as health and safety concerns could negatively impact adoption.
- Composting: Infrastructure costs (e.g., compost turners), Knowledge required to properly manage the composting system (e.g. ideal timing for turning), Additional management times required compared to stockpiling manure.

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10. Appendices

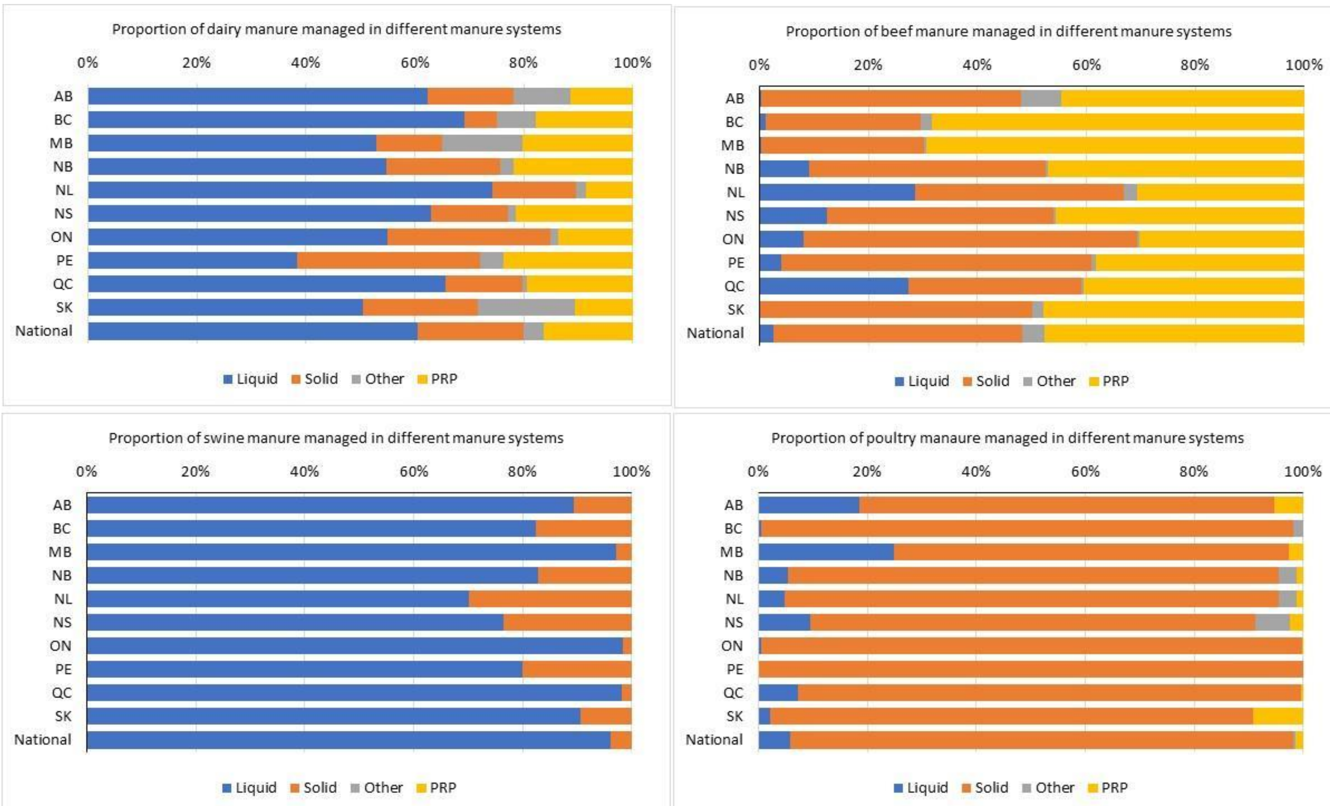

Fig S1. Proportion of manure managed using different manure management systems in the four major livestock groups in Canadian provinces (Data source: National GHG Inventory Report 2021).

Enteric methane emissions associated with livestock production in Canada:

Evidence, adoption, co-benefits, constraints and policy/programming to enhance adoption

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Greenhouse gas emissions from the Canadian cattle industry

The most recent data indicate that enteric fermentation (CH_4) is responsible for 40% of all GHG emissions associated with Canadian agriculture (Table 1; ECCC, 2021). Beef and dairy cattle, with a combined population size of 11.9 million head (Table 2), are responsible for 95% of all enteric CH_4 emissions or approximately 3.29% of total Canadian GHG emissions (ECCC, 2021). The remaining 5% is collectively associated with other livestock species (Table 2). The total enteric CH_4 emitted is 0.917 Mt yr^{-1}, which when converted to CO_2e, using a GWP100 factor of 25 is equivalent to 22.96 Mt annually (Table 2).

Table 1. 2019 Canadian agriculture total greenhouse gas emissions and enteric methane (CH_4) emissions associated with livestock

Category	GHG emissions (Mt CO_2e)
Agriculture total GHG emissions	59.0
Enteric CH_4 emissions	
Dairy cattle	3.5
Beef cattle[a]	19.0
Other[b]	1.1
Total	23.6

Source: ECCC (2021)

[a]Beef cattle includes dairy heifers

[b]Other includes buffalo, goats, horses, lamb/sheep, llamas, alpacas, swine, deer/elk, wild boar

Table 2. 2019 Canadian cattle inventory and enteric methane (CH_4) emissions

Category	Population	Enteric CH_4 emissions	
		Mt CH_4	Mt CO_2e
Beef Cows	3,676,400	0.442	11.06
Beef Heifers	592,700	0.054	1.35
Heifers for Slaughter	807,350	0.022	0.55
Bulls	215,500	0.027	0.67
Steers	1,354,200	0.033	0.83
Calves	3,845,350	0.168	4.21
Dairy Cows	972,900	0.138	3.46

Dairy Heifers	435,600	0.033	0.83
Total	11,900,000	0.917	23.0

Source: ECCC (2022)

More than 70% of the greenhouse gasses (GHGs) associated with beef and milk occur prior to the farm gate (Beauchemin, 2022). It is critical that Canada's livestock producers have the tools to aggressively adopt practices that will enable them to meet emission reductions targets established by industry and government while ensuring economic viability. That said, it is also important to note that the carbon footprint per unit of production for beef and milk to the farm gate in Canada are both less than 50% of global averages and continue to decrease due to advances in nutrition, genetics, health, manure management, feed production and farm management (Beauchemin, 2022). Legesse et al. (2016) reported a 15% reduction in GHG emission intensity in the Canadian beef herd from 14 to 12 kg CO_2e kg^{-1} live weight beef, over a 30-year period. Similarly, Jayasundara and Wagner-Riddle (2014) reported a decrease in emissions intensity in the Canadian dairy herd (CO_2e kg^{-1} fat- and protein-corrected milk) of 22% over a 20-yr time period (1991-2011).

Both the beef and dairy sectors have set ambitious goals to continue to reduce GHG emissions. The Canadian Roundtable for Sustainable Beef (https://crsb.ca/crsb-speaks-to-standing-committee-on-the-environmental-contribution-of-agriculture/) has targeted emission–reductions of a further 33%, as well as sequestering an additional 3.4 million tonnes of carbon per year and maintaining the 34 million acres of grassland in the care of beef producers. Dairy Farmers of Canada has committed to net-zero greenhouse gas emissions by 2050 (https://dairyfarmersofcanada.ca/en/dairy-in-canada/dairy-excellence/dfc-targets-net-zero-greenhouse-gas-emissions-2050) using a combination of emissions reduction and increased carbon sequestration. Given the Dairy Supply Management program limits on overproduction, Canada exports little of its dairy products, focusing on milk prices that are fair to producers and consumers, and enabling farmers to invest in and adopt more sustainable production practices.

It is important to note that continued improvements in production efficiency will help reduce GHG intensities but will not be adequate achieve the deep reductions in absolute GHGs emissions needed. Some promising technologies are on the horizon, but unless improvements in productivity or efficiency offset their cost, their adoption cannot be guaranteed. In the absence of a significant market driver, provincial and federal government support via research, new tools and technologies, and financial incentives will be almost certainly required to achieve the desired emission reduction targets.

Mitigation potential for the cattle sector

A review of the existing literature indicates the potential to reduce enteric CH_4 emissions from the Canadian herd by 5% to 10% with adoption of at least one best management practice (BMP), and the potential for further reduction if multiple mitigation strategies are adopted. The magnitude of impact and survey evidence indicating that some industry leaders are already applying these BMPs, suggests that they are a reasonable target for both producers and policy makers. As well, many of the BMPs have an added benefit of improving the productivity of Canadian cattle production systems. Implementation of best management practices to reduce GHG emissions by improving feed quality and quantity, as well as practices to mitigate risk due to extreme weather events associated with climate change, will inherently improve animal health. These strategies, ensuring an adequate, low-cost feed supply will also improve producer mental health and economic stability of the farm enterprise, as more than 60% of the cost of production in most livestock operations is associated with feed.

Implementation of most BMPs can also have a land sparing effect as either forage productivity or animal demand for feed is reduced. This provides significant opportunity for flexible land set aside programs, a strategy that can support environment sustainability, including biodiversity, and provide a feed inventory to cattle producers in extreme weather events such as prolonged droughts and flooding.

A significant challenge faced by the beef sector regarding BMP adoption is that a large number of farms (50.6%) represent farm operators over 55 years of age - the average age of beef cattle producers in Canada (Canfax Research Services, 2017a). Jelinski et al. (2018) previously reported that the ratio of older (> 60 yr) to younger beef producers (< 31 yr) increased from ~2:1 to ~7:1 between 1991 and 2011. For every younger producer entering the beef industry, there were > 7 producers nearing retirement. This age demographic will impact rate of adoption of BMPs as incentive for change and investment may be reduced for those nearing retirement. That said, the portion of new entrants is increased when comparing the 2011 and 2016 Census of Agriculture data. The proportion of cattle farms with operators under 35 years of age (with single or multiple operators) increased from 7.9% in 2011 to 10.7% in 2016, led by Manitoba (12.2%) and Saskatchewan (12.1%). The share of women operators, as the primary decision maker or present on a multi-operator farm, has been growing since 1996 (32.8%) and increased from 37.4 % in 2011 to 40.2% in 2016 (Canfax Research Services, 2017a).

As indicated above, most BMPs cited in the published literature (Boadi et al., 2004; Hirstov et al., 2013) result in a reduction in enteric CH_4 emissions by 5% to 10%, with few exceptions which go beyond 10% (Alemu et al., 2021a, 2021b). A 5% reduction in annual enteric CH_4 emissions from Canadian dairy and beef operations adopted across all cattle classes would result in a 1.1 Mt reduction in CO_2e annually (0.096 tCO_2e per head), assuming no change in the herd size. More specifically, a 5% reduction would equate to an annual reduction of 0.647, 0.214, and 0.285 MtCO_2e for i) beef cows and heifers, ii) dairy cows and heifers, and iii) bulls, steers, and calves, respectively (Table 3). Adoption of a second BMP, leading to an additional 5% reduction in the enteric CH_4 emissions would lead to a reduction of 2.3 Mt CO_2e yr^{-1} from Canadian cattle operations.

Table 3. Total enteric emissions and expected reduction using BMPs resulting in either a 5% or 10% decrease in enteric CH_4 emissions from the cattle sector using 2019 as the baseline

Category	Total Emissions ($MtCO_2e$)	Reduction from baseline ($MtCO_2e$)	Unit Conversion (Kg CO_2e)
5% Reduction			
Beef Cows	10.50	0.55	552,856,373
Beef Heifers	1.28	0.07	67,429,172
Heifers for Slaughter	0.52	0.03	27,442,271
Bulls	0.63	0.03	33,395,869
Steers (1+ yr)	0.79	0.04	41,511,473
Calves	4.00	0.21	210,282,620
Dairy Cows	3.29	0.17	172,927,297
Dairy Heifers	0.79	0.04	41,732,005
Total	21.80	1.15	1,147,577,081
10% Reduction			
Beef Cows	9.95	1.11	1,105,712,747
Beef Heifers	1.21	0.13	134,858,344
Heifers for Slaughter	0.49	0.05	54,884,543
Bulls	0.60	0.07	66,791,738
Steers	0.75	0.08	83,022,946
Calves	3.79	0	420,565,240
Dairy Cows	3.11	0	345,854,595
Dairy Heifers	0.75	0	83,464,011
Total	20.66	2.30	2,295,154,163

Evidence Associated with Adoption of Best Management Practices

Best management practices are categorized as follows: i) forage and grassland management and ii) precision feeding strategies for confined and pastured beef and dairy cattle.

Format for each best management practice:

Evidence: The direct CH_4 reduction as reported in published literature.

Current adoption rate: The rate of adoption currently present in the Canadian context, sourced from literature values, survey data, or a general estimate based on expert opinion.

Co-benefit: Additional GHG benefits that should be considered, which include carbon sequestration, natural N fixation, etc. These may be overlapping with other areas within the report, in which case, a note to the section is made.

Constraints: Similar to the co-benefits, the consequences, or potential constraints, that should also be considered are included in this section. These include land and/or geographical constraints and increases in other GHG emissions. Overlap with other sections of the report are also noted.

Policy/program direction: General policy suggestions which aim to incentivize and/or aid producers in the adoption of the BMPs are provided. When considering policy, there are several unique attributes to the beef and dairy sectors that should be considered. The number of operations in Canada reporting beef cows decreased by 12.4% from 61,425 to 53,837 between 2011 and 2016 (Jelinski and Waldner, 2018). It is of interest that the number of operations reporting dairy cattle decreased by 13.4% (n = 1988) during this period: 14,883 to 12,895 operations. Furthermore, dairy cattle numbers continued to decline and beef cattle numbers have remained stable during this period. The cow-calf industry is becoming more consolidated in the West, where the average herd is ~3 times larger than in the East. While the attrition rate for producers was 12.4% from 2011 and 2016, the proportion of producers > 60 yr of age reached a historic high. Therefore, the beef industry should expect a double-digit attrition rate for the next 2 census periods, after which time the large cohort of baby boomers will have retired.

Forage and grassland management

A significant portion (19,297,039 ha) of agricultural land in Canada is managed as perennial forage or pastureland with the majority managed by beef and dairy producers. Based on 2019 data (Brian McConkey, personal communication*,* 2022), the national forage and grassland areas reported were 3,254,221 ha of alfalfa and legume grass mix, 4,686,057 ha of improved pasture, 9,627,119 ha of unimproved pasture, and 1,729,642 ha of land designated as other hay and fodder. Unimproved pasture which represents native and naturalized grasslands make up almost 50% of this land area. Forage and grassland management BMPs are relevant to the following cattle categories from April-October: beef cows, beef lactating, stocker cattle, grassers, bulls, calves under 1 yr, replacement beef heifers, replacement dairy heifers, and a limited number of lactating dairy cows. Of the land used by the beef sector, 57.5 % of land used is owned, 20.2% is rented or leased from governments and the remainder is rented, or part of a crop sharing arrangement with other landowners.

1) **Improving nutritional quality of forage in pasture systems through incorporation of legumes into the stand or harvesting at an earlier stage of maturity**

Proposed BMP: Introducing legumes, such as alfalfa, sainfoin, cicer milk vetch, clover and birdsfoot trefoil into grass-only forage stands, at rates between 20 and 30% of the stand, can improve forage quality and reduce animal enteric methane emissions. Further, as forage in beef cattle production systems is often harvested at an advanced stage of maturity to increase available forage biomass, harvesting when in a vegetative state will lead to decreased fibre content, improved animal performance and reduced enteric methane emissions (Vargas et al., 2022).

Evidence:

- McCaughey et al. (1999) reported a 9.5% reduction in daily enteric CH_4 with inclusion of alfalfa (22%) in meadow bromegrass pasture (88%). Including legume in the stand leads to improved digestibility and therefore lower methane emissions. Emissions estimates included herein assume a 20% inclusion rate. Pure stands of legume forage may lead to increased reductions in emissions. For example, McAdam et al. (2022) demonstrated that pure stands of birdsfoot trefoil or cicer milk vetch can lead to emission reductions of 25 and 21.4% respectively, compared to meadow bromegrass. These legumes species, along with sainfoin and birdsfoot trefoil contain phenolic compounds (tannins) which can also reduce enteric CH_4 emissions via their anti-methanogenic activity.

- Several Canadian studies have demonstrated that harvesting forage at an earlier stage of maturity can decrease enteric methane emissions. Boadi et al. (2002) demonstrated a 29% reduction in energy loss as enteric CH_4 when steers grazed on early-season pasture compared to late-season pasture. In addition, a simulated farm using a life cycle assessment conducted by Beauchemin et al. (2011), showed a 7.2% reduction in enteric CH_4 when hay quality was improved by harvesting a grass-legume mixed forage stand at an earlier stage of maturity.

- Direct benefits of adopting best management practices have been garnered directly from literature. In addition to the direct benefit, "secondary emissions reductions" may be realized beyond the initial 5-10% noted above. For example, improving forage quality either by adding a legume or harvesting at an earlier stage of maturity may lead to improved cow body condition score, increased fertility, increased calf performance (number of calves and increased weaning weight), as depicted in Appendix 1. Using the Holos model to evaluate whole system level reductions in enteric emissions (Table 4), a 23% and 16% reduction in enteric CH_4 emissions for grazing cows and grazing calves, respectively is realized through the combined benefit of reduced daily emissions per animal and reduced animal numbers related to improved reproductive and weight gain performance, including an increase in calf crop from 85 to 90%, when offering improved forage quality to grazing animals.

Current adoption rate:

- Legume inclusion on grass-based forage lands will improve the forage quality, increase forage digestibility and intake, and improve animal performance with additional environmental benefits. Canadian farm focus group data (Canfax Research Services, 2021) has indicated that 27% of cow-calf operations have tame grass/legume mixed pasture while Sheppard et al. (2015) reported 81% of cow-calf operations graze tame pastures for some part of the grazing season. These survey results suggest that beef cattle producers have the equipment/means to incorporate legumes into their grazing systems. Data from 2019 (McConkey, 2022) indicates that the percentage of tame pasture and forage land that contains a pure legume or legume grass mixture is approximately 16.9%. Expert opinion (G. Friesen, April 2022) suggests that renewal of improved pasture for beef is between 6-15 years, indicating that legume content would be low. The 2017 Farm Management Survey (Table 33) reported that 50% of Canadian forage/beef producers had not broken up any forage stands in the last five years. This was highest in Quebec (72%) and Saskatchewan (56%). Those who had broken up forage stands in the last five years reported the stands to have been three to five years old (16%), six to ten years old (18%), or more than ten years old (12.6%). Those who rejuvenated stands more than ten years old where higher in Saskatchewan (19%) and Manitoba (17%) which may be a consequence of the higher risk of failed stand establishment due to climate. The age of stands would be younger (4-8yrs) for dairy producers, suggesting a greater likelihood that legume proportion in these stands remains high. Unimproved pastures would contain no or very low levels of native legume species.
- Depending on location, species diversity and at-risk data, it is unlikely that the BMP should be recommended for native pasture, which represents a small percentage of the area, however, there is good opportunity to improve land

productivity and decrease animal enteric CH4 emissions for both naturalized and improved pasture and hay lands. Naturalized pastureland is defined as "Forage species present are primarily introduced from other geographical regions that have become established and have persisted under the existing conditions of environment and management over a long time" (Allen et al., 2011).

Co-benefits:

- Improved production efficiency of cows and their calves related to improved forage quality will lead to further reductions or "secondary benefits" in enteric CH_4 emissions (Appendix 1; Table 4).
- Increased soil carbon sequestration and soil organic carbon (Griscom et al., 2017, see Technical Report of SOC-based Pathways, p. 133).
- Decreased need for nitrogen fertilizer application (Khatiwada et al., 2020) due to biological N fixing and recycling (see Improved Nitrogen Management, p.2).
- Direct seeding into pasture can break up soil compaction which will reduce surface runoff (Canadian Agronomist, 2021).
- Moderate grazing of legume grass mixed stands enhances plant species richness and arthropod communities as well as other ecological services (Pogue et al., 2018).

Constraints:

- Unless proportions of legume and grass are optimal to partially replace N fertilizer, added nitrogen through biological fixing may cause increased N_2O emissions from the land and manure (see Improved Nitrogen Management, p.2).
- Increased production costs due to cost of seed, labour and equipment.

- Producers must employ effective stand management to maintain legumes in the stand and to avoid animal health issues such as bloat when grazing non-bloat tolerant species.

- Inclusion of legume species may be challenging in landscapes characterized by high slope ratios, prairie potholes, brush, stoniness, etc. Therefore, alternative methods of incorporation should be explored and researched. For example, producers have included legume seeds in cattle mineral supplements as a means of dispersing forage seed throughout pastures via the manure.

Policy/program direction:

- Funding for knowledge transfer regarding species selection including new varieties, as well as successful stand establishment and management.
- Specific support for young and new beef cattle producers investing in this BMP, especially with respect to design and establishment of grazing plans/strategies utilizing legume grass pasture stands.

- Provide support which complements existing web-based tools developed by the cattle industry, including forage selection (https://upick.beefresearch.ca/) to advance knowledge and ensure successful implementation.

Table 4. Secondary enteric CH_4 reduction potential, when accounting for performance improvement associated with improved forage quality, using Holos, a whole farm model, for cow-calf pairs

Grazing scenario	Grass only pasture	Grass-legume pasture mixture	Performance Outcomes
Herd description‡			
Calving month	April	April	
Grazing period	May-Oct	May-Oct	
Calf birth weight (kg)	39	40	Increased birth weight
Initial calf weight, when turned into pasture (kg)	72	78	Increased initial weight
Weaning weight (kg)	270	306	Increased weaning weight
Milk production (kg)	7	7	Assume milk production did not change
Number of cows	100	95	Fewer cows required to maintain calf crop
Number of calves	85	85	Assume same output, increased % calves weaned
Calf-crop (%)	85	90	Increased calf-crop
Forage characteristics§			
Botanical composition	Grass	Grass-legume	
Crude protein (%)	9.0	13.1	Increased protein content
Total digestible nutrients (%)	59.8	59.2	
Enteric CH_4 emission factor§	0.095	0.071	Reduced feed gross energy loss in the form of enteric CH_4
Enteric CH_4 emission (kg CO2e head^{-1} grazing^{-1} period)*			
Cows	2529	1942	23% reduction
Calves	277	233	16% reduction

‡Obtained from Beauchemin et al. (2011); Sheppard et al. (2015); Alemu et al. (2017)
§Obtained from McCaughey et al. (1999)
*Estimated using a whole farm model, Holos 3 research version (Government of Canada, 2020)

1) <u>Rotational grazing</u>

Suggested BMP: To replace continuous grazing systems with rotational grazing. There are a wide range of rotational grazing strategies which differ in the number of paddocks and frequency of movement of cattle.

Evidence:

- Heavy continuous grazing on rangeland predominantly containing parry oat grass (*Danthonia parryi Scribn*.) bluebunch fescue (*Festuca idahoensis Elmer*) with a higher stocking density (2.4 and 2.8 animal unit month per hectare, AUM ha^{-1} for cow-calf and backgrounded cattle, respectively) resulted in 3.7% lower enteric CH_4 emission than a light continuous grazing on rough fescue grass (*Festuca carnpestris Rydb*.) and with the respective stocking densities of 1.2 and 1.4 AUM ha^{-1}. The reduced emission is attributed to the superior forage quality with higher digestible energy as well as crude protein (CP), and a less digestible fraction (Alemu et al., 2017). In the same study, enteric CH_4 emissions from the two categories of cattle further declined (9.5%) when rotational grazing was used instead of continuous grazing for the backgrounded animals. There was also a simultaneous improvement in production efficiencies (kg beef per hectare) with the heavy grazing group. The study highlights the importance of ensuring nutrient availability to meet requirement and maximize productivity while reducing enteric CH_4 emissions in grazing cattle.

Current adoption rate:

- Published literature values are limited, however, survey data suggests that continuous grazing continues to be the most common method (Sheppard et al., 2015). The 2006 Farm Environmental Management Survey reported that most farms in every ecoregion practice rotational grazing with adoption rates ranging between 55-80% (FEMS 2006 Grazing Livestock Management). An adoption rate of 55% was estimated by considering pastures grazed less than 2 months as rotational grazing (Canfax Research Services, 2017b).

Co-benefit:

- Rotational grazing improves pasture species composition. When pastures do not have sufficient rest after an initial grazing, the plants most palatable to livestock become less abundant. Hence, rotational grazing should improve forage productivity (biomass yield) and forage quality, reducing the land area required and the need for supplemental feeding in some situations.

Constraint:

- Added labour/time to move cattle, additional fencing requirements (if rotationally grazing sections within an established pasture), producer learning curve if new to this management style.

Policy/program direction:

- Funding for knowledge transfer regarding species selection, as well as successful stand establishment and management
- Specific support for new entrant beef and dairy cattle producers investing in this BMP, especially with respect to design and start-up costs associated with fencing and waterers in a multiple pasture grazing system.
- Funding to explore the benefits of new technologies including virtual fencing.

2) <u>Extended grazing period</u>

Suggested BMP: Maintain beef and dairy cattle on pasture for longer periods of time using a range of extended grazing strategies. Examples of extended grazing strategies include grazing stockpiled perennial forage, swath grazing, bale grazing, grazing annuals in spring or fall, standing corn grazing, and cereal residue grazing. All strategies serve to extend the feeding period from September to end of November or early December.

Evidence:

- Legesse et al. (2011) modeled enteric CH_4 emission (g kg^{-1} dry matter) for cows using IPCC Tier 2 approach for extended grazing on annual crops and perennial pastures, and drylot. Compared to drylot feeding using hay, straw/barley, and silage/straw, enteric CH_4 reductions in extended grazing were 23, 28.5 and 16.8%, respectively. In a similar study, Alemu et al. (2016) reported a 2.2% reduction in enteric CH_4 emission (kg CO_2e cow^{-1} day^{-1}) for corn swath grazing as compared to the traditional drylot feeding overwinter. With bale grazing, however, there is limited evidence which suggests an increase in enteric CH_4 (Donohoe et al.; manuscript in preparation). Enteric CH_4 reductions under extended grazing conditions are associated with diet quality and dry matter intake, and as such significant benefits can be obtained through improvements of forage quality.

Current adoption rate:

- Variability in rate of adoption may be attributed to regional differences in weather including annual precipitation as well as depth of snow and freeze thaw cycles which may limit use of some types of extended grazing systems. However, Canfax Research Services (2021) reported 56%.

Co-benefit:

- The greatest magnitude of response is related to reduced GHG emissions associated with manure, which would be deposited directly onto pasture rather than in a drylot, feedlot or barn. The manure GHG emissions are much lower when directly deposited on to pasture by an animal as opposed to being held in storage and then applied onto land using manure spreading equipment. Using nitrous oxide as an example, emissions are significantly lower (0.00209 kg N_2O-N/kg N) than for stored manure (0.01084 kg N_2O-N/kg

N) which is mechanically spread onto land. A shift from 242 days of confinement feeding for the Canadian cowherd to one in which 56% of the cowherd remains on pasture over winter would result in a decline in manure emissions from 4.6 to 2.5 Mt CO_2e annually (see Improved Nitrogen Management, p.2). Producers are receptive to this BMP because it has also been associated with reduced feed costs, reduced labour and reduced machinery requirements/fuel for delivering feed and manure handling (Sheppard et al., 2015). Other benefits include improved distribution of manure/nutrients on pasture lands.

Constraint:

- Extended grazing may be restricted by provincial legislation in some regions. Animal management and welfare is a consideration, therefore producers must ensure an available water supply and wind protection. There will be times when animals are removed from pasture early due to deep snow, and years where the practice can be extended for additional time. As for all feeding strategies designed to reduce enteric CH_4 emissions, feed testing and appropriate supplementation is necessary to optimize performance and realize emissions reductions.

Policy/program direction:

- Feed testing support
- Ration formulation support for beef producers because they use home-grown feedstuffs and do not traditionally work with feed companies/nutritionists
- Aid in the adoption of incorporating/seeding different forage species – knowledge transfer, machinery limitations
- Support for designing grazing systems
- Water management assistance – access to clean water, water testing
- Labour availability

Precision feeding systems

Precision feeding BMPs are relevant to all classes of beef cattle and dairy cattle.

1) **Ration formulation and precision feeding using: i) standard feed ingredients and ii) by-products associated with crop-livestock integration resulting from increased regional processing of commodities and upcycling food waste into livestock diets.**

Suggested BMP: Employ precision feeding practices which aim to match nutrient supply precisely with the nutrient requirements of individual animals, as well as monitoring intake and performance based on real-time feedback, where the technology exists to do so. This may include the use of a range of by-products including cereal, oilseed and pulse crop screenings as well as novel by-products/surplus food from processors and retailers to correct nutritional deficiencies or optimize nutrient balance. For high-performance animals this also includes strategic supplementation of micro-nutrients which requires precision mixing and feed delivery equipment.

More specifically, precision feeding strategies include:

1. Analysing feedstuffs and balancing diets to ensure that nutrients supplied via a range of feedstuffs meet the physiological demands of the animal for:

a) growth which includes backgrounding and finishing animals, as well as replacement heifers who are growing to achieve targeted gains. Achieving these targets results in improved average daily gain, fewer days on feed to reach target weights therefore less manure and lower GHG emissions

b) improved reproduction rate as feeding balanced rations to cows could lead to fewer open cows and improved calf crop %

2. Use of feed additives which no not directly improve the nutrient profile but improve availability of nutrients (ie enzymes and ionophores) in feedlot diets

3. Use of feed additives which do not directly improve the nutrient profile but reduce GHG emissions (ie 3NOP)

4. Equipment used for feed processing (chopping forage), mixing ingredients including feedstuffs and additives (TMR mixers) and delivery (Growsafe, SmartFeed Pro systems as well as others) which can deliver feed and provide feedback regarding individual animal intake.

Evidence:

Precision feeding is the delivery of a ration that has a nutrient profile that closely matches the need of an individual animal, based on its maintenance and productivity expectations. This reduces the total amount of feed required, the enteric methane emitted and the nutrients lost as manure. Producers also consider the cost of the feed formulation and delivery to the animal. For beef and dairy production, forage quality (discussed above) is a key factor in both precision feeding goals and total feed cost. The following evidence is focused on further options to achieve both goals.

The process of harvesting, storage, processing and retailing of crops grown in Canada generates many by-products and food waste products, some of which are disposed into landfills. It is essential to analyze by-products and waste and formulate into a balanced diet. This includes by-products from the processing of grains (screenings, hulls, etc.) and oilseeds (meals including canola and soybean), as well as commodities which fail to reach the quality grade required for human consumption (due to harvest failures, crop pests, poor growing conditions due to early frosts, floods or drought, or excess production that exceeds storage capacity). In recent years, crop and livestock producers have worked collaboratively to utilize novel feedstuffs including cull vegetables such as potatoes in feedlot diets (Herman Peters, personal communication) and retail waste (LOOP Resources https://loopresource.ca/). For example, substituting potatoes for barley or corn in feedlot diets at an inclusion rate of 15% and 30% on DM basis, led to a decrease of 2.1 and 4.7%, respectively, in emission intensities (kg CO_2e kg^{-1} live weight), as well as a 15 and 30% decrease in land use (Mengistu et al., 2022).

- Proteins: Increased regional processing of commodities, including new facilities to accommodate expanded consumer demands for plant-based proteins and other novel foodstuffs has created a range of by-products available for use as protein supplements in livestock production systems. Protein supplementation is an effective means to increase productivity and consequently decrease CH_4 emissions when forages do not meet rumen microbial and animal protein needs. Bernier et al. (2012) found enteric CH_4 emissions from mature cows reduced from 25.0 to 22.9 L kg $DMI^{-1}d^{-1}$ when diet CP concentration increased from 6.0 to 11.6%.

  Much research has been done on dried distillers' grains plus solubles (DDGS), a by-product of ethanol production, demonstrating both the opportunity for GHG mitigation and the risks when diets are not properly formulated. Canadian research data has demonstrated that inclusion of corn DDGS at 40% in a feedlot ration to partially replace barley grain resulted in a 2% reduction in enteric CH_4 emissions but increased total GHG emission by 9.3% (Hünerberg et al., 2014). The increase was related to excess corn DDGS CP intake, resulting in increased manure N_2O emissions. This underlines the importance of feedstuff analysis and precision ration formulation (precision feeding). McGinn et al. (2009) reported that inclusion of corn DDGS in a barley silage-based feedlot diet at 35% dietary DM reduced enteric CH_4 emissions (g kg^{-1} DM intake) by 16.4% in cattle compared to the control (60% barley silage, 35% barley grain, 2.0% total fat, DM basis), due most likely to the high fat concentration of corn DDGS treatment (5.1% fat, DM basis). Cost and availability are also key considerations that dictate use. Note that feeding CP above the animal's requirements will cause a higher manure N excretion (Beauchemin et al., 2011) and the potential to increase manure N_2O emissions.

- Fats: Dietary lipids are transformed by ruminal microbes and through processes of lipolysis and biohydrogenation (NRC, 2016). Lipolysis refers to the breakdown of fat and other lipids by hydrolysis to release fatty acids. Biohydrogenation of unsaturated fatty acids is identified as one of the mechanisms through which dietary fat supplementation reduces the amount of CH_4 emitted by ruminants, as it reduces the amount of H_2 available to methane-producing microbes (methanogens) in the animal's digestive system. (Johnson and Johnson, 1995). Fatty acids can also exert direct toxic effects on ruminal protozoa and methanogens, reducing both number and activity (Ivan et al., 2004; Maia, 2010). Tallow and sunflower added at 3.4% and sunflower seed at 8.9% to a growing beef cattle diet, reduced enteric CH_4 by 13.5, 13.9, and 32.6% respectively (Beauchemin et al., 2007). Dry matter intake was not affected except for sunflower seed, which could be related to the higher inclusion level. In an earlier study (Boadi et al., 2004), inclusion of fat in high forage feedlot diets reduced enteric CH_4 emission (L kg^{-1} dry matter) by 15%, however this decrease has been accompanied by a 14.3% reduction in dry matter intake and 17% ADG. These studies demonstrated the potential of fats to mitigate enteric CH_4, but optimal levels of inclusion need to be established to reduce enteric CH_4 while maintaining optimal performance. Grainger and Beauchemin (2011) conducted a meta-analysis of 27 studies and concluded that for diets in which fat supplementation was restricted to < 80 g fat kg^{-1} DM, a 10 g kg^{-1} increase in dietary fat decreased CH_4 yield by 1g kg^{-1} DM intake. However, fat concentration should be restricted to no more than 6-8% of dietary DM to prevent any adverse side effects such as reduced feed digestion, DMI, ruminal microbial health and subsequent animal production (Lovett et al., 2003; McGinn et al., 2004; Beauchemin and McGinn, 2006).

Current adoption rate:

- On dairy farms, precision feeding (feed delivery) is affected by the existing infrastructure on farms. Ration formulation is ~83% on dairy farms in western Canada and Ontario (Gee et al, 2021).

- As the average size of the Canadian cow herd is 69 (Statistics Canada, 2016), most cow-calf producers produce home-grown feeds and as such, do not utilize services of feed companies/nutritionists, as do dairy producers.

Co-benefit:

- Other benefits of precision feeding include reduced feed waste, improved nutrient use efficiency and reduced manure nutrient output. Further, if rations were balanced to include low-cost by-products and/or food waste products, the cost of production may be decreased while providing a viable solution to the national food waste challenge.

Constraints:

- Precision feeding for beef and dairy cattle requires investment in equipment to properly mix and deliver a balanced ration to the individual animal. The main constraint is the added costs associated with investment in new technologies and infrastructure. In most operations, particularly those in which livestock are raised in confinement, major infrastructure changes regarding ration formulation and delivery can only be accommodated with a new build or major renovation.
- Local availability of by-products or food waste, price and cost of transportation. Also, with increased interest in recycling there may be competing uses of some by-products over time, resulting in increased cost and/or reduced availability.
- Regulation regarding use of food waste and by-product often lag far behind the science.

Policy/program direction:

- Ration formulation support - beef producers traditionally utilize home-grown feedstuffs and therefore lack access to nutritionist/feed companies/nutritionists, as indicated above.
- A program to support producers to feed test all commodities including by-products and food waste which have increased variability in nutrient profile.
- CFIA support to conduct the feeding trials necessary to provide data necessary for approval regarding of novel feed ingredients (i.e., hemp meal and screenings).
- Aid in estimation of nutrients present in manure based on feed and feeding strategies to improve nutrient use efficiency.
- A program to support producers (with an emphasis on new entrants) in adopting strategies (i.e., cover or subsidize start-up costs)

2) **Productivity enhancing technology (PET's) use**

Suggested BMP: Continued regulatory approval for the use of PETs (ionophores, implants, beta-adrenergic agonists (BAA's)), which contribute to GHG mitigation and have been shown to be safe for use in Canadian feedlots from a food safety and animal welfare perspective. (Aboagye et al., 2021). Investigate the potential to approve use of a novel feed additive (3-Notrooxypropanol, 3_NOP), which has been shown to have significant potential to reduce enteric methane emissions.

Continued Government and public support for the use of PETs will prevent increases in enteric methane emissions due to less efficient feed utilization. (i.e., ensure these products will not be banned with a focus on government support to improve consumer awareness/understanding of the products to address the current narrative set forth by food advertisers). Many countries are currently investigating the potential use of 3-NOP which has the potential to significantly reduce enteric emissions by inactivating enzymes used by

archaea (microorganisms responsible for methanogenesis) (Yu et al., 2021).
The use of 3-NOP as a feed additive in Canada requires regulatory approval by CFIA. It has recently received regulatory approval in Brazil and Chile (Yu et al., 2021), as well as the European Union (https://www.feednavigator.com/Article/2022/02/24/DSM-gets-EU-market-approval-for-its-methane-reducing-feed-additive).

Evidence:

- 4.9-5.1% reduction in kg CO_2e kg^{-1} live weight with use of growth implants (progesterone, estradiol benzoate, trenbolone acetate, and estradiol) in feedlot cattle (Basarab et al., 2012)
- 1 to 4% reduction in kg CO_2e hd^{-1} with use of growth implants in feedlot cattle (melengestrol acetate, ractopamine hydrochloride, trenbolone acetate, estradiol, tylosin tartrate) (Boonstra, 2022)
- 9.6% to 16.4% reduction in g CH_4 kg^{-1} hot carcass weight with use of monensin and tylosin phosphate or with implantation with a combination of trenbolone acetate and estradiol in steers (Stackhouse-Lawson et al., 2013)
- 21.7% to 64% reduction in enteric CH_4 (g kg^{-1} dry matter intake) from 3-NOP use in feedlot cattle (Alemu et al., 2021a, 2021b)

Current adoption rate:

- Approximately ~80% of feedlots use growth implants in 2011 (Sheppard et al., 2015), which may have increased since publication of this survey. Currently, 3-NOP still has not been submitted for consideration in Canada and is pending approval within the USA (McAllister, personal communication)
- PETS are either not approved or more regulated for lactating dairy cows.

Co-benefit:

- Reduced water and land requirements, and reduction in ~~as well as~~ other GHG (i.e., N_2O) and ammonia emissions (Aboagye et al., 2021; Boonstra, 2022).

More specifically, reductions in water (4.8-11.1%) and land (4.8-10.9) requirements, and GHG (3.0-10.1%) and NH_3 (2.9-7.6%) emissions were reported, using growth implants in feedlot cattle (Boonstra, 2022).

Constraints:

- Producer use of PETs is influenced by consumer acceptance, market shifts, and concern of residues in adjacent environments. Although 3-NOP has potential to significantly reduce enteric CH_4 emissions and is available for use in the EU (https://www.dsm.com/corporate/news/news-archive/2022/dsm-receives-eu-approval-Bovaer.html), it does not have regulatory approval in Canada. Expert opinion suggests that it this process may take as long as 5 years (Tim McAllister, personal communication)
- Producer familiarity with the technology and market access may also limit rate of adoption.

Policy/program direction:

- CFIA support to conduct the feeding trials necessary to provide data necessary for approval regarding of novel feed additives (i.e., 3-NOP)
- Public communication and outreach on a Canadian scale to garner public trust regarding the environmental sustainability, food safety and welfare impacts associated with PET's.

Direct vs "secondary" level impacts of enteric methane mitigation strategies in animal agriculture

The impact of a best management practice is often measured with respect to reduced emissions per animal per day or per unit of animal product. This approach does not account for the other impacts that many BMPs may have on a livestock production system. Whenever the BMP improves animal health or performance as well as reducing the enteric methane emissions, there is opportunity to reduce the number of animals maintained or to reduce the number of days animals are on feed to achieve the desired level of production. In each case, the change in animal numbers or days on feed will also lead to a reduction in methane emissions with no change in the amount of product (milk, live calves, meat) generated.

Appendix 1 and Table 4 demonstrate how a BMP such as the inclusion of legumes in a grass pasture or forage stand can reduce daily enteric methane emissions by animals grazing the pasture or consuming harvested forage and improves cow body condition score which impacts reproductive performance (calving rate, herd calving interval, calf body weight) and the weight gain of offspring. The outcome is a reduction in the size of the cow herd to accomplish the same output and fewer days on feed to accomplish the same weight gain. In this example, comparing an unimproved, grass pasture (baseline) with a legume grass pasture demonstrated net reductions in enteric CH_4 emissions to be a 16% and 23% for calves and cows, respectively (Table 4).

Potential stacking effect when producers apply more than one BMP

Cattle producers may adopt two or more BMPs concurrently, i.e., adding a legume and adopting continuous rotational grazing practices. Doing so would elevate the GHG mitigation opportunity and improve production efficiency if either forage quality or biomass availability may be further improved relative to animal requirements for longer periods of time. Forage quality improvements result in increased availability of nutrients, faster passage rates through the animal digestive system, and favour production of propionate. Each of these changes can reduce the daily enteric CH_4 emissions by cattle and/or reduce emissions per unit product. If adding a second BMP extends the number of days animals can consume higher quality feeds, or results in forage that better matches animal requirements, there can be an additive effect. Few examples of the benefits of multiple BMPs exist. Beauchemin et al. (2011) noted that "the extent to which multiple mitigation practices result in additive reductions in GHG emissions from beef cattle is unknown, as experiments to measure these interactions have not been conducted". A more recent study which evaluated the enteric methane emission reductions from rotational grazing and using a higher stocking rate indicated that the combination of the two management practices resulted in a 15% reduction compared to the individual practices (higher stocking rate [12%] and rotational grazing [3 to 4%]; Alemu et al., 2017). Another approach might be to apply one BMP to summer grazing and while cattle are in overwintering environments.

Regional considerations

Greenhouse gas emissions associated with feed production account for a significant portion of emissions from livestock production and for a given crop, have been shown to vary between regions across Canada (Table 5; Desjardins et al., 2020). Therefore, BMPs leading to the lowest emissions may also differ between regions. In the prairies, the high no-till adoption rates combined with reduced summer fallow has led to increased carbon sequestration compared to eastern Canada (Desjardins et al., 2020). Also, increased land-use changes from perennials to annual crop cultivation has contributed to soil carbon loss in eastern Canada and potential increased nitrogen-based emissions associated with increased fertilizer use. Furthermore, large field sizes in the prairies allow efficient use of farm machinery and therefore, fossil fuels. Increased N_2O emissions reported in the east are mainly associated with the wetter climate (Sheppard et al., 2015; Desjardins et al., 2020). Overall, modeling the whole system impacts shows that regional differences should be considered when introducing BMPs to reduce GHG emissions.

Table 5. Carbon footprints (kg CO_2e kg^{-1} DM) of selected crops

Region	Forages	Alfalfa	Barley grain	Corn grain	Potato
Prairies					
Manitoba	0.15	-0.03	0.34	0.64	0.82
Saskatchewan	-0.05	-0.11	0.06	-	0.39
Alberta	0.14	-0.03	0.19	0.40	0.78
Eastern					
Ontario	0.32	0.11	0.59	1.06	1.58
Quebec	0.34	0.11	0.80	1.29	1.92

Source: Desjardins et al. (2020)

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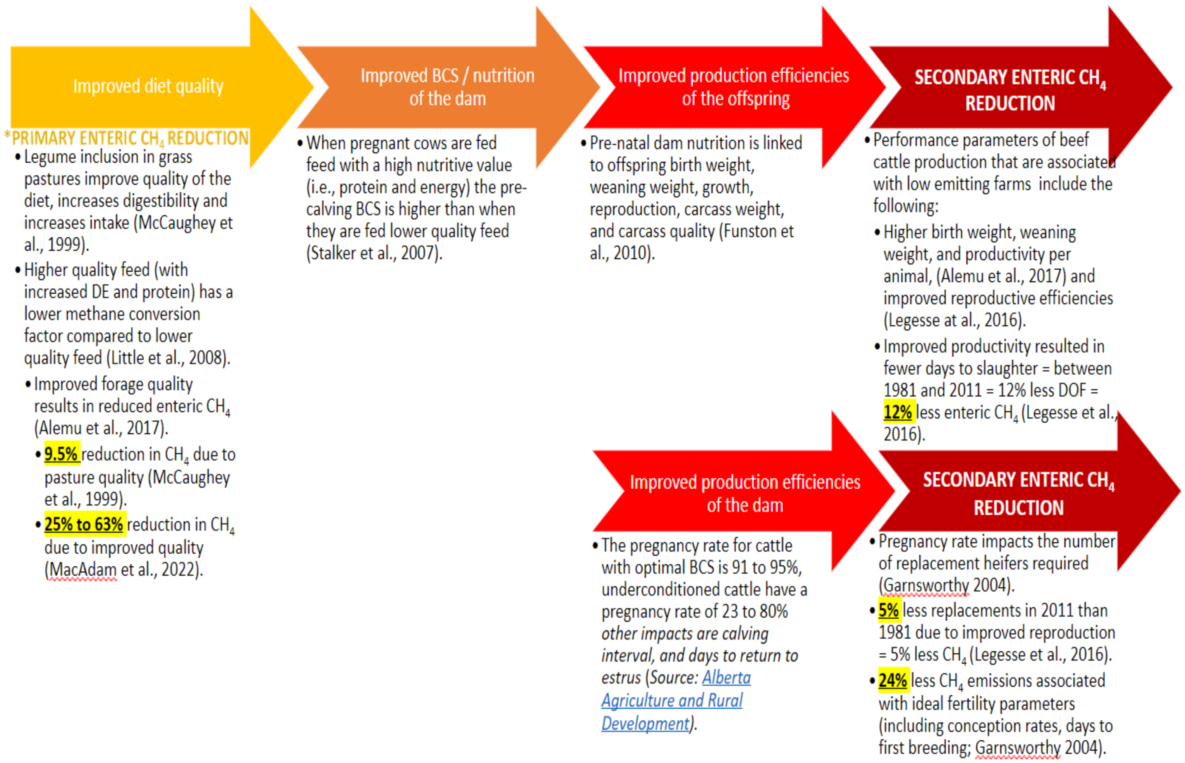
Appendix 1. Secondary enteric CH_4 reduction through improvements in cow productive and reproductive performance, as well as calf performance, could further the reduction attained through pasture quality improvement (9.5%)



Appendix 2. Reduction in enteric methane (CH_4) emissions associated with current and 100% adoption of best management practices (BMPs) by cattle class in 2019

BMP	Cattle class	Population ^	Enteric CH_4 Reduction (%)	Current Adoption rate (%)	Current annual enteric CH_4 emissions (Mt CO_{2e})	Portion of year BMP applied, reflecting current practices (%)	BMP enteric CH_4 emissions (Mt CO_{2e}) adjusted for proportion of year	Future farm Enteric CH_4 emissions reflecting 100% adoption by producers (Mt CO_{2e})	Total reduction of CH_4 enteric emissions (Mt CO_{2e})
1. Improving forage quality by incorporating legumes* i.e.. Alfalfa and bromegrass pasture									
	Cow	3676400	9.5	27	11.06	34	3.73	3.47	0.26
	Heifer	592700	9.5	27	1.35	34	0.45	0.42	0.03
	Bull	215500	9.5	27	0.67	34	0.23	0.21	0.02
2. Rotational grazing**									
	Cow	3676400	9.5	55	11.06	34	3.73	3.57	0.16
	Heifer	592700	9.5	55	1.35	34	0.45	0.44	0.02
	Bull	215500	9.5	55	0.67	34	0.23	0.22	0.01
3. Extended grazing using: Swath grazing on annual crops and perennial pastures***									
	Cow	3676400	22.8	56	11.06	55	6.12	5.51	0.61
	Heifer	592700	22.8	56	1.35	55	0.75	0.67	0.07
	Bull	215500	22.8	56	0.67	55	0.37	0.33	0.04
4. Ration Formulation and precision feeding****									
Protein/Nitrogen	Cow	3676400	2	Pending^^	11.06	66	7.33		
	Heifer	592700	2	Pending^^	1.35	66	0.90		

	Background/feedlot heifers	807350	2	Pending^^	0.55	100	0.55		
	Background/feedlot steers	1354200	2	Pending^^	0.83	100	0.83		
Fat	Background/feedlot heifers	807350	20	Pending^^	0.55	100	0.55		
	Background/feedlot steers	1354200	20	Pending^^	0.83	100	0.83		
5. Productivity enhancing technology*****									
Growth hormones	Background/feedlot heifers	807350	5	80	0.55	100	0.55	0.54	0.01
	Background/feedlot steers	1354200	5	80	0.83	100	0.83	0.82	0.01
3-NOP – Moderate estimates	Background/feedlot heifers	807350	21.7	0	0.55	100	0.55	0.20	0.35
	Background/feedlot steers	1354200	21.7	0	0.83	100	0.83	0.30	0.53
3-NOP – High estimates	Background/feedlot heifers	807350	64	0	0.55	100	0.55	0.43	0.12
	Background/feedlot steers	1354200	64	0	0.83	100	0.83	0.65	0.18

^ 2019 Statistics Canada inventory

^^ Current adoption rate is pending expert opinion as rates for Ration formulation in beef have not been determined

*Enteric CH_4 reduction reference – McCaughey et al. (1999), Adoption rate of legumes – Canfax Research Services (2017)

**Enteric CH_4 reduction reference – Alemu et al. (2017), Adoption rate of rotational grazing – Canfax Research Services (2017) (each pasture grazed less than 2 months)

***Enteric CH_4 reduction reference – Legesse et al. (2011), Adoption rate of extended grazing – Canfax Research Services (2021)

**** Enteric CH_4 reduction reference – Hünerberg et al. (2014) for Protein/N (DDGS) and Beauchemin et al. (2007) for fat

*****Enteric CH_4 reduction reference – Basarab et al. (2012) for GH and Alemu et al. (2021) for 3-NOP, Adoption rate – GH 80% finishing, 54% backgrounding – utilized finishing value (Alemu et al., 2021a; 2021b). Alemu et al. (2021a) reported average reductions of CH_4 yield of 21.7% (Moderate estimates) and Alemu et al. (2021b) reported average reduction of 64% (High estimates).

Technical Report of SOC-based Pathways for Farmers for Climate Solutions Input to

2023-2028 Agricultural Policy Framework

- **Trees in agricultural landscapes**
- **Cover cropping**
- **Intercropping**
- **Rotational grazing (land emissions)**
- **Increasing legumes in pasture and hayland (land emissions)**

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1 June, 2022

Increasing Trees in Agricultural Landscapes

A. Methods for estimating emission

a. Description of emission source and mechanisms

This pathway deals with four tree-based mitigation opportunities based on having more trees that enhance working agricultural landscapes. In all cases, the mitigation is based on change in biomass and soil carbon from the addition or avoided loss of trees. These pathways are

1) Trees as riparian buffers to improve water quality of surface waters by intercepting and reducing the amount soil, nutrients, and pesticides in surface runoff from agricultural fields that enters surface waters.
2) Tree alley cropping or intercropping with cash crops in Ontario and Quebec to improve economic returns.
3) Increasing trees in pastures to provide increased shade and shelter for grazing livestock and for pasture vegetation across Canada.
4) Avoiding shelterbelt removal by renovating existing shelterbelts in the Prairies to preserve sheltering and biodiversity benefits of

The quantification of the mitigation is based on that reported in Drever et al. (2021) with some changes to the adoption scenarios.

b. NIR reporting

The NIR includes estimates of woody biomass changes in cropland (note outside of the Semiarid Prairie and parts of the Montane Cordillera zones, cropland includes all pasture used for domestic livestock). However, the throughness of the estimates is not likely sufficient to capture all changes to woody biomass as described under assumption.

c. Quantities of emission

Between 2008 and 2016, conversion of 2,491.2 km of shelterbelts in Saskatchewan released 0.70 Tg CO_2e (or 0.09 Tg CO_2e/yr) of aboveground carbon into the atmosphere (Amichev et al. 2020).

The rate of new trees added through alley cropping, riparian plantings, and new silvopasture development is not known and assumed neglible.

B. Details of of Proposed Emission-reduction measures

a. Description of measures and emissions

Alley cropping

The analysis of alley cropping or intercropped with trees was confined to Ontario and Quebec because research from the University of Guelph suggests that good productivity and economic returns from intercropping are possible within this area and vegetation class (Toor et al., 2012) Area with Class 3 soils using Land Capability for Agriculture mapping from the Canada Land Inventory (Agriculture and Agri-Food Canada, 2013) and that were cropped or used for hay (Agriculture and Agri-Food Canada, 2020). Class 3 soils were chosen because these soils have moderate to severe limitations on production and are less likely to be used for high value crops. The estimates area was 797,298 ha (45% in Quebec, 55% in Ontario). The intercropping with hybrid poplar (*Populus* spp.) or hardwood species (e.g., red oak (*Quercus rubra*) or Black walnut (*Juglans nigra*)). We modeled establishment of tree intercropping of 1 M tree per year (9009 ha/yr) from 2025 and 2030.

From a review of literature across North America, the establishment of hybrid poplar or hardwood species can capture an additional 0.70 Mg C/ha/yr (mean ± SD) in biomass and 0.17 Mg C/ha/yr in soil (Gordon et al., 2005; Oelbermann et al., 2006; Peichl et al., 2006; Oelbermann and Voroney, 2007; Toor et al., 2012; Winans et al., 2014; Wotherspoon et al., 2014; Winans et al., 2015; Grant et al., 2017). The estimates were based on the commonly analyzed and recommended intercropping configuration from southern Ontario (Peichl et al., 2006; Wotherspoon et al., 2014). This configuration has 15-m tree row spacing and 6-m spacing within a row, or 111 trees/ha. The flux measurements were scaled to this tree density, i.e., 1 ha is a ha of cropland with the intercropped trees.

Silvopasture

Increased CO_2e sequestration from expansion of practices that integrate trees and livestock in the same area to manage simultaneously for tree crops, livestock grazing and forage. We assumed a goal of 1 M trees planted on existing pasture per year allocated across provinces according to their pasture area (9009 ha per year total).

There is particularly limited research on silvopasture systems in Canada. We identified five studies from North America that described biomass and soil carbon accumulation resulting from the introduction of deciduous trees into existing pasture (Gordon and Thevathasan, 2005; Dold et al. 2019; Lim, S.-S. et al. , 2018; Cardinael et al., 2018). Across these studies, silvopasture was estimated to capture an additional 0.63 C/ha/yr in biomass and 0.20 Mg C/ha/yr in soil compared to pasture-only lands. This is for a density of 111 trees per hectare of pasture land with trees.

Riparian trees

Land suited to forest cover was identified as occurring in the 13 forest vegetation classes of Vegetation Zones of Canada map (Baldwin et al. 2018). Within those forestable zones, 6-m wide buffers (3 m on each riparian shore) were mapped around all streams, rivers and lakes whose land cover (Agriculture and Agri-Food Canada, 2020) was currently cropland, grassland, or pasture forage. There was 40,063 ha of potential riparian area. We assumed 1 M trees was planted each year between 2025 and 2030. This would amount to 625 ha of new riparian area per year. Drever et al. (2021) estimates there are 40,063 ha of area that would benefit from adding trees to currently untried riparian areas.

Drever et al. (2021) estimated that the tree growth would be slower for riparian areas based on 30 m wide treed buffers. For narrower buffers considered in this study, we assumed accumulation of C would be similar to silvopasture.

Avoided conversion of shelterbelt

Detailed estimates of mitigation potential from the avoided conversion of shelterbelts are available for Saskatchewan (Amichev et a. 2020; Piwowar et al. 2016). These analyses estimate there are over 51,000 km of shelterbelts within Saskatchewan, of which 2,491.2 km were lost between 2008 and 2016. The Saskatchewan-specific estimates to Alberta and Manitoba by first assuming that shelterbelt density (i.e., km/ha of agricultural area) was proportional by soil type across provinces, and thus used province-specific estimates of agricultural area by soil type in Saskatchewan to estimate shelterbelt extent in Alberta and Manitoba. This resulted in an estimated potential to avoid conversion of 586 km/yr of shelterbelts across the three Prairie provinces (227, 56, and 303 km/yr in Alberta, Manitoba, and Saskatchewan, respectively).

Emissions depended on vegetation type (i.e., coniferous or deciduous trees or shrubs) and the distribution of vegetation type was influenced by soil type. The estimated estimated average avoided loss of carbon was 79.5 Mg C/km of shelterbelt, weighted by soil and vegetation type, from protecting existing shelterbelts.

Protecting existing shelterbelts means 1) retaining exsiting shelterbelts that are still healthy and functioning, 2) renovating shelterbelts by replacing dead and unhealthy trees or 3) replacing shelterbelts where the shelterbelt has reached its end of its useful life.

For this study we assumed that all the protection was through retaining healthy shelterbelts or minimal renovation of a some unhealthy trees or shrubs in existing shelterbelts. Through these protective actions, there would only be minimal loss of C that had accumulated in the replaced unhealthy trees. If the shelterbelt is replaced, then there would be the large loss of accumulated biomass carbon that will only be slowly replaced over many decades.

We assumed with appropripriate action, the rate of shelterbelt loss could be reduced one-half of the 2008-2016 removal rates in the future.

Albedo effects

Tree cover reflects less radiation than crops, hay or pasture. This adds to their to their radiative forcing. Coniferous trees have the lowest refectance, particularly in regions with extended snow cover. Definitely there should be no new conifer plantings in the prairies because over the lifetime of those trees, they will cause more warming from albedo effects than cooling from C sequestration (Mykleby et al. 2017). Therefore the equivalent warming effect from albedo changes (Drever, 2021) for the pathways was included.

Scaling

The riparian, alley cropping, and silvopasture will scale with number of trees. The avoided conversion of shelterbelts is limited by the shelterbelt area.

b. Quantification of Emission

Tables 1-4 give the emission reductions according to the scenarios described above.

Table 1 Mitigation for alley cropping between tree rows

Jurisdiction	Amount* (ha)	---------------------- Mitigation in 2030 ----------------------		
		C (kt/yr)	Albedo (kt CO2e/yr)	Net (kt CO2e/yr)
ON	4955	25.9	-2.05	92.9
QC	4054	21.2	-1.41	76.3
Canada	9009	47.0	-3.46	169.1

*area of alley cropping including cropped area

Table 2 Mitigation for silvopasture.

Jurisdiction	Amount (ha)	---------------------- Mitigation in 2030 ----------------------		
		C (kt/yr)	Albedo (kt CO2e/yr)	Net (kt CO2e/yr)
AB	4024	20.0	-4.79	68.8
BC	245	1.2	-0.07	4.4
MB	443	2.2	-0.51	7.6
NB	28	0.1	-0.01	0.5
NF	0	0.0	0.00	0.0
NS	13	0.1	0.00	0.2
ON	621	3.1	-0.30	11.0
PE	26	0.1	-0.01	0.5
QC	212	1.1	-0.09	3.8

SK	3396	16.9	-4.60	57.5
Canada	9009	44.9	-10.38	154.3

*area of silvopasture including area of herbaceous vegetation

Table 3 Mitigation for new riparian trees.

		---------------------- Mitigation in 2030 ----------------------		
Jurisdiction	Amount (ha)	C (kt/yr)	Albedo (kt CO2e/yr)	Net (kt CO2e/yr)
AB	116	8.61	-1.71	29.9
BC	9	0.68	-0.03	2.4
MB	31	2.31	-0.44	8.0
NB	3	0.21	-0.01	0.7
NF	0	0	0.00	0.0
NS	7	0.53	-0.02	1.9
ON	317	23.5	-1.89	84.3
PE	6	0.41	-0.02	1.5
QC	71	5.24	-0.35	18.9
SK	65	4.81	-1.09	16.6
Canada	625	46.27	-5.57	164.2

Table 4. Mitigation for avoided shelterbelt conversion.

		---------------------- Mitigation in 2030 ----------------------

Jurisdiction	Amount (km)	C (kt/yr)	Albedo (kt CO2e/yr)	Net (kt CO2e/yr)
AB	114	9.7	-0.14	35.5
MB	26	2.2	-0.03	8.1
SK	152	13.0	-0.21	47.4
Canada	291	9.7	-0.14	35.5

c. Assumptions

The tree pathways with new tree plantings scale easily so the area of new trees has the main effect on total mitigation.

The growth rates of trees will inevitably vary with climate and soil conditions and these are not well represented in the analysis. More detail on growth would improve estiamtes especially between provinces.

d. Current adoption

We assumed neglible adoption of alley cropping currently.

Many pastures have abundant trees and there is no data on area of pastures with few trees that would benefit from added trees. Hence the area for new silvopasture is uncertain within and between provinces. Any removal of trees from pasture, particularly from tree encroachment, is not included in the estiamtes.

We assumed there was neglibile retention of shelterblets from replacement or renovation. We also assumed there is negligible new field shelterbelt adoption in prairies. This seems reasonable since the no free trees are being supplied from Agriculture and Agri-Food Canada any more. Farmstead plantings removals or additions were not included but anecdotal evidence is that they are occuring.

e. Barriers to Adoption

Tree rows in cropland are often considered a nuisance with large agricultural machinery. Also, the width of machinery has to match the field with between rows. The benefits of trees to adjacent crop production is often underestimated. Trees can be damaged by herebicides so tree rows may cause restrictions of types and timing of herbicide application. Overcoming these concerns is necessaary to promote field tree rows. Trees in rows or riparain areas reduces the area of cropland.

Trees in tame pasture may need to removed to convert the pasture to cropland if that was desired. Therefore silvopasture could affect the value of the land.

The cost for trees, tree planting, and maintenance requirement until well established are major barriers. Protecting the trees from livestock until they are sufficiently large to coexist with livestock is a barrier to silvopasture.

C. Changes needed to NIR

Claiming mitigation from retaining some trees or adding new trees is only possible if all changes to woody biomass are estimated with equal accuracy. Anecdotely, tree removal from fencelines and riparian areas near wetlands is occuring but is not reported in the NIR. There is also removal of small areas of trees and shrubs that do not make the definition of forest to expand the cropland area. These would also need to be included in the NIR if desire to include the increased or avoided loss of trees in the pathways described in this section.

Therefore, there is new investment needed to improve estimation of woody biomass changes on cropland and agriucltural grasslands to include these tree pathways in the NIR.

Currently the NIR does not consider the radiative forcing (warming) effect of albedo changes. Since trees are long lived land surface change, including albedo effects enables evaluation with that effect that would be expected to be included at some point in national impacts on the climate.

D. Co-benefits

The co-benefits of more trees in the agricultural landscape are numerous, significant, and well documented.

These include: increased diversity and abundance of native bees (Paterson et al. 2019), other beneficial insects (Thevathasan and Gordon, 2004); microbial community (Mafa-Attove et al. 2020; Bainard et al. 2011) and benthic insect and fish diversity (Oelbermann et al. 2008); sustained biodiversity of plants, reptiles, birds, and small mammals (Gibbs et al. 2016; Jobin et al. 2004) , increased stable soil organic matter pool in agricultural lands (Baah-Acheamfour et al. 2015); increased SOC, total N, and C:N ratio (Thevathasan and Gordon, 2004, Rivest et al. 2019; Coleman et al. 2018; Ashiq et al. 2018; Nasielski et al. 2015; Isaac et al. 2014; Bambrick et al. 2010; Oelbermann et al. 2006); reduced soil erosion (Kulshereshtha and Kort, 2009), improved microclimate conditions (Peng et al. 2015) and water use efficiency (Link et al. 2015), reduced leaching of dissolved organic N (Bergeron et al. 2011) and NO3-N (Dougherty et al. 2009; Martin et al. 1999); reduced E. Coli leaching and nutrient run off from agricultural fields (Oelbermann and Gordon, 2000)

E. Limitations

The availablity of suitable tree seedlings and planting equipment could limit the new tree pathways.

The acceptability of alley cropping is not well established.

The best configurations of trees for performance and establishment for silvopature is not well known. Planting trees on natural pastures will need to be done carefully with naturally occuring species so as not to disrupt biodiversity.

The loss of accumulated C from shelterbelts that have reached the end of their useful life should be included to provide a full picture of how shelterbelt management affects mitigation. Unfortunately, we do not have the detailed data on the health of shelterbelts to make that assessment.

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Cover Crops

A. Methods for Estimating Emissions

a. Description of emission source and mechanisms

Introduction

A cover crop was defined as crop grown in addition to normal production of cash crops that are harvested for grain. Cover crops build soil organic matter, improve soil structure, increase soil microbial diversity, protect the soil from erosion, reduce nitrogen leaching, and reduce the need for nitrogen fertilizer via biological nitrogen fixation by legumes in the cover. Other benefits include reducing pests and diseases of cash crops and reduce weed problems in cash crops.

Cover crops are either interseeded (planted within) a cash crop or seeded after cash crop harvest. The cover crop growth continues after cash crop harvest for the fall, or, for a winter cover crop, continues to grow the next spring before the next cash crop is grown. Particularly for later-harvested crops, rather than trying to seed post-harvest for emergence that fall, the cover crop can be seeded later into frozen ground so that it germinates and grows in following spring before the next cash crop.

Where the normal production practice is to fallow the land by not growing a crop in the normal growing season, a cover crop grown on that fallow but is not harvested is a good option to provide many soil benefits. Some growers, particularly organic growers, may grow an unharvested crop, that may be termed a cover crop, for soil improvement instead of a cash crop. These crops are also called green manure crops. For a green-manure cover crop grown on planned fallow in semiarid areas, the carbon sequestration benefits are about the same as growing a cash crop instead of fallow (Campbell et al. 2007). Therefore, there is no SOC benefit to green manure crops rather than a cash crop in semiarid region. The GHG effects for this practice of growing a cover crop instead of a cash crop are not sufficiently studied in more productive climates to estimate the C sequestration. The economics of growing a cover crop instead of a non-organic cash crop in productive environments also needs to be considered. Nevertheless, if there is a situation where fallow was planned, then planting a cover crop on that land is preferred to bare fallow for climate mitigation, soil health, water quality, and biodiversity reasons.

Forages established within or immediately after a cash crop are not considered cover crops in this study when the forage grows for one or more subsequent growing seasons. The practice of interseeding forages with a cash crop, often called companion cropping, is already considered a normal practice for forage establishment. Forage crops provide many soil and environmental benefits, but these are due to the forage production over years, not to the interseeding during the establishment year. An intercrop, when two or more crop types are grown together, but all harvested for grain, was not considered to be a cover crop. Winter cereals grown for grain harvest provide some of the benefits of the cover crop in terms of reducing nitrate leaching and protecting the soil from erosion in the fall, winter, and early spring, but are not addition to normal production so are not considered cover crops.

There are many species options for cover crops including grasses (winter cereals such as wheat and rye, spring cereals such as oat or barley, forage grasses such a ryegrasses), legumes (alfalfa, vetch, clover, pea, soybean) and non-legume broadleaves (radish, buckwheat, marigold). An increasing practice is to have a mix of species and types to both better capture the various benefits provided by each and to have at least some species that will be suitable for whatever weather is received.

The benefits of cover crops were assumed to be closely related to the biomass produced by cover crops. The rational is that the biomass affects total uptake of soil nitration, the amount of root growth to affect the soil structure and soil microbial community, the amount of C input to the soil, the amount of growth promoting substances or disease/pest suppression provided by the cover crop. The cover crop biomass varies with expected cash crop harvest date and/or suitability of interseeding of the previous cash crop, meaning potential benefits and feasibility vary with the climate zone and the type of previous cash crop.

b. Current coverage in the NIR

The impact of cover crops, either on SOC or N2O emissions, is not currently covered in the NIR.

c. Quantities of Emissions

Cover crops increase C input to the soil that increases SOC from what would have been. Cover crops with legumes provide N that reduces need for N fertilizer. This reduces N2O emission and the emissions to produce substituted N fertilizer. Finally, cover crops reduce indirect N2O emission by reducing N leaching and runoff.

B. Details of Proposed Emission-Reduction measures including mechanisms of reduction action

a. Description of proposed reduction measures

Soil Carbon

Cover crops mitigate GHG emissions through increased C sequestration (Abdalla *et al.*, 2019; Bai *et al.*, 2019). The measured C sequestration rates for the Mixed Wood Plains zone are 0.24 Mg C ha^{-1} yr^{-1}(Agomoh *et al.*, 2020), 0.67 Mg C ha^{-1} yr^{-1} (Yang and Kay, 2001). However, Jarecki *et al.* (2018) and N'Dayegamiye and Tran (2001) found no effect on SOC in the Mixed Wood Plain while Poeplau *et al.* (2015) found a rate 0.27 Mg C ha^{-1} yr^{-1} in a similar climate of S. Sweden. There are fewer 0.49 Mg C ha^{-1} yr^{-1} for climate similar to Pacific maritime (Poeplau et al. 2015) and 0.2 Mg C ha^{-1} (Campbell *et al.*, 2007) to 0.32 Mg ha^{-1} (Biederbeck *et al.*, 1998) for cover crop as fallow replacement in the Brown soil zone.

The strong relationship between C inputs and SOC stocks is well established (Liang *et al.*, 1998; Frank *et al.*, 2012; Maillard *et al.*, 2018; Smith *et al.*, 2018) and has been used to estimate effect of cover crops on SOC (Koga *et al.*, 2011; Poeplau and Don, 2015). The global mean C input from the 1.87 Mg C ha^{-1} yr^{-1} from the cover crop (Poeplau and Don, 2015). Comparable values of C input have been observed for mixed wood plains ecozone for the favourable situation of cover crops seeded with wheat, assuming 0.28 for annual grains and 0.44 for legume cover crops (Hu *et al.*, 2018): average of 1.4 Mg C ha^{-1} yr^{-1} (Garand *et al.*, 2001), 1.8 (Wagner-Riddle *et al.*, 1994), and 2.8 Mg C ha^{-1} yr^{-1} (N'Dayegamiye *et al.*, 2015). Therefore, we the 1.87 Mg C ha^{-1} yr^{-1} for cover crop for the favourable mixed wood plains climate zone. We assumed this rate would also apply to the Atlantic maritime climate zone. Although there is a trend for legume cover crop to produce lower SOC increases than non-legume cover crops (Poeplau and Don, 2015; Abdalla *et al.*, 2019), the difference is not significant, so we assumed that the C sequestration was not different between legumes and non-legumes of well-adapted cover crop cultivars.

Both the cash crop and the climate zone affect the perspective C input. The earlier the previous cash crop is harvested the greater the expected growth: estimated input of 1.27 Mg C ha^{-1} yr^{-1} for clover interseeded into spring wheat versus 0.31 Mg C ha^{-1} yr^{-1} when interseeded into later maturing corn (N'Dayegamiye *et al.*, 2015). For soil zones, the C input from cover crops in Black soil zone, cover biomass yields about 0.5 to 0.6 Mg C ha^{-1} yr^{-1} (Martens *et al.*, 2001; Thiessen-Martens *et al.*, 2015). In the semiarid prairie in North Dakota, Hendrickson et al. (2021) found cover crop C yields for range of cover crop types and mixtures seeded in late August ranged from near 0 in a dry year to 0.6 Mg ha^{-1} yr^{-1}. Also in the semiarid prairie, but in southern Alberta, C input from cover crops ranging from near 0 in dry years to 0.3 Mg ha^{-1} yr^{-1} in moister years (Blackshaw *et al.*, 2010). At Saskatoon, Saskatchewan, Farzadfar et al. (2021) had rye cover crop yields seeded in late August to mid September that produced 0.2 to 1.3 Mg C ha^{-1} yr^{-1}.

The C input for other zones and previous cash crops were estimated by interpolation and extrapolation from these values by expert opinion based on limitation of the climate and the characteristics of previous cover crops. The assumed rates are in for the subset of the field situations within climate-previous cover crop combinations that are best suited for cover crop adoption. The zones used the Reporting Zones in the NIR. The Mixed Wood Plains reporting zone most favourable climate for cover crop in Canada is the Pacific Maritime followed by the Mixed Wood Plains and Atlantic Maritime, which was assumed to have one-half the growth potential for fall or winter cover crops (Table 1). The Subhumid Prairies zone is most favourable climate for the Prairies but both cold and lack of water were assumed to restrict the growth of cover crops by about one-half compared to the Mixed Wood Plains. Within the Prairies, cover crop potential becomes increasingly restricted by cold moving from Subhumid Prairies to the Boreal Plains and by lack of water moving from the Subhumid to the Semiarid Prairies. The Montane Cordilleran and Boreal Shield zones were considered similar overall to the Subhumid Prairies albeit ignoring the significant variation in climate in those zones, particularly the Montane Cordilleran, that would affect the ratings on a local area basis.

Table 1 C input by reporting zone and crop.

Zone	Winter wheat, fall rye[1], winter canola	Pea	Barley, oat, mustard	Fallow replacement	potato, sugar beet, chickpea	Grain corn, sunflower	bean, flax, lentil	Spring canola, spring rye[1], silage corn, canaryseed, spring wheat[2], soybean
	-- Previous Cash Crop---							
	-- C input (Mg C ha^{-1} yr^{-1})--							
Mixed Wood Plains, Atlantic Maritime, and Pacific Maritime Zones	1.87	1.12	1.31	5.61	0.56	0.75	0.75	0.94
Boreal Shield East	0.94	0.56	0.65	2.81	0.28	0.37	0.37	0.47
Boreal Shield West	0.56	0.34	0.39	1.68	0.17	0.22	0.22	0.28
Semiarid Prairies	0.37	0.22	0.26	1.12	0.11	0.15	0.15	0.19
Subhumid Prairies & Montane Cordillera	0.75	0.45	0.52	2.24	0.22	0.30	0.30	0.37
Boreal Plains	0.56	0.34	0.39	1.68	0.17	0.22	0.22	0.28

[1] includes triticale, [2] includes durum

Nitrous oxide emission

Based on meta-analyses (Basche *et al.*, 2014; Poeplau and Don, 2015; Han *et al.*, 2017; Abdalla *et al.*, 2019; Muhammad *et al.*, 2019) we estimated that non-legume crop reduced annual direct N_2O emissions in cold climates (Muhammad *et al.*, 2019) while they would be increased by legume cover crops; the latter is consistent with the only comparison we found for Canada (Quesnel *et al.*, 2019). The effect was assumed to be 10% increase or decrease dependent on the fraction of legume biomass in the cover crop:

$$FdN_2O = 0.9 + Pleg*0.2 \qquad \text{(eq. CC.1)}$$

where FdN_2O, is non-dimensional factor for cover crop effects on direct N2O emissions estimated by the methods of Rochette *et al.* (2008) as implemented by ECCC (2019) and Pleg is the fraction of legumes for whole cover crops area in a SLC polygon. The effect of cover crops on direct N_2O emission are provided in Table 2.

Compared to no cover crops, cover crops significantly reduce nitrate leaching (Thapa *et al.*, 2018) with reducting increasing linearly to estimated biomass C input of about 1.87 Mg C ha^{-1} yr^{-1}. Abdalla *et al.* (2019) found leaching reduced with non-legume cover crops was about 50% and that with legume cover crops about 30%. Using the latter rates, and scaling the reduction by the estimated cover crop biomass (Table 1), the effect was estimated as:

Fleach = 1-min [1, C input (Mg ha^{-1} yr^{-1})/1.87 (Mg ha^{-1} yr^{-1})] * (0.5-Pleg * 0.2) (eq. CC.2)

where Fleach is the dimensionless leaching reduction factor applied to estimated N leaching and subsequent indirect N2O emission as calculated using method of Rochette et al. (2008) as implemented by ECCC (2019) and Pleg is the fraction of legume biomass in the cover crop. Table 3 provides the effect of cover crops on avoided N leaching. The avoided leached N was assumed to be cycled through the cover crop and then available to the next cash crop.

N provided by legumes

The estimated N credit from productive legume cover crops in Ontario is 73 kg N ha^{-1} for the following crop (OMAFRA, 2017). The credit was assumed to decrease linearly with the cover crop legume. Thus, the N credit for cover crops, CCN (kg ha^{-1}) was:

CCN = Cinput (Mg ha^{-1})/1.87 (Mg ha^{-1}) * Pleg * 73 (eq. CC.3)

Fertilizer N replaced with N credit was assumed to have embodied C footprint of 3.180 kg CO2e/kg N (Cheminfo Services Inc., 2016).

Effect of grazing and harvest of cover crops

Grazing cover crops can provide important economic benefit for grazing cover crops (Thiessen-Martens and Entz, 2011). There is little information on how grazing affects the GHG balance. Assuming that grazing decreases total growth by 20%, grazing removes 70% of above ground growth with 80% digestibility, and a root:shoot ratio for cover crop of 0.2 in upper 30 cm of soil (Hu *et al.*, 2018), the grazing would reduce by about 50% the C returned to the soil. The effect of grazing on N leaching of cover crop is more complicated but, to be conservative, we assumed that 50% of N that would have been prevented from leaching by the cover crop did not occur due to reduced growth from grazing and return of readably leachable N in grazing livestock urine. Based on two recent studies (Abagandura *et al.*, 2019; Singh *et al.*, 2020), we assumed that grazing of cover crops had no effect on direct N2O emissions from the soil. The indirect GHG effects of the new feed from cover crop is complex. Assuming the livestock numbers are not affected, that displaced feeds had similar diet quality, and the displacing of feed by cover crops does not increase GHG emissions elsewhere, the simplest assumption there is no additional GWP effect from grazing cover crops beyond the direct effects on C sequestration and N leaching.

If the cover crop, whether a conventional cover crop that is additional to cash crop or a green-manure cover crop is harvested for forage, at this point there is insufficient data to estimate how it affects SOC change and N_2O emissions.

For simplicity, we assumed no grazing of cover crops, but, based on the previous discussion, if the cover crops are grazed, the GHG impacts need to be considered.

Other emissions

There are additional emissions for cover crops. We estimated 14 kg CO_2e ha^{-1} as the fossil fuel emissions from shallow soil disturbance for the seeding (Dyer and Desjardins, 2003). We also used 91 CO_2e ha^{-1} for the embodied emissions in the cover crop seed (Dewayne, 2013). Cover crop may require a separate operation for termination and possible mechanical treatment such as crimping. We assumed that the energy for these operations were equivalent to conventional seedbed preparation.

b. Quantification of emissions

Potential Adoption

In Canada generally, limiting fall conditions are insufficient warmth and/or suboptimal soil moisture for successful growth and time conflict with cash crop harvest. For these reasons, a previous cash crop that mature early and/or are suitable for interseeded cover crops (e.g., winter cereals) have much higher technically feasible adoption that later maturing cover crops and/or cash crops that require post-harvest cover crop seeding (e.g., potato) as the latter is more likely to conflict with harvest of other cash crops. Favourable spring conditions for cover crops depend on the type and/or proportion of subsequent cash crops that will be seeded relatively late in the normal spring seeding window to allow time for appreciable spring growth of winter cover crops and, in drier climates, opportunity for spring precipitation to replenish surface soil moisture after spring termination of winter or early spring cover crops. We assumed that the technical challenges of interseeding into many crops were being overcome. Therefore, crops for which an interseeded cover crop does not deleteriously affect cash crop harvest (i.e., little cover crop biomass is harvested with the cash crop), or the cash-crop harvest is not deleterious to an interseeded cover crop (e.g., potato) would be more feasible for cover crops.

We assumed farmers would adopt cover crops only for the subset of their fields that have the most favorable conditions in any specific year for successful cover crop production (e.g., timing of cover crop seeding opportunity, soil tractability for seeding, planned subsequent cash crop expected to perform well after the prior cover crop). Those fields with unfavourable conditions (too dry, too wet, harvest too late) were assumed to not have practical potential for post-harvest seeding. Therefore, there was a maximum potential adoption assigned be climate zone (zone is the LULUCF reporting zone in ECCC (2019) and previous cash crop (Table 2.) These potential adoption rates need to be viewed as long-term, regional rates and would not apply on local, annual basis. If conditions are especially unfavourable for an area in one year, there may be no potential adoption in that year and area, whereas, in another year that same area may have especially favourable conditions an adoption could exceed the long-term potential. Where the climate is favourable for cover crops, such as the mixed wood plains of southern Ontario and Southern Quebec, the potential subset of fields for cover crops would be the majority of fields whereas, where the climate is more unfavourable for cover crops (such as the Semiarid Prairies of the Saskatchewan and Alberta, that subset of potential fields would be less than half of all fields.

Table 2. Estimated Maximum Feasible Adoption Rate by zone and previous cash crop.

Zone	Winter wheat, fall rye[1], winter canola	Pea	Barley, oat, mustard	Fallow replacement	potato, sugar beet, chickpea	Grain corn, sunflower	bean, flax, lentil	Spring canola, spring rye[1], silage corn, canaryseed, spring wheat[2], soybean
	---------- Previous Cash Crop ----------							
	---------- Maximum Potential Adoption (% of crop area) ----------							
Atlantic Maritime	80	48	56	100	24	32	32	40
Mixed Wood Plains	100	60	70	100	30	40	40	50
Boreal Shield East	60	36	42	100	18	24	24	30
Boreal Shield West	50	30	35	100	15	20	20	25
Semiarid Prairies	40	24	28	100	12	16	16	20
Subhumid Prairies & Montane Cordillera	60	36	42	100	18	24	24	30
Boreal Plains	40	24	28	100	12	16	16	20
Pacific Maritime	100	60	70	100	30	40	40	50

[1] includes triticale, [2] includes durum

Allocating cover crops among cash crops

We assume that the potential maximum adoption decreases as the potential biomass production of the cover crop decreases. This lower biomass production will also reduce the various greenhouse gas and agronomic benefits of cover crops would also decrease. Thus, cash crop-zone combinations with lower potential adoption will have lower mitigation benefits than those with higher potential adoption. Therefore, there is expected to be a relationship between mitigation and the cash crop and zone combinations. The allocation of adoption among zones and cash crops is important.

The actual adoption is expected to decrease as the potential adoption increases since the agronomic and soil benefits of cover crops will also reduce with expected cover crop biomass production. This expected biomass effect was included by calculating creating an indicator of expected propensity to adopt that was the product of maximum adoption (Table 2) and relative expected C input (Table 1). Adoption was preferentially allotted to the higher propensities while still having limited adoption for low propensities to reflect grower diversity.

Rates of SOC change

The SOC change was estimated by using the IPCC Tier 2 steady state model (IPCC, 2019), applied at the ecodistrict level, a method that was first implemented in Canadian NIR for 2022. The model is based on the well-accepted Century model and estimates SOC change based on average C input to the soil and annual weather. The simulation started in 1971, after SOC initialization to match the SOC in Canada's National Soil Database (Agriculture and Agri-Food Canada, 2021), based on actual crop yield and weather data to 2019. This modelling approach should represent average Canadian soil conditions as affected by previous crop production practices. The 2020 to 2030 was set to the average weather factors and crop area for 2015-2019.

The cash crop yields for the 2020 to 2030 period were increased based on linear continuation of 2005-2018 trend on an ecodistrict basis. There was no change to cover crop yields over time. The SOC change rate in 2030 calculated from the difference between 2029 to 2030 SOC change between baseline and program scenario. Note that the modelled SOC change in 2030 will depend on both the crop yields and weather that occur preceding and during 2030. Therefore, when real data is available during that period in the future, the modelled SOC change will be different from the modelled estimates for this study.

Table 3 summarizes the mitigation opportunity for increasing the cover crop area for a baseline of estimated 2020 adoption to 40% in ON and QC, 20% in Atlantic Canada, and 5% in western Canada of total cropland using a 50% legume cover crop. In central and Atlantic Canada, where baseline cover crop adoption was significant, the most favoured cover crop opportunities were already assumed to be largely exploited in the baseline. Emission reductions come from a combination of SOC increase, reductions in N2O emissions, reduction in fossil fuel use for N fertilizer manufacture.

The rates of SOC increase were generally consistent to that those that have been reported in the literature where available. An important advantage of SOC modelling approach was making estimates for C sequestration for most of Canada where there are no measurements of SOC change in the literature.

The effect on N2O emission increased as climate became wetter due to the increasing benefit of cover crops for reducing indirect N2O emissions by reducing nitrate leaching. Therefore, in CO2e terms, the mitigation benefit of N2O emission reduction relative to that from SOC sequestration ranged from 30% in Saskatchewan, to 50% in Ontario, to 117% in Nova Scotia. In the prairies, the SOC change was the largest component of mitigation. Outside of the prairies, the positive effects of cover crops on N budget were a significant contributor to mitigation.

Fully 44% of the mitigation potential for scenario of cover crop adoption in Canada is in the provinces of Ontario. On a per ha basis, the highest mitigation opportunities for cover crops were in Atlantic Canada.

C. Changes needed to include cover crops in the Canadian National Inventory Report.

Since cover crops are now present and affect GHG emissions and removals, Canada is obligated under the United Nations Framework Convention on Climate Change to estimate their effect on GHG. This would also be needed to have an inventory that meets Canada's obligations under the Paris Agreement.

First, there is a need to have data on the area, types, at least regarding legume proportion of total biomass, and production of cover crops. That activity data also needs to be estimated for the past as well. A time series of cover crop data is needed for modelling SOC change in response to C inputs.

To have confidence in the estimates, better refinement of the effect of cover crops on N2O emissions, including leaching, is needed that is based on Canada-relevant investigation.

Table 3. Area and emission reduction from baseline for a scenario of increased adoption (5% of cropland area in western Canada, 40% in Ontario and Quebec, and 20% in Atlantic Canada by 2030).

Jurisdiction	2030 baseline cover crop adoption (kha)	2030 scenario cover crop adoption (kha)	-------------------- per ha of additional cover crop -------------------				Total Emission Reductions from the Baseline (Mt CO2e/yr)
			SOC increase (t C/ha/yr)	Decrease in direct and indirect N2O-N emissions (kg/ha/yr)	Added N credit (kg N/ha/yr)	Additional emission for cover crop seed and machinery operations (t CO2e/ha/yr)	
AB	33.2	437.1	0.11	0.28	8	0.105	0.178
BC	0.7	9.3	0.24	1.81	23	0.105	0.014
MB	15.5	203.8	0.08	0.33	8	0.105	0.067
NB	3.4	10.9	0.16	1.51	21	0.105	0.009
NF*	0.0	0.0	0	0.00	0	0	0.000
NS	1.4	4.3	0.16	1.89	25	0.105	0.004
ON	541.0	1144.9	0.20	0.77	23	0.105	0.635
PE	6.6	20.9	0.11	1.18	21	0.105	0.013
QC	213.2	451.2	0.15	1.06	19	0.105	0.242
SK	58.8	773.9	0.09	0.23	8	0.105	0.267
Canada	873.8	3056.4					1.431

*Negligible predicted adoption in NF

c. Assumptions and sensitivities

The benefits of cover crops outside of the Prairies depends on our knowledge of their effects on both direct and indirect N2O emissions. As there are little data on that effect, the mitigation is sensitive to changes to the effect.

The mitigation is sensitive to estimates of C input from cover crops.

The total mitigation is sensitive to the baseline adoption. These affects the amount of mitigation that will occur for a set level of adoption in scenarios of increased future adoption. Importantly, the baseline also affects the mitigation for each additional unit area of new adoption owing to the assumption that the situations with more potential to produce cover crop biomass and thereby mitigation are exploited first.

d. Current adoption

Based on the 2017 Farm Management Survey (D. Cerkowniak, AAFC, personal communication), we estimated there are currently 630,000 ha of cover crops, ranging from 13.5% of cropland in the Mixed Wood Plains to 0.4% in the Black soil zone of the Prairie (data not shown). We assumed that adoption in the Atlantic Maritime and Pacific Maritime zones was one-third of those in the mixed wood plains. We assumed the 0.4% adoption for all other zones.

The 2020 Ontario survey showed that 40% of current producers had only been using cover crops for 5 years or less and almost no producers surveyed had have stopped using cover crops (Morrison and Lawley, 2021). Therefore, we increased adoption in the mixed wood plains by 40% to 19.2% for 2020.

e. Barriers to adoption

The cost for cover crop seed and its sowing is a major barrier to adoption. Other important practical barriers are labour and equipment constraints for seeding cover crops.

The uncertainty regarding the timing and amount of private benefits from cover crops is also a barrier. Farmers generally report that benefits for improved soil resilience to various stresses, reduced loss of soil nutrients with soil erosion and leaching, extras tillage to repair channeling from soil erosion, better soil biological health that improves soil structure and nutrient cycling, and/or less expensive management of weeds, diseases, and/or pests are at least sufficient to pay for seeding costs (Bergtold *et al.*, 2017; Roesch-McNally *et al.*, 2018). Based on survey for farmers across the US, many of the soil benefits increase over time and some benefits only occur periodically depending on conditions, so it may take 3 years to just cover seeding costs and 5 years of continual use to have total benefits that exceed annual costs for seeding cover crops (Myers et al. 2019). Thus, the benefits are difficult to quantify exactly. Nevertheless, the benefits can be high; O'Reilly *et al.* (2011) reported on private value of cover crops as high as \$600 ha^{-1} yr^{-1} for seed corn in southern Ontario. However, in some cases, cover crops can negatively affect the agronomic performance including yield of a subsequent crop. The magnitudes of benefits likely vary regionally and that would greatly affect adoption by region. The lack of experience with cover crops in many areas of Canada makes it difficult for land managers to accurately assess the benefits and risks of cover crops.

More research is needed on the short- and long-term benefits of cover crops to improve decision making from farmers and society regarding cover crops across Canada.

D. Co-benefits

Positive

With the possible exception of increased N_2O emissions, in a meta-analysis, Daryanto *et al.* (2018) found that, overall, the ecosystem services from cover crops are positive and they should be a recommended practice for all croplands. Cover crops reduce dust from wind erosion (Baumhardt *et al.*, 2015), increase biodiversity of soil organisms (Elhakeem *et al.*, 2019) and increases animal population by providing browse, nectar, and/or cover. The reduces soil erosion, increases soil health including organic carbon (Daryanto *et al.*, 2018). They reduce nitrate leaching (Thapa et al. 2018) and can reduce nutrient loss in runoff (Dabney *et al.*, 2001). There are increased economic opportunities in rural areas for growing and processing cover crop seed and for potential contracted services of planting and/or terminating cover crops.

Negative

There is concern about cover crop increasing P losses in winter and spring runoff (Daryanto *et al.*, 2018), an important potential P-loss pathway for Canada (Liu *et al.*, 2019). However, limited field studies with cover crops in Canada have not shown an increase in P loss (Lozier *et al.*, 2017; Schneider *et al.*, 2019). Further investigation is needed to determine if cover crop adoption may need some restrictions because of P losses to surface water (Liu *et al.*, 2019).

E. Limitations and opportunities

Need for research

The need for new investigation to develop data on past, current, and future cover crop use and to develop better estimates of cover crops on GHG emission and removals is outlined under C. Changes needed to include cover crops in the Canadian National Inventory Report.

The various agronomic benefits of cover crops directly from the cover crops themselves (and their effect on soil health needs to be better quantified to inform land managers about merits of cover crop adoption. The effect of cover crop on N balance and on the response to N of important subsequent cash crops requires more research across Canadian conditions.

The effect of cover crops on P loss to surface water requires more research to determine if there needs to be restrictions on cover crop adoption in some watershed.

Research and development on cover crop species, mixes, and cultivars that are provide maximum agronomic and soil benefits under conditions of low potential biomass production is important, particularly outside of warm and moist Canada. Interseeding typically improve the performance of cover crops when there is a short season after cash crop harvest. Therefore, developing practical low-cost techniques to interseed cover crops into a wider range of cash crops is needed.

Programming

Although different, there are some similarities between cover cropping and intercropping in that they both aim to produce more biomass than conventional cropping practice. Therefore, they could be within the same programming.

There is also some potential conflict between cover crops and intercropping. Cover crops are very well suited after winter wheat but if the winter wheat is used for relay cropping with soybean, that intercrop is less well suited to cover crops. Intercropping would not be amenable to interseeded cover crops. The land after intercropping may be drier than after monocrops and those drier conditions would make cover crops riskier.

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Intercropping

A. Methods for Estimating Emissions

a. Description of emission source and mechanisms

Introduction

Intercropping consists of growing two or more grain crops together. Typically, in short growing seasons, the crops are harvested together, and then the grains are separated. Intercropping is more common in organic production systems but is gaining interest for application in conventional production systems.

The main advantage of intercropping typically produces land-equivalent ratio (LER)>1. LER is defined as:

$$LER = \frac{YC1_{intercrop}}{YC1_{monocrop}} + \frac{YC2_{intercrop}}{YC2_{monocrop}}$$

Where $YC1_{intercrop}$ is the yield (kg/ha) of crop 1 in the intercrop, $YC2_{intercrop}$ is the yield of crop 2 in the intercrop, $YC1_{monocrop}$ is the yield of crop 1 as a monocrop, and $YC2_{monocrop}$ is the yield of crop 2 as a monocrop.

A LER > 1 means that the amount of grain produced from a unit area of intercrop is greater than that from the intercrop being grown as monocultures on their fraction of the intercrop unit area To illustrate, 1 ha of intercrop of crop A and B would produce more total grain than that from 0.5 ha of crop A alone plus 0.5 ha of crop B alone. If 30% more grain was produced, then the LER would be 1.3.

Intercrops of legumes and non-legumes have been shown to be complementary and produce better of use efficiencies of use of water, nutrients, and light compared with monocrops (Duchene *et al.*, 2017). This provides an N advantage to the non-legume from N transferred from the legume (Chapagain and Riseman, 2014; Duchene *et al.*, 2017). Intercropping with a non-legume also increases biological N fixation of the legume in the intercrop (Chapagain and Riseman, 2014; Cong *et al.*, 2015). Other agronomic advantages of intercropping are reduced disease pressure and better weed suppression (Rob *et al.*, 2015; Gu *et al.*, 2021). For example, mycosphaerella blight has become the most widespread and economically damaging disease in pea in Canada but its incidence can be greatly reduced when the pea is intercropped (Dowling *et al.*, 2021). For legumes that require a drying condition to maximize grain yields, such as chickpea and lentil, then the water extraction by the non-legume intercrop can increase the legume grain yield (Dowling *et al.*, 2021).

For legumes that require drying conditions to maximize grain yields, such as chickpea and lentil, then the water extraction by the non-legume intercrop can increase the legume grain yield (Dowling *et al.*, 2021). Considering the effects of intercrops on both disease and

The engineering challenges for planting the intercrop appropriately is a limitation. When the intercrops can be growing together in the same row, this planting method is not a limitation for many modern planters. However, if the best configuration for intercrops requires planting in different rows, then, either the planters is modified and made more complex to produce that configuration or the two the intercrops are planted in separate passes with the planting equipment.

Growing intercrops greatly limits the options for herbicides to control weeds so requires an integrated weed management strategy over time.

Managing the harvest so all harvested grain is in good condition with minimal harvest losses is a challenge that challenge will differ annually depending on the weather. For intercrops grown together, this requires careful choice of crops and cultivars so that reach harvest state close together in time.

The most important limitation to intercropping is the extra operation to separate the intercropped grains that are harvested together. However, if the grains are harvested together but fed as a mixture as livestock feed then separation is not necessary. Some intercrops are grown for silage and there is also no need for separation. Legumes are often included with cereals in intercropped mixture to benefit from the N fixation of the legumes and higher protein content of legumes. Examples of intercrops grown for silage include pea with barley and soybean with corn. Also, for relay cropping where one of the intercrops is harvested separately before the other, then grain separation is not necessary. For relay cropping, the limitation then becomes the added time and cost for two harvest operations.

To separate two grains that are harvested together requires additional equipment for separating grains based on physical characteristics. This also limits the types of intercrops since the two grains must differ physically sufficiently for high-throughput separation at relatively low cost. It also requires additional grain storage as some storage is needed for the mixed grains before separation. Labour availability to separate the busy harvest season is a limitation. This labour constraint is made worse by the fact that conventional seed cleaning equipment does not have a high throughput since they were designed to separate a relatively small proportion of material from mixture that is primarily one grain. For mixed grains from intercropping, a large proportion must be separated from the mixture. Commercial equipment purpose-designed for high throughput separation of the grain mixtures from intercropping is not available.

ii) Intercrop in this study

There are many intercrop possibilities but those involving a legume and non-legume are particularly attractive to increase efficiency of using applied N fertilizer. Four non-legume-legume intercrops were included in this study were: 1) pea and canola, 2) chickpea and flax, 3) barley and lentil, 4) lentil and canola, and 5) winter wheat and soybean. Pea and canola and chickpea and flax are relatively well studied in Canada (Dowling *et al.*, 2021). The barley and lentil and canola and lentil has been shown to work successively elsewhere (Martin-Guay *et al.*, 2018, Dowling *et al.*, 2021). Canola and lentil are much different seed sizes so can be separated easily and barley and lentil can also be mechanically separated relatively easily (Milligan, 2009). Winter wheat and soybean is grown as relay with the soybean seeded into the growing winter wheat in the spring and harvested in a separate operation after the wheat. There is considerable interest in this cropping practice (McIntosh, 2021).

Some intercrops are hayed or ensiled to feed cattle. Mixing a legume with cereal is common in such intercrops to produce a feed that is both high energy and high protein while taking advantage of the N fixation capability of the legume to reduce N fertilizer needed. Example intercrops include pea and barley and soybean and corn. These intercrops were not considered in this study because most of the crop residue is removed in harvest so the intercrops would not increase SOC much beyond monocrops.

To identify the potential area for these crops, ecodistricts (note, there are 443 ecodistricts in Canada with agriculture) were identified that produced the two crops in the intercrop. This is obviously only an indicator as it shows that the two crops are commercially grown in the same ecodistrict (each ecosditrct is defined as having a single climate and related soil landscapes), but does not show they are grown on the same farm. The potential intercrop area was twice the smallest of the area of the two intercrops was used as the potential area of the intercrop. To illustrate, if 1100 ha of chickpea and 1350ha of flax were grown in an ecodistrict, the potential chickpea and flax intercrop area was 2*1100ha = 2200ha with 250 ha of flax remaining as a monocrop. This done so there would be no change required of the area of other crops from intercropping. The latter was a requirement for this investigation. Canola could appear in two different intercrops but only the canola area in excess to that needed for intercrop with existing pea area was available for the canola-lentil intercrop.

Typically, the more competitive non-legume in the intercrop will yield most (Echarte *et al.*, 2011; Chapagain and Riseman, 2014). The row configuration and plant populations were assumed to produce a legume yield per unit area that is at least one-half of that of a monocrop legume. The non-legume was thus assumed yield more per unit area than one-half that for a monocrop non-legume. With this assumption there is at least equal production of the grains with and without intercropping.

The winter wheat and soybean relay cropping was limited to southern Manitoba (prairie ecozone), southern Ontario and Quebec (mixed wood plains ecozone), and Atlantic maritime ecozone of Quebec, New Brunswick, PEI, and Nova Scotia). The rationale was that areas outside of the above would not likely have the precipitation and/or heat after winter wheat harvest needed for good soybean production.

dThe more efficient use of water, nutrients, and sunlight produce more biomass than monocrops. This increases the C (carbon) input to the soil compared to monocrops. The LER of biomass is typically greater than the LER for grain (Chapagain and Riseman, 2014; Cong *et al.*, 2015). This is because the competition between intercrops produces relatively more leaf, stem, and root biomass than in monocrops. The LER if usually lower in drier areas and, in fact, the LER for grain can be less than 1 for intercrops under dry conditions, such as southwest Saskatchewan (Dowling *et al.*, 2021). However, to understand the effect on SOC, we only need the LER for C input (residue biomass) which is expected to more consistent and higher than the LER for grain. Based on literature, the C input LER were chosen as: 1.1 for semiarid Prairies (Brown and Dark Brown soil zones), 1.2 for rest of western Canada and western boreal shield in Ontario, and 1.3 for rest of Ontario eastward. These zonal divisions within provinces are based on the divisions existing in Canada's NIR.

The SOC should increase with increased C input from intercropping and intercrop have been shown to increase SOC over monocrops (Cong *et al.*, 2015). The SOC change was estimated by using the IPCC Tier 2 steady state model (IPCC, 2019), applied at the ecodistrict level, a method that was implemented in the Canadian NIR starting in 2022. The model is based on the well-accepted Century model and estimates SOC change based on average C input to the soil and annual weather. The simulation started in 1971, after SOC initialization to match the SOC in Canada's National Soil Database (Agriculture and Agri-Food Canada, 2021), using actual crop yield and weather data to 2019. This modelling approach should represent average Canadian soil conditions as affected by previous crop production practices. The 2020 to 2030 was set to the average weather factors and crop area for 2015-2019. The yields for the 2020 to 2030 period were increased based on continuation of 2005-2018 trend on an ecodistrict basis. The SOC change in 2030 was then estimated as the difference between baseline and scenario for SOC change between 2029 to 2030. Note that the modelled SOC change in 2030 will depend on both the crop yields and weather that occur preceding and during 2030. Therefore, when real data is available during that period in the future, the modelled SOC change will be different from the estimates for this study. Also note that the rate of SOC change will decrease if the amount and LER of intercrop remains unchanged from 2030 onwards as the SOC amount moves towards equilibrium between C inputs and C losses. This effect will be captured using the Tier 2 model for SOC.

The assumption was that the area of intercrop would receive the same total N as one-half the area in non-legume intercrop. Therefore, total fertilizer N application across the intercrop would be the same as that across the two crops grown as monocrops. Under the production assumptions, the total N return to the land in crop residue would be increased by intercrop. However, where studied, N2O emissions are less in intercrop than as monocrop (Dyer *et al.*, 2012; Huang *et al.*, 2019). For simplicity, we assumed that the two effects balance so there is no net increase in N2O emissions from intercrop. The N2O footprint of the non-legume crop in the intercrop would be less than as a monocrop but absolute total N2O emissions across the total area of intercrop, with the above assumptions, are the same as the total area with the intercrops grown as monocrops.

Scenarios were developed as a percent of total annual crop area by province. That area was split between intercrop by their proportion of total crop area.

b. Intercrops in the NIR

The impacts of intercrops are not currently captured in the NIR. At present, there is an unknown amount of intercropping.

c. Quantities of emissions under consideration and trendlines

Intercropping will influence the change in SOC by changing C input to the soil. Against the backdrop of C inputs to the soil from crops and manure, the impact of intercropping will be small.

For relay cropping, there is an additional sowing and harvest operation. Dyer and Desjardins (2003) estimate emissions of 0.038 kg CO2e/ha for sowing and 0.030 kg CO2e/ha for harvest in central Canada. We could not find an estimate for grain separation required for other intercrops in the literature. Therefore, we used the 0.030 kg CO2e/ha emission for harvest as an estimate for grain separation since the combine-harvester involves the actions needed for separation of intercropped grains: separation of grain from much other material plus lateral conveyance and elevation of the grain. If electricity with a low carbon footprint is used for some or all of the power used for on-farm separation of intercropped grain, the emissions would be lower for that operation than the diesel powered combine-harvester.

The maximum potential area and mitigation from this study are listed in Table 1. The largest potential area for intercrops were in Saskatchewan followed by Alberta, Ontario, and Manitoba. Based on concurrence of the crops for the included intercrops, 22.7% of Canadian cropland could be those intercrops. Pea-canola was most common opportunity and that potential intercrop existed in all provinces except Newfoundland and Nova Scotia. Based on concurrence, the area for flax-chickpea intercrop is much smaller than the other intercrops and was only in Alberta and Saskatchewan.

The estimated 2030 SOC gains for 2030 are listed in Table 2 below by province with 2% of annually cropped area with intercrops. The scenario ramped up adoption linearly from no intercropping in 2024 the set percent in 2030.

About of Canada's annual cropland could be in the four intercrops used in this study based simply on the existing geographical coincidence of production of the crops involved.

Table 1 Maximum indicated potential mitigation from intercropping.

Jurisdiction	---------potential area of intercropping (ha) --------------------					SOC change	Additional Operations	Mitigation
	flax-chickpea	pea-canola	lentil-barley	w. wheat-soybean	lentil-canola	(t C/ha/yr)	(t CO2e/ha/yr)	(Mt CO2e/yr)
AB	15125	1543285	285757	0	1934	0.15	0.030	0.9851
BC	0	40448	194	0	0	0.28	0.030	0.0403
MB	0	159554	12633	108752	0	0.17	0.045	0.1609
NB	0	27	0	527	0	0.14	0.066	0.0002
NF	0	0	0	0	0	0.00	0.000	0.0000
NS	0	0	0	355	0	0.15	0.068	0.0002
ON	0	5984	0	867720	0	0.22	0.068	0.6406
PE	0	97	0	6172	0	0.12	0.067	0.0023
QC	0	3004	19	30541	0	0.18	0.065	0.0201
SK	108830	1713345	1099960	0	1409346	0.11	0.030	1.6127
Canada	123955	3465743	1398563	1014067	1411280	0.14	0.035	3.4625

Table 2 Intercropping area and SOC change and mitigation with 2% of provincial annual cropland in intercrops*

Jurisdiction	------------------ area of intercropping (ha) ---------------------					SOC change	Additional Operations	Mitigation
	flax-chickpea	pea-canola	lentil-barley	w. wheat-soybean	lentil-canola	(t C/ha/yr)	(t CO2e/ha/yr)	(Mt CO2e/yr)
AB	1433	146171	27065	0	183	0.16	0.030	0.0973
BC	0	3698	18	0	0	0.28	0.030	0.0037
MB	0	46304	3666	31561	0	0.17	0.045	0.0474
NB	0	27	0	527	0	0.10	0.066	0.0002
NF	0	0	0	0	0	0.00	0.000	0.0000
NS	0	0	0	355	0	0.13	0.068	0.0001
ON	0	392	0	56854	0	0.22	0.068	0.0427
PE	0	32	0	2061	0	0.12	0.067	0.0008
QC	0	2019	13	20529	0	0.18	0.065	0.0136
SK	7778	122446	78610	0	100720	0.11	0.030	0.1153
Canada	9210	321089	109371	111887	100903	0.14	0.037	0.3212

*based on current coincidence of crops, there is no potential area for intercrops in Newfoundland and Labrador and a maximum of 1% and 1.6% were available for intercrops in New Brunswick and Nova Scotia, respectively.

B. Details of proposed emission-reduction measures

a. Mechanisms of reduction action

Reduction action comes from achieving LER > 1 through intercropping. Good agronomic management is required to achieve this result. Suitable seeding systems will be needed. Importantly additional equipment will be needed for grain separation except for relay cropping.

There is a potential economic advantage from the overyielding (grain LER>1, but programming to incentivize intercropping would create experience, enable optimal seeding rates and configurations to be determined by trial and error, and provide a market for modified and/or purpose-designed machinery needed for planting and harvested grain separation for successful intercroping.

b. Quantification of Emissions

Better understanding of the effect of intercrops on reducing N2O emission would increase the mitigation. The effects of intercrops on herbicide, fungicide, and pesticide application would also improve estimates of the emission; the application is expected to be reduced from monocrops. The SOC change would be affected by the estimated increase in C input. Finally, more exact GHG emissions for practical high throughput grain separation systems is needed.

c. Exploration of Assumptions

Enough is known that LER for grain is usually greater than 1. However, it can be close to or less than one, and, in some cases it can be much higher than 1.3; Dowling *et al.* (2021) reports on a flax-chickpea intercrop in Manitoba where both the flax and chickpea yielded more than their monocrop counterparts, an LER >2. More knowledge of the performance of intercrops for Canada is needed both for grain yield and biomass residue yield. The latter provides the C input that drives SOC change. More knowledge is also needed on the optimal N fertility management and on how intercrop affect N2O emission.

d. Current adoption

We assume negligible current adoption. Data is not collected on intercrops other than mixed grains that are used for feed and not separated into individual grains.

e. Barriers to adoption

Seeding: Seeding mixed grains at the same time in the same row is not difficult with modern air seeders. However, having different grains in different rows, that can be better than mixed, would require modifications. Seeding is a busy time so making modifications to seeding equipment between different monocrop and different intercrops would be a barrier. Seed costs will be higher for the less competitive crop in the mixture compared to having the same production done as a monocrop.

Crop management: Intercrops will have limited options for herbicides so some weeds may be difficult to control. Agronomic information on thresholds for control of various pest and diseases are not established for intercrops.

Harvest: Managing harvest so that all grain is in good condition while minimizing harvest losses is difficult. There is risk of shattering and/or weathering losses if harvest of one crop has to be delayed because of the other crop. If harvest of one crop is earlier than optimal to meet the harvest needs of the other crop, then there may be quality and/or quantity loss.

Post-harvest: Without a doubt, separating grains is the major barrier. It is an additional operation required during what is typically the busiest season. Equipment for conventional seed cleaning is designed to separate a relatively small amount of material from one grain. When applied to an intercrop grain mixture, it has low throughput because it is not designed to separate a large amount of material from one grain. If the harvested grain from on crop in the intercrop, the drying while the other does not, the whole mixture may need to be aerated or dried if it can not be separated quickly enough. This could involve extra costs and facilities.

Grain value downgrade: Suboptimal harvest timing that lower grain grade and/or excessive contamination of the grains with the other grain will lower the quality and, thereby, the market value of the grain. Where the grain production in contracted, there can be significant penalties if the grain does not meet the minimum quality required by the contract.

Crop Insurance: Intercrop would only be eligible as a single crop under crop insurance so coverage would be partial.   Since the yield as intercrop is expected to be less than as a monocrop, claims for one crop would lower the yield for coverage of that crop as a monocrop for that farmer.

In summary there are important extra costs, particularly of labour during busy periods, and extra risks associated with intercrops. If a LER above 1 is achieved consistently, the additional revenue could make it profitable without external incentives.

C. Changes Needed to Include in the National Inventory

Canada's method for reporting grain production assumes monocrops. Therefore, a new classification of intercrops would be needed. Further, the yields of the grain constituents in the intercrop would be needed so that production of individual grains could be determined. Grain production estimates have multiple uses in the market so the reporting changes required to account for intercrop are needed more fundamentally that just for GHG reporting.

If intercrop data is collected and there is knowledge of how to convert yields to C input for intercrops, there is no major changes required to the inventory since SOC change is based on SOC modelling.

D. Co-benefits

The greater amount and types of residue can improve soil health and resistance to soil erosion. Some wildlife may prefer the more complex canopy and vegetation of an intercrop to the relatively uniform canopy of a monocrop.

E. Beyond current scope

Although different, there are some similarities between cover cropping and intercropping that they could be within the same programming. Cover crops are well suited after winter wheat but if winter wheat land is used for relay cropping with soybean, that intercrop is less well suited to cover crops. Failed intercrops where one crop does not do well could be described as a form of cover crop and failed crop would be expected to provide some soil health benefits similar to cover crops. Some growers will include a non-harvested cover crop, often a legume, with a intercrop do gain more of the fertility and other soil health benefits of a diverse crop mixture.

Intercrops create new agronomic challenges. Intercrops crop rotation design more complex if one want to keep a set temporal separation between specific crops. The winter wheat-soybean relay cropping adds the requirement that preceding crop to winter wheat is harvested early enough for timely seeding of winter wheat after harvest. Corn is usually harvested too late for timely winter wheat sowing. A corn-soybean-winter wheat/soybean intercrop may be feasible but has successive soybean crops that may impact soybean pests and diseases. Therefore, there is need for more research and development of optimal agronomic management of intercrops including how they are best incorporated into long-term cropping systems.

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Rotational Grazing (land emissions)

A. Methods for estimating emissions

Introduction

In Canada, grazing land covers 14.3 million hectares is 2019. Grazing land consists of natural land used for grazing and tame pasture. The former is normally permanent with low level of external inputs and consists of native species, a mix of native and tame species (possibly seeded tame or invasive tame), or primarily tame species (the latter sometimes called naturalized grassland (Sheppard et al., 2015)). In contrast, tame pastures are typically terminated and reseeded periodically when productivity declines, when desired species are not present, and/or when there is a presence of excessive undesired plant species. In 2006, 32% of tame pasture managers rejuvenated tame pasture every 5 years or less, and 40% every 6-10 years, with 11% never being rejuvenated (Sheppard et al., 2015). In 2011, 13% of tame pastures received fertilizer (Sheppard et al., 2015) and, on average, 22% of the vegetation sward was legume (Sheppard et al., 2015) – two practices that increase productivity and forage quality.

Rotational grazing is the practice of moving grazing cattle through a set of paddocks. It is in contrast to continuous grazing where cattle are in a single paddock through the grazing season. The main advantage of rotational grazing is increased vegetation growth (Alemu et al., 2019; Sanderman et al., 2015) and better graze quality (Wang et al., 2015), although Popp et al. (1997) found no significant effect on either herbage or quality from rotational grazing in Manitoba. There is a wide range of grazing practices within rotational grazing. Basic rotational grazing provides the opportunity for grazed plants to recover. Intensive rotational grazing has much shorter grazing periods, moving animals more often, to reduce stress on the plant from grazing (sometimes referred to as avoiding the "second bite" of any plant during a grazing period) and allowing for sufficient time for plant recovery after grazing. Unfortunately, there are not widely accepted definitions of this range of practices.

For this analysis we divided rotational grazing into 4 categories:

1) Continuous: no rotational grazing, continuous season-long grazing
2) Basic: grazing in which animals are rotated through multiple paddocks at least once.
3) Intermediate: multiple paddocks, in which animals are rotated through each paddock two or more times in a season and/or grazing is intentionally deferred in each paddock during critical vegetation growth periods over time. Therefore, grazing periods are shortened and there are longer and more strategic rest periods between grazing than basic rotational grazing.
4) Intensive: short grazing duration (< 8 days) per paddock with rest period between grazing on each paddock based on sufficient time to reach vegetation state consistent with long-term vegetation health. In some of these systems, the livestock may be moved to a new paddock daily.

Adaptive multi-paddock grazing (AMP) is rotation grazing with flexibility to adjust grazing based on conditions of the pastures and needs of the herd. Both intermediate and intensive rotational grazing could fit under the definition of AMP.

Rotational grazing is applicable for all grazing animals including sheep, goats, horses, and cattle. This report will look at rotation grazing specifically for beef cattle because this is the largest livestock group for pasture management in Canada.

a. Description of emission source and mechanism

This section deals only with SOC changes expected from adoption of rotational grazing. Changes in enteric emission from grazing ruminants is provided in report section of rotational grazing.

b. NIR reporting

The National Inventory report does not report any GHG effect for rotational grazing. Data would need to be collected on grazing practices.

c. Quantities

The mitigation is from soil C sequestration from increase in the adoption of rotation grazing.

B. Details of emission reduction measures

a. description of proposed reduction measures

Background SOC Changes

The grasslands of Canada are gaining an average of 130 kg C ha^{-1} yr^{-1} during the early 2000s based on atmospheric inversion models (USGCRP, 2018), although this value refers primarily arctic tundra grasslands in additions grazing land. In the Great Plains, grasslands in the same period were a sink of 240 kg C ha^{-1} yr^{-1} and are expected to remain a sink at a similar rate to 2050 (USGCRP, 2018). Nevertheless, the rate varies widely by year, including being a source in drought years, in response to weather. Grazing generally increases SOC compared to no grazing (McSherry and Ritchie, 2013) with rates of 72 to 190 kg C ha^{-1} yr^{-1} in the northern Great Plains (Wang et al., 2016; Wang et al., 2014). Therefore, much of observed increases may be due to recent improved grassland management that is restoring SOC that was lost from past poor management. In agreement with this, Wang et al. (2014) relates the increase in SOC on rangelands from simply grazing in the Northern Great Plains to likely restoration of SOC after mismanagement, particularly over stocking, in the first half of the 20th century. Similarly, initial findings showed that European grasslands appeared to be a continual sink of C as high as 1.29 Mg C ha^{-1} yr^{-1}, sufficient to more than offset the emissions of CH_4 and N_2O from the grazing livestock (Soussana et al., 2010). However, Chang et al. (2016) showed that this SOC increase is the result of significant lowering of stocking on European grasslands due to policy changes during the 1980s and 1990s. Smith (2014) cautions that grasslands cannot be expected to be a perpetual sink as they will come to an equilibrium C after which there will not be sustained increases in C stocks. So, the general grassland C sink will decrease over time as it approaches a new SOC equilibrium. For this study, we estimated the difference in C sequestration to continuous so any SOC recovery from an historically more soil-degraded condition of the grazing lands would cancel out. However, some of the studies that do not have good pairing with a control grazing system may include the differing amounts of SOC recovery for the control and the rotational grazing systems.

SOC change from adoption of rotational grazing

New adoption of rotational grazing represents an opportunity to increase SOC on pastures. The available data (Appendix) does not allow robust analysis of additional C sequestration from adoption of rotational grazing in Canada since there are few studies for Canada and results are variable elsewhere. The general results globally are that rotational grazing increases SOC (Byrnes et al., 2018). This is supported by the subset of studies that have relevance to Canada as all that are based on measurements either had increase or no effect on SOC; there was only one study measuring SOC loss, and that study simulated grazing with mechanical harvest rather than livestock. If rotational grazing had no benefit and the distribution of results is symmetrical about the mean, we would expect there to be more studies showing loss of SOC from rotational grazing.

Byrnes et al. (2018) found that rotational grazing had greatest positive effects in humid climates. Compared with continuous grazing, grazing exclusion tends to increase SOC in wetter climates and decrease SOC in drier climates with the effect being linear with precipitation in the range of 200 to 1000 mm (Derner and Schuman, 2007; Hu et al., 2016; McSherry and Ritchie, 2013). This also supports the concept that rotational grazing will be more effective for increasing SOC as precipitation increases as the long vegetation recovery time without grazing inherent to rotational grazing mimics some aspects of no grazing.

Having legumes in pasture has been shown to improve C sequestration (Conant et al., 2017; Fornara and Tilman, 2008; Henderson et al., 2015) and improve herbage quality (Bélanger et al., 2017; Peprah et al., 2018). The recovery periods and reduced sustained grazing stress with rotational grazing improves longevity and maintenance of seeded legumes (Forsythe, 2018). We assumed that all natural and tame pasture under intensive grazing will also be managed so that they will have sufficient legumes to provide N needs of the sward whereas the continuous and basic scenarios may or may not have adequate legume content.

Wang *et al.* (2021) point out the difficulties of detecting the soil impacts of grazing systems including interactions between climate, soil, vegetation, animal species, and management. Other difficulties include variation in actual pasture management practice so that the management over time is hard to place into a specific category.

Based the available data (Appendix) and our expert opinion, we estimated a conservative average C sequestration rates that would be applicable over 30 years (Table1). For this purpose, we divided Canada into 3 general climatic zones: moist and warm Canada (Mixed Wood Plains, Pacific Maritime, and Atlantic Maritime Ecozones), dry Canada (Semiarid Prairie NIR reporting zone), and moist and cool Canada (Montane Cordillera, Boreal Plains, Subhumid Prairie, and Boreal Shield Ecozones). We expect there will be a wide range of values on a paddock-by-paddock and year-by-year basis depending on the initial state of soil degradation when rotational grazing is adopted, the weather patterns, and, especially for natural pasture, the initial species mix. Note that rates of 100 kg C ha^{-1} yr^{-1} or less, even over a 30-year period, would be difficult to detect through measurement and so may be reported in scientific literature as no change (Maillard *et al.*, 2017). The values are highly uncertain due to limited amount of evidence specific to Canada. Therefore, we suggest that uncertainties would be in the order of ±100%, i.e., ranging from no change to double the derived gains.

Table1: Estimated mean rates of C sequestration from changing from continuous grazing for different levels of rotational grazing and pasture area for climatic zone in Canada.

Pasture Type	Grazing	Zone Moist and warm Canada*	Dry Canada*	Moist and cool Canada*
Natural land	2019 Area (M ha) =	0.962	6.228	2.437
	Grazing method	--- C sequestration from continuous (kg C ha^{-1} yr^{-1}) ----		
	Basic	0	0	0
	Intermediate	80	20	40
	Intensive	300	90	180
Tame	Area (M ha) =	0.244	1.696	2.746
	Grazing Method	--- C sequestration from continuous (kg C ha^{-1} yr^{-1}) ----		
	Basic	0	0	0
	Intermediate	100	30	60
	Intensive	400	120	240

* Moist and warm is mixed wood plains, Atlantic maritime, and Pacific maritime, ecozone, dry is the semiarid Prairie zone of Alberta and Saskatchewan in the NIR, moist and cool Canada is the remainder of Canada that is either situated north of warm and moist or in subhumid western Canada.

Adoption Potential

The primary barriers to adoption of rotational grazing are cost for necessary watering facilities and fencing, increased labour requirement to carefully monitor pastures and to move cattle between paddocks. A well-designed grazing plan is necessary both to design the infrastructure and to operationalize rotational grazing.

By 2030, we assumed that there is technical potential to have substantial increases in advanced basic and intensive rotational grazing, particularly in the Moist and Warm and Moist and Cool climates. Table 2 lists the estimated current and potential 2020 adoption rates. To realize this potential, there needs to be sufficient capacity for grazing practices, either from advisors or from farmer/rancher training, and building confidence that rotational grazing will have economic benefits that are larger than the increased costs. Cost-share for the costs, especially for up-front costs for infrastructure improvements, help build that confidence of positive net economic benefit from rotational grazing. With greater experience and more evidence of positive results gleaned from nearby adopters over time, more farmers should increase confidence of the merits of adoption without necessarily requiring any cost-share. Water availability was assumed to limit the extent of adoption of intensive grazing in dry climates, particularly for natural pastures.

The scenario of current (see subsection d) and technical potential adoption and the associated GHG reduction from a baseline of current adoption are shown in Table 2.

Table 2. Current and technical potential adoption rates by 2030 by zone.

	Climate					
	Moist and Warm		Dry		Moist and Cool	
Grazing System	Current	2030	Current	2030	Current	2030
	------------ Natural Pasture Adoption Rates (% of area) --------------					
Basic	50	20	50	50	50	25
Intermediate	10	20	10	25	10	25
Intensive	10	55	5	15	10	45
	------------ Tame Pasture Adoption Rates (% of area) --------------					
Basic	40	20	50	35	40	20
Intermediate	20	15	10	35	20	20
Intensive	10	60	5	25	10	55

b. Quantification of potential removals

Table 3 shows the additional C sequestration assuming the technical potential is reached in 2030 compared to continuation of current adoption rates to 2030. The estimated C sequestration is small.

Table 3. Maximum C sequestration mitigation by jurisdiction and pasture type.

Jurisdiction	2030 Mitigation (Mt CO2e/yr)		
	Tame	Natural	Total
AB	0.384	0.123	0.507
BC	0.040	0.021	0.062
MB	0.089	0.031	0.120
NB	0.004	0.007	0.011
NL	0.000	0.000	0.000
NS	0.006	0.001	0.008
ON	0.074	0.150	0.224
PE	0.005	0.006	0.011
QC	0.038	0.040	0.078
SK	0.243	0.076	0.320
Canada	0.885	0.457	1.341

The effect of rotational grazing on enteric fermentation is described under that livestock mitigation pathway elsewhere in this report.

c. Assumptions

The primary assumption is the C sequestration rates. Since there is much variation in the literature and relatively few studies, the rates are very uncertain. The adoption rates are also assumed. The duration of applicability of the sequestration rates is also uncertain.

d. Current adoption

Currently, about 50% of beef producers use rotational grazing according to 2016 Census of agriculture (Beef Cattle Research Council, 2019) with a percentage adoption similar across provinces. In 2011, about 25% of beef producers reported using continuous grazing on tame pasture and 35% using continuous grazing on native pastures (Sheppard et al., 2015). Fully 66% of beef producers had 2-4 paddocks for tame pasture and 58% had 2-4 paddocks for native pastures in 2011 (Sheppard et al., 2015). These would be classed as basic rotational grazing by our definition. In 2014, 10.8% and 7.8% of cow-calf producers in western Canada used intensive rotational grazing management on owned tame pasture and native range, respectively. In northern Ontario and Quebec in 2015, about 30% used continuous grazing, 50% basic rotational grazing and about 20% use more intensive rotational grazing (Beef Cattle Research Council, 2019). The lack of standard definitions makes it difficult to interpret and reconcile surveys. Because of characteristics of different pasture areas and the different nutritional requirements of different groups of livestock, a producer could have some pasture area with basic rotational grazing and some with more intensive rotational grazing; this adds to confusion when survey asks for only one type of grazing system.

Kristine et al. (2021) surveyed 97 pastures on 28 ranches distributed across southern and central Alberta to assess the effect of grazing and other factors on range health. The ranchers where volunteers so may have been more inclined to be interested in range health and thereby possibly more likely to use rotational grazing. Nevertheless, only two pastures, both tame, had a grazing period of 1 day. Eighteen pastures (included 4 native pastures) had grazing period of 2-8 days so would be intensive in our typology. Thirteen pastures (7 native and 6 tame) had a grazing period over 60 days indicating continuous or a rudimentary basic rotation grazing. Twenty-three pastures had a grazing period of 9-21 days which were assumed to represent intermediate rotational grazing in our typology. The remaining 42 pastures with grazing period between 22-60 days would be basic rotational grazing. Of note, range health scores, for both native and tame pastures, tended to decrease linearly as grazing period lengthened. This is consistent with the concept that transition to more intensive rotational grazing improves the quality of pasture which improves soil quality.

The general trend in Canada is towards increased rotational grazing and a shift towards more intensive rotational where the pasture area is suitable in terms of availability of livestock watering sources and soil-vegetation landscapes amenable to numerous small paddocks. Rotational grazing is promoted by the Canadian beef industry and governments.

Table 2, presented earlier, shows the current adoption estimates.

d. Barriers to Adoption

The barriers to adopting intermediate rotational grazing from continuous or basic practices are more investment in fencing and water capacity and more labour for pasture assessment, grazing infrastructure maintenance, and cattle movement. The adoption of intensive requires even more infrastructure and labour and can involve a lifestyle change because of the need for frequent cattle movement. Producers are probably more likely to move incrementally than to make large jumps in management, i.e., preferring to transition from continuous to basic, from basic to intermediate, and from intermediate to advanced. Consequently, to increase adoption of intensive grazing requires increasing the transition from continuous to basic and basic to intermediate. Farmers might not transition all their herd to an improved grazing management so could have a mix of grazing practices during transition.

C. Changes to NIR

To include the C sequestration for rotational grazing requires both good data on grazing management over time and good data on C sequestration rates. The complexity of grazing systems over time will make it difficult to categorize grazing systems and assigning a C sequestration rate to that land. Methods based on estimating C input to the soil and then modelling C change may be more practical. The C input can be estimated from estimates of above-ground biomass over the growing season using remote sensing. The SOC can than be modelling in response to changes in C input.

D. Co-benefits

Positive

Rotational grazing has important co-benefits of maintaining and increasing biodiversity. Rotational grazing improves soil health (Byrnes et al., 2018), increases above and below ground biodiversity (Reshmi et al., 2020; Teague and Kreuter, 2020), and maintains legumes that reduce need for nitrogen fertilizer (Forsythe, 2018). Natural grazing lands are important reservoirs of plant, animal and soil biota biodiversity within the land base and support biodiversity of many animals that use grasslands but also migrate beyond that grazing land base.

Negative

Moving to more advanced and intensive rotational grazing will probably reduce the area of grazing land required as the same amount of cattle can be fed on smaller land area. Other things equal, this leads to a drop in grazing area and incentive to convert grazing land to cropland. This conversion results in loss of biodiversity, loss of soil, nutrients and pesticides to the environment, increased greenhouse gas emissions from nitrogen, and loss of soil organic carbon.

E. Limitations and opportunities

The primary limitation is categorizing the grazing systems, estimating the rates of adoption, and then estimating the impact of the grazing system on C sequestration. The problems with estimates of grazing systems also affects the estimation of the impact of rotational grazing on the enteric fermentation. Nevertheless, there are more advantages to rotational grazing including that adaptive rotational grazing can be a more resilient grazing system to weather stresses. Consequently, investing in a better understanding how rotational grazing impacts GHG emissions and removals is needed to understand how promoting this practice will affect GHG mitigation.

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G. Appendix: Rates of Carbon Sequestration in the Literature

Table A.1: Values of SOC sequestration for rotational grazing

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
-- Multi Continents --						
Global	1-98	Meta-analysis of published results	Rotational vs continuous	Can't be calculated from data provided	(Byrnes et al., 2018)	Rotation 32% higher (ln RR = 0.28)
Temperate (location not defined but model only validated for Montana ranches)	80 (equilibrium (Derner and Schuman, 2007))	Modelling	rotational vs continuous	16-pasture vs 4 pasture, 0-60; 4 pasture vs continuous: 0-1000+	(Ritchie, 2020)	Reduced loss since continuous grazing was estimated to be losing SOC, rotational benefit increases as stocking rate increases
Global		Review of published literature	Rotational (Holistic) grazing vs continuous	0	(Hawkins, 2017)	No evidence of difference from available studies
Global	N/A	Review of published results	Additional from "improved grazing" (assumed to be rotational)	280	(Conant et al., 2017)	
-- Canada --						
Saskatchewan	N/A	Modelling	Change to rotational (basic) grazing	65	(Lynch et al., 2005)	

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
			on tame pasture in Black soil zone			
Saskatchewan	18	measurement	Rotational (intermediate) grazing compared to continuous native species mix established on cropland	200 (0-60 cm)	Iwaasa (unpublished), experiment described in (Alemu et al., 2019)	P=0.09
Manitoba	5	Measurement	Rotational grazing (intensive) on tame pasture vs continuous	340	(Manas et al., 2000)	Results not statistically significant
Alberta	30	Modelling with validated Century model across Alberta	Change to rotational grazing for rangeland	Rotational grazing with long duration grazing: -400 to -100 (loss) Rotational grazing with short duration grazing duration: 100-200 kg C/ha/yr 10% reduction in stocking rate 200-300 across all grazing practices	(Iravani et al., 2020)	
Prairies	?	Measurement across ranches	AMP vs conventional practices	0	(Breitkreuz et al., 2019)	No evidence of difference
Alberta	5	Native rangeland	Deferred rotational vs non grazing	0 difference between treatments	(Dormaar et al., 1997)	Grazing pressure was very light
Alberta	?	Primarily tame pasture	Rotational vs continuous	Rotational had 6% less SOC concentration than	(Pyle *et al.*, 2019)	No pairing between management systems to reduce

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
				continuous, not significant		confounding effects on SOC
Prairies	10+	Ranch grasslands	Adaptive multi-paddock grazing vs non-AMP between ranches	Soil under AMP has increased CH4 uptake and no increase in CO2 or N2O emission	(Shrestha et al., 2020)	Lab incubation study so can not be extrapolated to actual rates in the field
--United States --						
New York, USA	N/A	Modelling (Comet-VR)	Cropland to rotational grazing	3510	(Rosenzweig et al., 2010)	
Virginia, USA	20		Change to rotational grazing on existing pasture	790	(Bosch et al., 2008)	
US grazing lands	N/A	Expert opinion		Rangeland: 70 to 300 Tame pasture: 300 to 1400	(Morgan et al., 2010)	
Michigan	4	Measurement	Rotational (intensive) grazing (change over time, no comparison)	3540	(Stanley et al., 2018)	Authors caution that rate may not continue for long duration
SE US	7	Measurement across farms, comparison	Change from intensive grazing on pasture established on	8000	(Machmuller et al., 2015)	Dr. A. Franzleubers, in written comments to journal,

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
		with row crop agriculture	long-term cropland			points out flaws in analysis and suggests that rate of 1590 kg C/ha/yr is more plausible based on the study data
SE US	10-29	Tame pasture	AMP vs continuous grazing; 5 pairs	500 for 0-100 cm; 400 for 0-30 cm	(Mosier *et al.*, 2021)	
South Dakota	30+	Measurement across ranches	Rotational grazing vs continuous	0	(Hillenbrand et al., 2019)	
Texas (tallgrass prairie)	15 yr	CO_2, N_2O, and CH_4 fluxes for 2 years on neighbouring ranches	Continuous vs Adaptive multi-paddock grazing (AMP=rotational grazing)	CO_2 emissions smaller proportion of plant production for AMP vs continuous, AMP N_2O fluxes about ½ of continuous	(Dowhower et al., 2020)	Not possible to derive a SOC sequestration rate
Texas (tallgrass prairie)	9+	Soil sampling on neighbouring ranches	Continuous vs AMP	1300 (continuous heavy vs AMP heavy stocking rate) 130 continuous light stocking vs AMP heavy stocking rate)	(Teague et al., 2011)	Data does not allow for precise derivation of rate, value based on 15 years in practice.
Wyoming	11	Experiment on native rangeland	Continuous heavy vs deferred heavy and short duration heavy	0 for continuous heavy vs. short duration, -590 (loss) for continuous heavy vs deferred heavy grazing	(Manley et al., 1995)	all treatments with stocking for heavy grazing had less SOC than

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
						continuous light grazing
-- Outside of North America --						
Australia	5-15	Measurement across adjacent farm paddocks		0	(Sanderman et al., 2015)	
Belgium		CO2 flux by eddy covariance, 1 pasture each system	Rotational vs continuous	0	(Gourlez de la Motte et al., 2018)	
The Netherlands	5	Tame pasture	Continuous vs rotational experiment	-300 (loss) for rotational for 0-30 cm, 0 for 0-60 cm	(Hoogsteen et al., 2020)	Grazing was simulated with vegetation harvest
Switzerland	1	Tame pasture	Flux measurement of rotation grazing only	With rotational grazing and considering CO2, CH4, and N2O, the system was net reduction of global warming potential (net sink in terms of soil C)	(Voglmeier et al., 2020)	No comparison with alternate grazing systems
Argentina	8	Ranch, saline soils	Rotational vs continuous	560 for rotational vs continuous	(Vecchio et al., 2018)	Difficult to estimate precisely from data provided

Increasing Legumes in Pastures and Haylands (land emissions)

A. Method for estimating emissions

Introduction

If there are insufficient legumes in grass-legume pastures and forages, the productivity and protein decline. Increasing legumes refers to the practices of 1) directly planting or overseeding legumes in pastures or forages that have become depleted in legumes, 2) including non-bloating legumes in pasture where bloat concerns caused common legumes of alfalfa and clover from being used in the pasture seed mix, and 3) managing the pasture and forage to extend the longevity and productivity of legumes in the plant stand. Pastures are more likely to be deficient in legumes than forages. In 2011, pastures for beef cattle had an average of 22% legume in the stand compared with 42% in perennial forages (Sheppard *et al.*, 2015). Pyle *et al.* (2018) randomly selected 102 tame pastures in central Alberta and, although 55% of these had been seeded with legumes, these pastures contained few legumes when they were surveyed.

Including more legumes can eliminate the need for fertilizer N on pasture, reducing N2O emission and the embodied GHG emission in fertilizer N.

b. Inclusion in NIR

N application to pastures is estimated and the N2O emission calculated. Emission from N manufacture in Canada is reported under Industrial Processes.

c. Quantities of Emissions

In 2019, 0.317 Mt of fertilizer N was applied on pasture and haylands in Canada. This produced 0.258 Mt CO2e as direct and indirect N2O and 1.01 Mt CO2e for fertilizer N manufacture.

B. Details of Proposed Emission Reduction Measures

a. Proposed reduction measures and mechanisms .

Much of the benefit of legumes is the N they add to the pasture and forage stands from biological N fixation. The addition of mineral N fertilizer to forage and pasture stands with few legumes will increase both productivity and protein production. In 2011, 13 and 19% of perennial pasture and perennial forages received average mineral N applications of 51 and 50 kg/ha, respectively (Sheppard *et al.*, 2015). In that year, coincidently, 13 and 19% of perennial pasture and perennial forage received manure application beyond that deposited during grazing (Sheppard *et al.*, 2015). Some herd managers intentionally do not want to seed legumes are concerned about bloat in cattle that can be caused by traditional alfalfa or clover and so use N fertilizer instead of legumes. Birdsfoot trefoil, sainfoin, or cicer milk-vetch that do not cause bloat are suitable legumes for these pastures (Khatiwada *et al.*, 2020). The mineral N fertilizer has economic costs for the land manager. It causes GHG emissions as N2O and additional embodied GHG emission for N fertilizer manufacture. Therefore, a major benefit of increase legumes in pasture and forage stands with few legumes is by reducing the need for N fertilizer. The manure applications to pasture and forages provide more nutrients than just N so an optimal proportion legumes would not necessarily mean that at least some manure application would not be needed on pasture and forages. Nevertheless, with more legumes, some manure could be applied to other crops instead of fertilizer N.

If pastures or forages are N deficient, then increased legumes in forages and pastures will increase N supply and some of that N will be emitted as N2O. Henderson *et al.* (2015) estimated these additional N2O emission amounted to 0.07 Mg CO2e/ha in the United States. In the case that N from legumes is only just replacing that supplied by mineral N fertilizer, the latter would not cause a source of additional N2O emissions. This was the assumption we used in the analysis.

SOC effects

The possible effects to SOC from increased legumes in pastures and forages come through two mechanisms. The first mechanism is extending the longevity of pasture productivity. The proportion of legumes in legume-grass stands decreases over time due to competition with the grasses and/or winterkill. When the productivity of the pasture and forage performance declines sufficiently, it is typically rejuvenated through termination and reseeding. This termination, especially with tillage, is related to loss of SOC (Issah *et al.*, 2021) and SOC decreases as forage stand duration decreases (Bolinder *et al.*, 2012). Establishing legumes into the pasture can effectively rejuvenate the pasture (Khatiwada *et al.*, 2020) and thereby reduce the need for termination and reseeding to rejuvenate. There is not sufficient data on the exact effect of stand longevity on SOC. In Ireland, other factors, including increasing soil bulk density, affect SOC such that no clear relationship between stand age and SOC.

The second mechanism is from increased SOC by the effect of legumes increasing total stand biomass production and, thereby, increasing input of C to the soil. In their global modeling study of the SOC benefit from adding legumes to pastures, Henderson *et al.* (2015) concluded that pastures in Canada were generally not amenable to SOC increases from increasing legumes in pastures although regions of the US adjacent to BC, Ontario Quebec, and the maritime provinces were amenable (Fargione *et al.*, 2018). Bork *et al.* (2017) found that quality of biomass was improved by having sufficient legumes in legume-grass stands but biomass was not increased from situation of few legumes in the stand.

There is no Canadian data on the effect of adequate legumes on SOC so was not included.

Natural land not included

We did not include natural land used for grazing for increasing legumes do to concerns about disrupting biodiversity.

Enteric Emissions

Mixed legume-grass pastures with adequate proportion of legumes improve cattle nutrition and cattle performance (Bélanger *et al.*, 2017; Peprah *et al.*, 2018) The improved cattle performance reduces GHG per unit of cattle-derived product (milk or meat); initial analysis shows this could reduce the footprint of cattle-derived products by 20% in Saskatchewan (The Cattle Site, 2022). An estimate of this effect is included in this report under Enteric Emissions

Increasing legumes through multiple practices

The most straightforward method is to terminate the pasture or hay and reseed with mix including leguems. This is costly operation and also has the implied cost that production during the establishment year is typically much lower than in the later production years.

To increase legumes in existing stands where the land is suitable, no-till drills can plant legumes directly into the pasture or forage. Overseeding forages and pastures does not plant the legume seeds precisely in the soil but spreads the seed over the surface by broadcasting or passing through livestock gut and depends on action of rugged harrows, rain or snowmelt, or the hoof action of grazing livestock to incorporate the seeds into the soil.

Appropriate management of the grazing such as to allowing existing legumes to set seed and of the forage stand such as to have the legumes in healthy state at freeze-up can increase the longevity of legumes is also part of the strategy to increase grain legumes.

b. Quantification of Emissions

The estimate of N applied to tame pasture was based on the N allocation method described in Yang et al. (2007) as implemented in the NIR (ECCC, 2021). The emission of direct and indirect N_2O were estimated using the Tier 2 method (Rochette *et al.*, 2008) as implemented in Canada's National Inventory Report (ECCC, 2019). However, the perennial crop direct N2O reduction modifier of 0.19 (Liang *et al.*, 2020) was applied. The total N2O emissions from N addition to pasture and forage in Canada was 0.26 Mt CO2e in 2019. Emissions for N fertilizer manufacture were estimated as 3.180 kg CO_2e/kg N (Cheminfo Services, 2016). The total embodied emission for N manufacture was 1.01 Mt CO2e.

Table 1 shows the area, emission, and N use on pasture and hay by province. The actual reduction in emission would depend on the extent that the fertilizer N is reduced.

Table 1. Estimated area, GHG emissions, and N use on pasture and hayland by province

Jurisdiction	Area (kha)	N2O (Mt CO2e)	Fertilizer N (Mt)	N manufacture (Mt CO2e)
AB	2391	0.089	0.117	0.373
BC	278	0.015	0.016	0.051
MB	511	0.021	0.029	0.092
NB	56	0.004	0.003	0.009
NL	5	0.000	0.000	0.001
NS	56	0.004	0.003	0.009
ON	388	0.030	0.022	0.070

PE	42	0.003	0.002	0.007
QC	480	0.038	0.026	0.084
SK	2209	0.054	0.098	0.312
Canada	6416	0.258	0.317	1.008

c. Assumptions

A key assumption is that estimated fertilizer N use on pasture and hayland represents the extent of legume insufficiency on the lands. The legume insufficiency could be greater. Therefore, legumes could increase N additions beyond that from current estimated N fertilizer use.

Another assumption is that addition of legumes avoids all need for fertilizer N. Some fertilizer N is added with other nutrients such as monoammonium-phosphate that is 11% N in the fertilizer product. The price of N fertilizer relative to cost of pasture and forage legumes would also affect the adoption of increasing legumes.

Some hay and pasture are purposely grown not to have legumes such as pure timothy hay for horses. These lands would not be suitable for legumes so would not have any emission reductions from avoiding use of N fertilizer.

d. Current Adoption

Legumes are typically included in hay mixes when initially seeded and in many pastures. Seeding legumes into existing stands is also done. The analysis assumes that N fertilizer use is an indicator of the effectiveness of those practices so that estimating current adoption is not essential to estimate the incremental mitigation.

e. Barriers to Adoptions

There are significant barriers to increasing legumes in pasture and hayland.

Terminating and reseeding the entire stand with a seed mix with legumes is expensive and the yield of forage during the establishment year for the newly seeded stand is typically much lower than later production years.

A major barrier to adoption is the cost of seeding legumes into existing stands considering there is also a risk that the seeded legume will not establish successfully and/or not persist long.

Another barrier is that the management of the pasture may need to be changed, such as to rotational grazing, to enhance the persistence of legumes (see the Rotational grazing section for barriers for that adoption). The management of hay may also need to be changed from past management to improve legume persistence. This could decrease the yield of hay.

In comparison to these barriers, applying N fertilizer can appear to be a more attractive management for insufficient legumes in the stand.

C. Changes required for the NIR

Better knowledge of the stand management practices would enable better estimate the amount of N fertilizer used on pastures and hayland. Otherwise, there is no changes required to NIR methods.

D. Co-benefits

The main cobenefit is reduced N leaching and N contamination of runoff from applied N fertilizer application.

The legumes add to biodiversity by creating a more attractive habitat for many animals.

E. Limitations and opportunities

The success of establishing legumes into existing stands needs to be better understood to improve the analysis of this mitigation pathway.

The programming could consider overall improved management of pastures and haylands. This would include rotational grazing. The improved management of these lands would be best if part of programing for management of enteric emissions from ruminant livestock.

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The Cattle Site, 2022. Reducing beef cattle GHG emissions

A. Methods for estimating emissions associated with wetland restoration and avoided conversion

a. Description of emissions source and mechanisms

Humans have been draining/converting wetlands for centuries if not millennia and this degradation continues, driven largely due to population growth and increasing economic pressure (Davidson 2014). In fact, according to the Global Wetland Outlook (Ramsar Convention, 2018), wetlands are our most threatened ecosystems, disappearing at a rate 3 times faster than forests, with 35% of global wetland area lost since 1970. In Canada, wetlands loss since the pre-settlement 1800s has been estimated at 20 million hectares nationally (Government of Canada, 1991). While there is uncertainty associated with the degree of wetland loss in the southern agricultural landscapes of Canada, it is generally accepted that between 40-70 % of historical freshwater mineral wetlands have been converted to other land uses (Bartzen et al., 2010; Dahl and Watmough 2007). This scale of wetland loss can have large implications for regional carbon cycling. For example, Byun et al., (2018) found that since European settlement roughly only 11% of historical marsh extent remains in southern Ontario and this has resulted in substantial reduction in soil organic carbon stocks of up to 1.58 Tg. Additionally, despite increasing recognition of the multiple ecosystem services provided by wetlands and their value to society (Asare et al., 2022; Davidson et al., 2019; Creed et al., 2017; de Groot et al., 2012), these vital ecosystems continue to be lost at an alarming rate in Canadian agricultural landscapes (Watmough and Schmoll 2007; Watmough et al., 2017).

There is growing interest in applying nature-based climate solutions (NbCS) (Cohen-Shacham et al., 2016), including the restoration and/or avoided conversion of intact landscapes, to help countries achieve augment greenhouse gas reduction targets. A previous study concluded that implementation of cost effective NbCS - based on conservation, restoration, and improved land management actions that increase carbon storage and/or avoid greenhouse gas emissions from forests, wetlands, grasslands and agricultural landscapes - could help meet up to one-third of the global greenhouse gas reductions required by 2030 to keep warming below 2 C (Griscom et al., 2017). Similarly, Drever et al., (2021) found that Canada could meet approximately 35% of its 2030 target, 78.2 Tg CO_2e/yr, through NbCS, and that avoided conversion of natural habitats like grasslands and wetlands were among the most effective NbCS actions. This study also found that restoration of freshwater mineral soil wetlands could reduce greenhouse gas emissions by 0.4 Tg CO_2e/yr by 2030. While this is not a substantial contribution with regards to the total greenhouse gas emissions reductions required nationally, it could be quite significant for helping the agricultural sector and producers meet sector specific greenhouse reduction targets.

Because wetlands are among the most productive and carbon dense habitats, their conversion can release substantial amounts of carbon back to the atmosphere. For example, over the last 60 years it is estimated that more than 665,000 ha of wetlands have been lost in the agricultural landscapes of the Canadian Prairies and southern Ontario resulting in a reduction in soil organic carbon stocks of approximately 59M tonnes. As a result, the ongoing conversion of these habitats has the potential to offset many of the GHG reductions that have been achieved through agricultural BMPs such as conservation tillage and nutrient management.

b. How does the NIR currently measure and report on wetlands

Currently, the Wetlands category within Canada's NIR includes emissions from peatlands managed for peat extraction and from flooded lands (hydroelectric reservoirs) (ECCC 2021). Trends in this category are mainly driven by the creation of large reservoirs before 1990. However, the NIR does not track GHG emissions or reductions associated with the conversion or restoration of freshwater mineral wetland habitats or for unmanaged peatlands in Canada. Moving forward is it important to address this issue as the National Inventory Report must adhere to the principle of completeness so there are both demands and actions to include these emissions before 2030. This will be particularly important for agricultural landscapes where the bulk of wetland conversion and restoration takes place.

B. Details of proposed emission-reduction measures

For this analysis we specifically consider GHG emissions reductions associated with the avoided conversion and restoration of freshwater mineral soil wetlands as this is the most abundant wetland class within agricultural landscapes based both on area and frequency. The basis of our analysis is derived largely from Drever et al., (2021) with the exception of the following modifications. Firstly, the analysis by Drever et al., (2021) focused on maximum reduction potentials. For this analysis we have scaled the area of impact/scale of program delivery to what we believe is achievable under the time frame of APF3. Secondly, to align with reporting within Canada's NIR we used GWP100 when converting methane emissions to CO2eq, as opposed to the sustained global warming potential (SGWP) that was used in the analysis by Drever et al., (2021). Thirdly, because there is often significant above ground woody biomass that is lost with the conversion of intact wetlands, we have added a calculation to estimate the impact on emissions. Lastly, we updated the GHG emissions associated with cropland based on the summary of cropland emissions presented in Burton et al., (2021).

a. Avoided wetland conversion

GHG reductions for avoided conversion are calculated using the following information. Estimated stock change of 89 (±41.8) Mg C ha^{-1} (Badiou et al., 2011). This value is almost identical to the value presented by Bedard-Haughn (2006) and within the range of those reported by Euliss et al., (2006; 10.1 tonnes/ha) and Nahlik and Fennessy (2016; 150 tonnes/ha). We project stock to decrease linearly over 20 years for an annual emissions reduction of 16.3 tonnes CO2eq/ha/yr. From this values we deduct ongoing CH_4 emissions associated with wetlands using the default emission factor from IPCC (2014) for natural wetlands ((136 kg CH4/ha/yr ± 83 [95CI];) = 3.808 tonnes CO2e/ha ± 2.324 [95CI] based on GWP100). As previously mentioned, we opted to use GWP100 for this analysis to remain consistent with the methodology applied in the NIR.

In order to account for the loss of carbon associated with woody biomass when wetlands are converted, we assumed that 7% of avoided wetland area would have woody biomass such as willows and/or poplar species. This is estimated based on doubling the proportion of prairie parkland that is forested (3.5%; Huffman et al., 2015), knowing that remaining forest stands in this region are often associated with wetland habitats. This is likely a conservative estimate based on the higher proportion of woody biomass present in other agricultural landscapes of Canada such as the Mixed Wood Plains and Niagara Region agricultural landscapes. The carbon stock and associated GHG reduction potential was based on willow biomass associated with wetlands in SK from Mirck and Schroeder (2013) where the mid-range of dry weight was estimated to be 10 t/ha. This is converted to carbon using a carbon fraction of dry matter of 0.5 (Huffman et al., 2015).

b. Potential reductions achieved through avoided wetland conversion

For our analysis, avoided wetland conversion program delivery was set at 15,000 ha yr^{-1} between 2023-and 2028 which is expected to offset current annual loss rates and result in a total enrolment of 90,000 wetland hectares over the course of the program. Such a program would result in an annual GHG reduction of 1.17M tonnes CO2eq by 2028, with a cumulative reduction of 4.15M tonnes CO2eq between 2023 and 2028. The cost of delivering avoided wetlands conversion through CEs in the Canadian Prairies is approximately $1500/ha. As a result, the cost to implement this strategy would be $129.5M or $21.6M/yr. Based on these cost estimates GHG emissions reduction would be delivered at a cost of approximately $31 per tonne of CO2eq. While the total program cost might seem large, it should be noted that these costs would be shared and that any investment from the federal government would allow conservation organizations to leverage significant contributions through other sources.

c. Wetland restoration

GHG reductions for wetland restoration were calculated using the following information. We estimate increases in SOC based on accumulation of SOC to previous stock levels over 40 years (2.225 tonnes of C/ha/yr or 8.15685 tonnes of CO2eq/ha/yr – based on estimated stock change of 89 tonnes of C/ha). Similarly, Euliss et al., (2006) estimated a SOC increase of 3.05 tonnes /ha/yr in restored semi-permanent wetlands for the Prairie Pothole Region of the US. As previously discussed, our stock change estimates associated with wetland conversion, which drive the sequestration potential of restoration are similar to those reported by Bedard-Haughn et al., (2006) and lower than those for freshwater mineral soil wetlands of the conterminous US (Nahlik and Fenessy, 2016). We deduct methane emissions for restored wetlands based on the IPCCs (IPPC 2014) default emission factor for created or rewetted wetlands (153 kg CH4/ha/yr ± 148 [95CI] = 4.284 tonnes CO2e/ha ± 4.144 [95CI] based on GWP100). Unlike our estimates for avoided wetland conversion we did not account for the potential development of woody biomass around restored wetlands as this has not been widely assessed and tree planting is not part of current wetland restoration activities. Avoided cropland emissions were factored using these same estimates presented for the avoided conversion of wetlands.

d. Potential reductions achieved through wetland restoration

Wetland restoration program delivery was set at 1,000 ha yr^{-1} between 2023-and 2028 for our scenario which would roughly double the current habitat objective set by the PHJV and represents the bulk of restoration activities in Canada largest contiguous agricultural landscape. Such a program would result in an annual GHG reduction of 0.026M tonnes CO2eq by 2028, with a cumulative reduction of 0.090M tonnes CO2eq between 2023 and 2028. The cost of delivering wetland restoration is significantly higher relative to the cost of avoided conversion via conservation easements. In the Canadian Prairies the cost to restore wetlands is approximately $5,200/ha. However, this is substantially lower relative to wetland restoration costs in southern Ontario which are estimated at approximately $31,000/ha. At 1,000 ha/yr (from 2023 to 2028) to total project cost would be $31.2M or $5.2M/yr. Based on these cost estimates GHG emissions reduction would be delivered at a cost of approximately $200 per tonne of CO2eq (in 2028) in the Prairies and $1,192 per tonne in southern Ontario (in 2028).

Given the significant differences in the return on investment in terms of carbon offsets between avoided conversion of wetlands and restored wetlands, ensuring adherence to the mitigation sequence of avoid, minimize, compensate should be central to any program that seeks to achieve GHG reductions via wetlands as nature-based solutions. This is an important consideration given the current preference for restoration activities, even though ecosystem conservation and management are more efficient providers of natural climate solutions (Cook-Patton et al., 2021).

C. Co-benefits associated with wetland programming

Wetland conservation and restoration provide many other co-benefits that more traditional carbon offset methodologies do not. Additionally, some of the co-benefits associated with wetland restoration and conservation will help mitigate some of the impacts predicted to occur as a result of climate change. For example, droughts and flooding are both predicted to worsen in the future as a result of anthropogenic climate change. Wetlands, have the potential to moderate extreme drought and flood conditions and can help keep water on the land when it is scarce by sustaining base flows in stream. Conversely, under extreme snowmelt and precipitation conditions wetlands can store water and reduce the speed at which water is transmitted through a watershed, thereby desynchronizing flood waters, resulting in reduced flow volumes and flood peaks.

As a result of their importance in regulating flows at a watershed scale, wetlands are equally important in their role of nutrient sequestration and mitigation of non-point source pollution. This is important as results from a long-running shallow lake mesocosm experiment suggest that nutrient loading can be an important driver of total and individual GHG flux (Davidson et al., 2015). Other studies have noted the link between trophic state of aquatic ecosystems and increased methane emissions, for example constructed wetlands for stormwater management dominated by emergent and submerged vegetation had significantly lower GHG emissions relative to conventional stormwater basins that were mostly open-water and often plagued by blue-green algal blooms (Badiou et al., 2019). Based on these findings, nutrient sequestration in wetlands where submerged macrophytes are typically abundant should help reduce GHG emissions. This is achieved by preventing nutrient loading of shallow lakes in agricultural environments where eutrophication could potentially stimulate GHG emissions. This is particularly important as it has been suggested that global warming and eutrophication in freshwaters may mutually reinforce the symptoms they express and thus the problems they cause (Moss et al., 2011). The value of water quality benefits associated with wetland conservation and restoration are substantial. For example, a recent study by Aziz et al., (2021) estimated that the value of P removal by all wetlands in southern Ontario amounts to $4.2B/yr, and that it would cost significantly more to offset the P removal function of these wetlands via traditional BMPs and/or waste water treatment plant upgrades.

While wetlands play an important role in earth's climate as a result of their influence on the global carbon cycle and GHG emissions, they also have the ability to regulate local and regional climates by their presence on the landscape. For example, inland waters such as lakes and wetlands are important landscape features in many parts of North America and can comprise significant portions of regional landscapes such as in the Great Lakes region, the Boreal region, and the Prairie Pothole Region. Inland waters of these regions likely play an important role in regional climate as a result of large differences in albedo, heat capacity, roughness, and energy exchange compared to more terrestrial systems (Bonan 1995). A study by Krinner (2003) found that inundated land surface (wetlands and lakes) in the boreal region had strong impacts on climate simulations, particularly in the summer. Furthermore, this research found that wetlands played a more important role relative to lakes in cooling the boreal region in summer and humidifying the atmosphere. Recently, there has been a number of studies that have demonstrated the importance of wetlands in regional climate and that have documented negative correlations between wetland abundance and regional air temperatures (Yunlong et al. 2011, Bai et al. 2013, Yan et al. 2015). Conversely, both Yunlong et al. 2011 and Yan et al. 2015 demonstrated that wetland conversion reversed the cooling/humidifying effect of wetlands on regional climate and resulted in increased maximum and minimum air temperatures. More recently, Zhang et al., (2022) demonstrated the importance of small wetlands in the regional climate of the Canadian prairies and found that the presence of wetlands can reduced summer temperatures between 1 to 3C and significantly reduce the number of hot days during heat waves.

As previously mentioned, a number of studies have found that wetland restoration and creation can result in significant rates of net carbon sequestration. However, the radiative switchover time associated with such projects can be quite long due to the GHG emissions and differences in the fate of CO_2 and CH_4 in the atmosphere. While wetland restoration and/or creation of temperate FWMS wetlands might require upwards of a century to become net radiative sinks based on carbon sequestration and GHG emissions they will have an immediate impact on regional climates through their ability to cool and humidify the atmosphere. This should help alleviate some of the immediate impacts that climate change is having regionally, thereby potentially offsetting the increased radiative forcing associated with wetland restorations prior to becoming net radiative sinks.

D. Limitations and opportunities beyond current scope (incl. linkages to other programs/measures and possible future scale up)

Unfortunately, Canada does not currently have a complete national wetland inventory at a resolution that captures the numerous small wetlands that characterize the southern agricultural regions of Canada. Furthermore, aside from the periodic reporting conducted by the CWS on habitat change there is no effective mechanism in place to quantify annual loss rates for these systems. Furthermore, we do not currently have tools available at a national scale to identify potentially restorable wetlands.

The rapid development of remote sensing technology will likely facilitate quantifying trends in loss and recovery of wetland habitat within the next decade. Merging this information with data derived from precision agriculture should be explored to target wetland conservation and restoration to marginal lands where GHG reduction benefits can be maximized while minimizing economic impacts to agricultural producers and potentially enhancing returns through a combination of incentives tied to carbon offsets and by reducing input costs.

Lastly, there is emerging science that suggests that wetland management can help reduce GHG emissions from intact and restored wetlands. Preliminary research results from surveys of pothole wetlands in the Canadian Prairies have found significantly lower GHG emissions for wetlands embedded in perennial cover relative to those in cropland (DUC, unpublished data), particularly for methane. Additionally, with information regarding the key drivers of methane emissions from wetlands it may be possible to target wetland restorations in places that minimize methane emissions. Furthermore, there are other potential interventions, such as soil conditioners, that could be explored as a mechanism to reduce methane emissions from restored wetlands in order to maximize GHG reductions through wetland restoration. Lastly, in order to understand the full extent of wetland conversion of GHG emissions it will be important to assess the emissions associated with the extensive drainage features that have been excavated to facilitate the removal of wetlands from the agricultural landscape. To date this has not been attempted for freshwater mineral soil wetlands to our knowledge.

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