A Review of the Efficacy and Cost-Effectiveness of On-Farm BMPs for Mitigating Soil-Related GHG Emissions

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Executive Summary

The greenhouse gas (GHG) sources and carbon (C) sinks of Ontario's agricultural soils and the impacts of management practices and potential strategies to reduce emissions and increase sinks are not well-quantified. In addition, there is a need to determine the economics behind different practices to better inform future program design. The objective of this review was to provide a synthesis of the science on the efficacy and cost-effectiveness of best management practices (BMPs) at mitigating GHG emissions and increasing sinks from soils, landscapes, climates and production systems relevant to Ontario.

A baseline of nitrous oxide (N_2O) emissions from Ontario agricultural land was calculated from the 1990-2016 Census of Agriculture data. The total N_2O in 2016 was estimated at 17.2 Gg N_2O/y or 2.6 kg N_2O-N/ha. Synthetic fertilizer nitrogen (N) and residue-N constituted the major contributors to direct N_2O emissions. Corn crops were one of the major contributors to N_2O emissions because of the relatively large acreage devoted to corn production and high N-fertilization rates used.

The mitigation potential from BMPs on GHGs was assessed quantitatively through the compilation of data ranges from published experimental findings and from meta-analyses. These were then assessed qualitatively by evaluating the consistency of mitigation findings and scrutinizing the data and conditions.

BMPs that showed promise based on the reduction of GHGs or the increase of soil organic carbon (SOC) were identified as (in no specific order):

- Matching of N rate to crop needs
- Use of nitrification inhibitor (NI) or nitrification plus urease inhibitors (NI+UI)
- Cover crops (CCs)
- Afforestation
- Biomass crop
- Conservation tillage

In addition to the potential for GHG mitigation, the on-farm financial returns generated through the implementation of the above BMPs were assessed based through a literature review. In general, farms for which these BMPs are profitable have adopted them. However, extent of the returns (or losses) to investment for these practices depends on site characteristics and year.

Matching N rate to crop needs usually results in a reduction from the conventional rate. That said, for a selected N rate, aspects such as yield response, price, and risk still need to be addressed. N rate reduction is especially important as N fertilization beyond the optimal level for a given crop results in an increase in N_2O emissions. The resulting GHG levels from the selected N rate depend on interactions between the N rate and application time, method, and/or type of fertilizer. These interactions are complex and site-specific and further complicated by risk factors such as weather. The rate of N can be reduced through side-dressing. This allows N rates to be adjusted based on crop appearance part way through the season. Technologies to find the site-specific optimal level of N exist but currently have high implementation costs that outweigh the potential benefits for most fields unless those fields exhibit a great deal of variability.

Inhibitors, especially NI and NI+UI, have been shown to consistently reduce N_2O emissions and lower N leaching and volatilization losses (depending on condition and use), the latter constituting an indirect N_2O emission. The few studies on the economics of NI and UI use have shown them to be potentially cost-effective means for GHG mitigation.

CCs were selected because they increase SOC, reduce indirect N_2O emissions, and recycle N in the soil thus providing a means to potentially reduce fertilizer use. There are general soil health benefits to CCs that are beyond the scope of this review but nonetheless provide an added incentive for their use. These increases in soil health from CCs and complexity in crop rotation are associated with an increase in on-farm profits over time.

Afforestation, biomass crops, and short-rotation crops that are harvested for biomass were also found to be promising because of their C sequestration potential which is shown repeatedly in the literature. Their benefits come from a combination of large rooting systems, reduction of tillage and soil management operations, and buildup of soil C. The on-farm returns from afforestation and inter-cropping depend critically on a sufficiently high price for C and/or biomass.

The use of conservation tillage (no-till (NT) or reduced tillage) has been shown to improve SOC and in many cases to reduce N_2O emissions. This is especially true with reduced tillage. The yield reduction that is usually associated with NT in humid climates was found to be either eliminated or lowered when reduced tillage such as zone-tillage is used instead of NT. While conservation tillage can reduce labour and fuel costs, the potential decrease in average yield and the increase in its variability affect the likelihood of adoption. The profitability of conservation tillage depends on the soil type and crop rotation of the individual farmer.

Other BMPs with the potential for GHG mitigation were not reviewed here because there was either not enough data/literature or the data was inconsistent in demonstrating their benefits.

Contents

Abbreviations

4R	Right source at the Right rate, at the Right time and the Right place
AA	Anhydrous Ammonia
AAFC	Agriculture and Agri-food Canada
ACR	American Carbon Registry
AFOLU	Agriculture, Forestry and Other Land Use
AGGP	Agricultural Greenhouse Gases Program
AN	Ammonium Nitrate
ARB	Air Resources Board of the California Environmental Protection Agency
ASN	Ammonium sulphate nitrate
BMP	Best Management Practice
BT	Biennial Tillage (with a chisel plow)
C	Carbon
CC	Cover Crop
CCAP	Climate Change Action Plan
CCX	Chicago Climate Exchange
CDC	Centers for Disease Control and Prevention (United States)
CI	Confidence Interval
CAR	Climate Action Reserve
CO_{2e}	CO_2 equivalents
CRF	Controlled Release Fertilizer
CSHA	Cornell Soil Health Assessment
CHT	Tillage from a chisel plow
CT	Conventional Tillage
DCD	dicyandiamide (nitrification inhibitor)
DNDC	DeNitrification-DeComposition
DMPP	3,4-dimethylpyrazol phosphate (nitrification inhibitor)
EQIP	Environmental Quality Incentives Program
EF	Emission Factor (%)
ERT	Emissions Reduction Ton

ESN	Environmentally Smart Fertilizer
ESRD	Environment and Sustainable Resource Development (Alberta)
EWUE	Ecosystem Water Use Efficiency
FCC	Fall Dominated N application Cover Crop
FIE	Fertilizer Induced Emission (%)
FORCARB-ON	FORest CARBon Budget Model adapted for Ontario
GHG	Greenhouse Gas
GWP	Global Warming Potential
HP	Hybrid Polar
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
LRR	Land Resource Regions (part of the USDA)
LSM	Liquid Swine Manure
LUCI	Land Use Carbon Inventory
LULUCF	Land Use, Land-use Change and Forestry
MSU	Michigan State University
MSU-EPRI	MSU-Electric Power Research Institute
N	Nitrogen
NASS/ERS	National Agricultural Statistics Survey/ Economic Research
NBPT	N-(N-butyl)-thiophosphoric triamide (urease inhibitor)
NDVI	Normalized Difference Vegetation Index
NECB	Net Ecosystem C Balance
NI	Nitrification Inhibitor
NIR	National Inventory Report
NT	No-Till
OM	Organic Matter
OMAFRA	Ontario Ministry of Agriculture, Food and Rural Affairs
OSHA	Ontario Soil Health Assessment
OSR	Oil Seed Radish
P/PE	Ratio of Precipitation to Potential Evapotranspiration
PCU	Polymer-coated Urea

PRP	Paddock, Range and Pasture
PSNT	Pre-Sidedress Nitrogen Test
PT	Tillage from a moldboard plow
QA/QC	Quality Assurance/ Quality Control
SCC	Spring Dominated N application Cover Crop
SFN	Synthetic Fertilizer N
SOC	Soil Organic Carbon
SRC	Short rotation coppicing or crop
UAN	Urea Ammonium Nitrate
UI	Urease Inhibitor
UNFCCC	United Nations Framework Convention on Climate Change
Urea+NI+UI	Urea treated with a urease inhibitor and a nitrification inhibitor
USDA	United States Department of Agriculture
V4	Descriptor of corn plant vegetative development where four leaves are present
V6	Descriptor of corn plant vegetative development where six leaves are present
V8	Descriptor of corn plant vegetative development where eight leaves are present
V14	Descriptor of corn plant vegetative development where 14 leaves are present
VCS	Verified Carbon Standard
VRF	Variable Rate Fertilization
VVB	Verified Verification Body
WSG	Warm-Season Grasses
ZT	Zone Tillage

1. Introduction

In Canada, the agricultural sector produces 8% of the total GHG (N_2O and CH_4) emissions in the country making it the second largest emitting sector after energy (Environment and Climate Change Canada, 2017). The contribution of the agricultural sector to Ontario's total GHG emissions is about 5.8%. The agricultural sector and Land Use, Land-use Change and Forestry (LULUCF) are currently net sinks of C (CO_2 emissions and SOC storage). Whether agricultural land is a net sink or a source of GHGs depends on management practices and land use decisions. The relationship between changes in the input and output decisions of farmers and the trends in net GHG levels since 1990 are highlighted in the National Inventory Report. However, the impact of BMPs is not fixed but rather dynamic reflecting the nature of soil processes as well as biotic and abiotic factors. The efficient design of policies to encourage farmers to adopt BMPs to reduce GHG emissions requires an understanding of the complex physical relationship between farming practises and GHG mitigation along with the economic impacts associated with the implementation of those practices.

The objective of this report is to review the literature and conduct a synthesis of science/knowledge of GHG emissions associated with agricultural management practices in Ontario. The synthesis includes an economic assessment of these practices. A review of the efficacy of different relevant GHG mitigation program policies is also conducted. In addition, this synthesis provides information on GHG mitigation options of BMPs specific to Ontario crop production systems. Qualitatively, it is known that certain practices have GHG benefits, but not all of the practices have been extensively and systematically measured. This literature review includes studies from regions that share similar climate and production systems to Ontario (i.e. mainly eastern Canada, and the northeastern U.S. regions). A geographically broader scope of studies was included in some cases where the literature availability was limited. An additional outcome of this synthesis is the identification of knowledge gaps and future research needs.

The report begins with a county-level baseline of Ontario N_2O emissions from agricultural soils calculated from the 1991-2016 Census of Agriculture data. Direct (N_2O from soil application of N and from the decomposition of organic matter) and indirect (N_2O from leaching and ammonia volatilization) emissions are estimated based on the emission factors (EF) as used by the National Inventory Report (NIR) based on Rochette et al. (2008a) methodology. This baseline can be used to calculate province-wide N_2O mitigation potential of specific BMPs.

The means to reduce these baseline levels are reviewed in section 3. After an initial broad review of the literature, a number of BMPs with available data were identified as having potential in terms of efficacy and adaptability to the region. GHG reduction ranges were collected and these ranges acted as qualitative confidence limits for each BMP. The promising BMPs were also evaluated either quantitatively (if possible) or qualitatively (if not) in terms of their on-farm profitability (section 4). Next, where experimental field data from eastern Canada and similar climatic regions was available, a meta-analysis was conducted to provide statistically-assessed

GHG reduction means with 95% confidence intervals (CI). It is important to note, that although the means were statistically assessed, the quality of the results is restricted by available data. For this reason, the quantitatively produced means were further compared qualitatively with the ranges from the literature and evaluated based on scientific knowledge of soil processes i.e. a mean mitigation potential of a BMP that statistically seems to be a good BMP might not rank high on the list because it cannot be verified after scientific scrutiny.

GHG emissions from soils are mitigated through: (1) a reduction of N_2O, (2) a reduction of CO_2, (3) a reduction CH_4 from soil, and (4) an increase in C sequestration in soil and long-term biomass. Within crop agriculture, the main processes that serve to mitigate GHG levels are reduction in the emissions of N_2O and increase in C sequestration. Consequently, these are the main effects assessed in this report. Indirect emissions from the leaching of nitrate and the volatilization of ammonia were also included in the assessment. Soils in upland agriculture (not submerged) act as a CH_4 sink and are rarely a CH_4 source in Ontario soils.

This report is intended to provide information for future program designs focused on soil GHG mitigation through adoption of on-farm BMPs. The synthesis will determine the initial focus to act as a starting point on developing suitable coefficients and/or estimates of BMP GHG mitigation potentials in different soils and landscapes for various and specific production systems.

2. Baseline Agricultural Soil N_2O Emissions in Ontario 1990-2016

The National Inventory Report by Environment and Climate Change Canada (ECCC) (2017) reports that agriculture accounts for 5.8% of Ontario's total GHG emissions. In particular, this sector is a large emitter of N_2O, a gas that is 298 times stronger than CO_2 in terms of its global warming potential (IPCC, 2007). In 2015, 65% (16.9 kilotonne N_2O) of Ontario's N_2O emissions (26 kilotonne N_2O) were generated by agricultural activities (ECCC 2017). The main agricultural sources of N_2O are agricultural soils, manure management, and burning of residue (ECCC 2017). This chapter reports only on N_2O emissions which fall under the Agriculture GHG category in the NIR. This category does not include C sequestration or CO_2 emission except for the CO_2 that is generated from liming, urea application and other carbon-containing fertilizers. Soil C change and CO_2 emission or uptake from cropland and forestry and grasslands, which are reported under the LULUCF in the NIR are not reported in this chapter. It is worth mentioning that Ontario, under the Climate Change Action Plan, is in the process of creating a Land Use Carbon Inventory (LUCI) for the estimation of GHGs emissions from agriculture, forestry and other land uses (Climate Change Action Plan, 2017).

While having an inventory of N_2O emissions at the provincial scale (such as the NIR) is undoubtedly useful for GHG mitigation efforts, having an estimate of emissions at the county level can make those efforts more targeted to the areas that produce higher rates of emissions. For this reason, Jayasundara and Wagner-Riddle (2010) developed a county-scale inventory of N_2O and methane emissions from Ontario's agricultural soils and livestock production systems for the years 1990 to 2007. The purpose of the current study was to update this inventory focusing on N_2O emissions from agricultural soils. This report presents the results of the updated inventory and discusses the trends and changes that have occurred since 1990. The updated emissions will serve as baseline for potential emission reduction scenarios that are presented in section 3.

2.1. Methodology

To conduct the county-scale inventory of N_2O and methane emissions from Ontario agriculture, Jayasundara and Wagner-Riddle (2010) used data from the Census of Agriculture. Statistics Canada conducts the census every five years, so census data was available for the years 1991, 1996, 2001, and 2006. Linear interpolation was used to estimate data in the inter-census years. The current report used the 2011 and 2016 censuses to update the inventory for those two years (Statistics Canada, 2017). Due to time restrictions, data was not estimated for the inter-census years, although this can be done in future studies.

Following the approach of Jayasundara and Wagner-Riddle (2010), this study applied the methodology of Rochette et al. (2008a) to estimate N_2O emissions from agricultural soils. The N_2O emission estimation methods developed by Rochette et al. (2008a) are used by ECCC in the NIR, but were adjusted here for county-scale calculation. The estimated N_2O emissions were multiplied by a factor of 298 to obtain CO_2-equivalent emissions.

Full details of the methodology and assumptions made for this report are given in the appendix.

2.2. Results and discussion

In 1990 and 2016, total N_2O emissions from agricultural soils in Ontario were 16.15 Gg yr^{-1} and 17.17 Gg yr^{-1}, respectively. This represents an increase of 6.3% in total N_2O emissions between 1990 and 2016. Scaled by area of cultivated land, estimated N_2O emissions from soils were 2.27 kg N_2O-N ha^{-1} (1.06 Mg CO_{2e} ha^{-1}) in 1990 and 2.61 kg N_2O-N ha^{-1} (1.22 Mg CO_{2e} ha^{-1}) in 2016. In terms of area-scaled emissions, there has been an increase in N_2O emissions of 15% between 1990 and 2016. This percentage increase is greater than the 6.3% increase in total emissions. A reason for this is that the area of cultivated land decreased by 7.5% from 1990 to 2016. Thus, although total N_2O emissions from agricultural soils in Ontario have increased by 6.3%, the emissions per hectare of cultivated land have increase by more than that, i.e. 15%.

Figure 2.1 illustrates the total N_2O emissions from agricultural soils in Ontario since 1990, broken down into direct and indirect emissions. Direct N_2O emissions result from activities such as N fertilizer application, manure application, residue, and irrigation, while indirect emissions result from ammonia volatilization and leaching. The breakdown of the direct emissions is presented in Figure 2.2. As seen from this figure, synthetic fertilizer N (SFN), manure N, and residue N comprise the majority of direct N_2O emissions, while tillage, summer fallow, irrigation, histosols (organic soils), and paddock, range and pasture (PRP) contribute a small fraction.

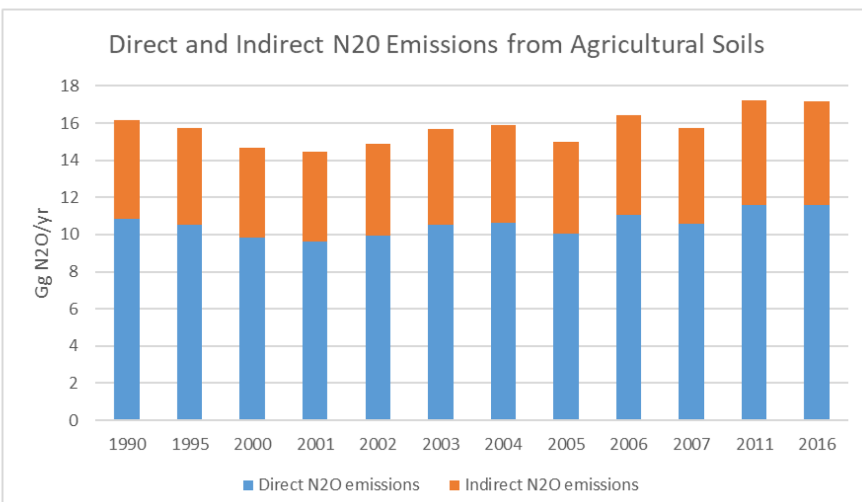

Figure 2.1. Direct and indirect N_2O emissions from agricultural soils in Ontario.

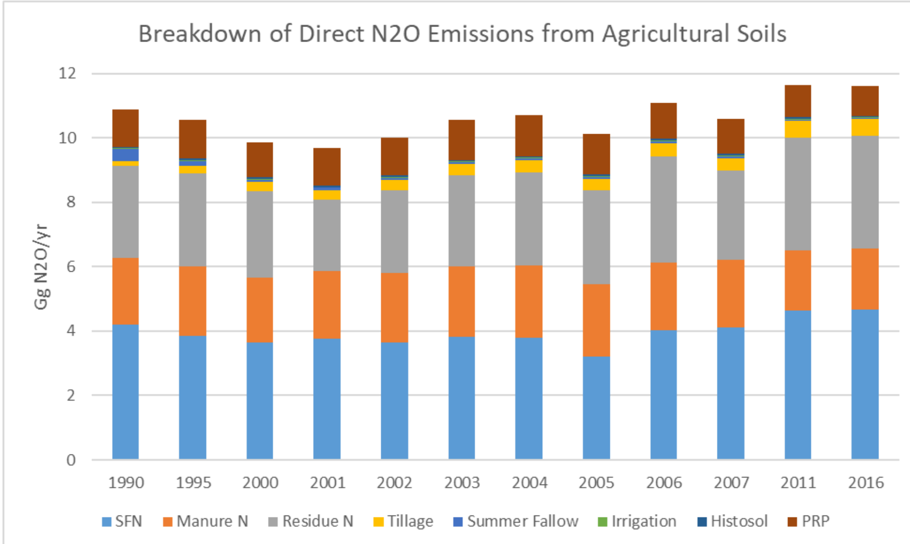

Figure 2.2. Breakdown of direct N_2O emissions from agricultural soils in Ontario.

The county level direct and indirect emissions in 1990 and 2016 and the breakdown of direct emissions for 1990 are given in Figures 2A, 3A, and 4A, respectively, in the appendix. Figure 2.3 shows the breakdown of the direct emissions by source in 2016. The counties are in order of the five Ontario regions, starting with Southern Ontario counties on the left, followed by Western Ontario, Central Ontario, Eastern Ontario, and Northern Ontario on the far right. SFN and residue are the major sources of N_2O emissions in most counties. Manure N is also a large contributor of emissions, although of varying degrees across different counties.

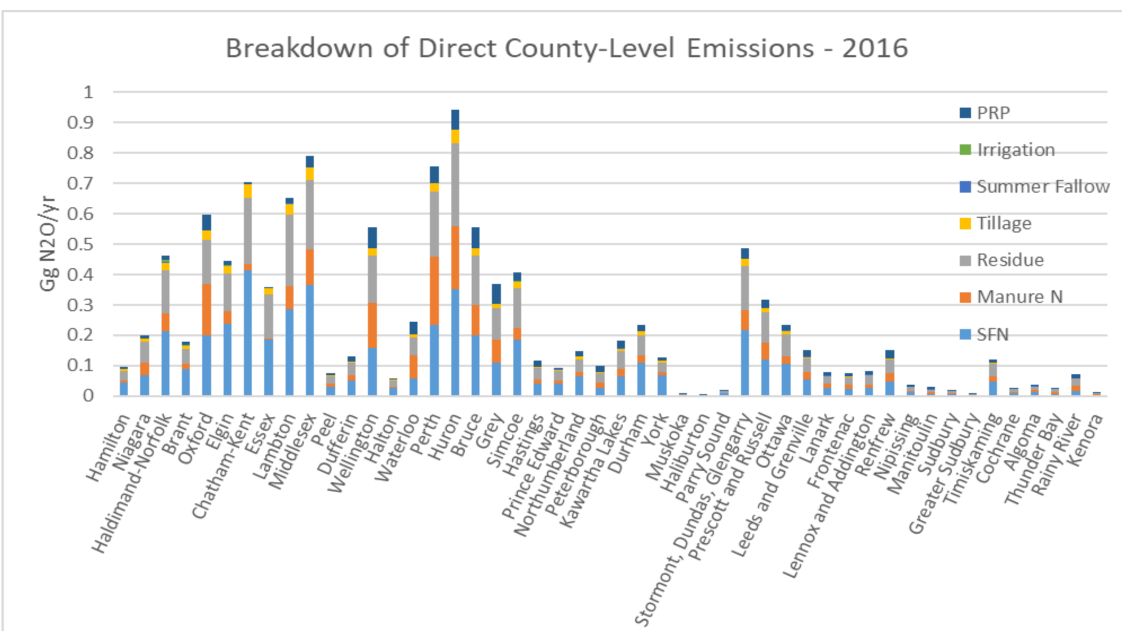

Figure 2.3. Breakdown of direct N_2O emissions per county in Ontario in 2016.

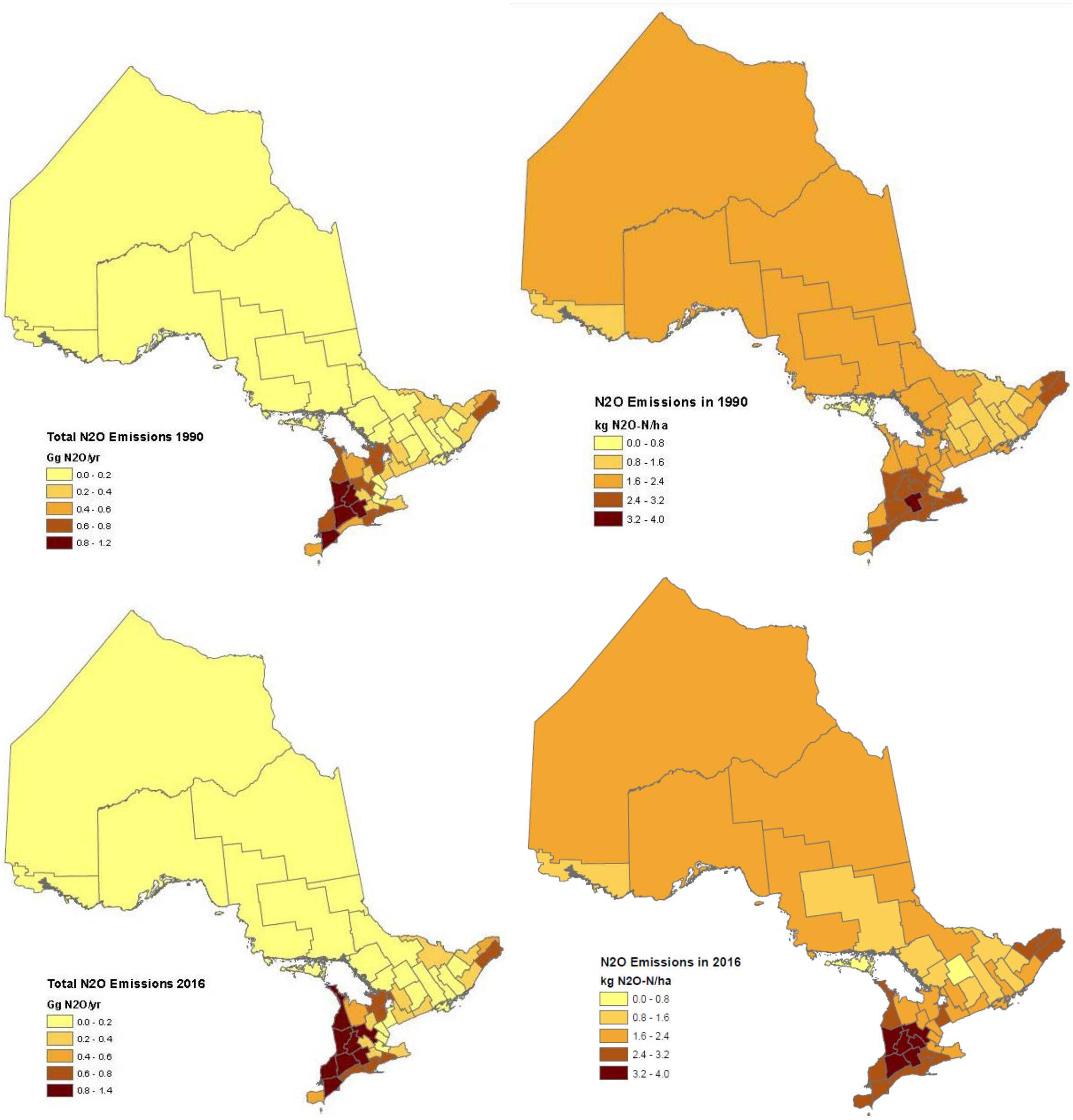

Figure 2.4. Total N_2O emissions from agricultural soils in Ontario (Gg N_2O yr^{-1}) and N_2O emissions per hectare of cultivated land (kg N_2O-N ha^{-1}) in 1990 and 2016.

The change in total N_2O emissions as well as in emissions scaled by area of cultivated land in Ontario counties between 1990 and 2016 is shown in Figure 2.4. The top two maps are for 1990 and the bottom two are for 2016. The maps on the left illustrate the total N_2O emissions per county in 1990 and 2016, and the maps on the right illustrate the N_2O emissions per hectare of cultivated land. The colours on the maps correlate to the level of N_2O emissions in each county with yellow being low and the darkest orange being high. A comparison between the 1990 and 2016 maps reveals which counties have seen a decrease in N_2O emissions over the past 26 years and which ones have experienced an increase.

The trends in major field crops since 1990 are given in Figure 1A in the appendix. Corn, the second largest crop in area, is one of the major drivers of N_2O emissions from Ontario agriculture due to the high amounts of N used to fertilize corn. As such, trends in corn production may be linked with trends in N_2O emissions. The map in Figure 2.5 shows the area of land planted to corn in each Ontario county in 1990 and 2016. It can be noted that many of the counties in which corn production is high (Figure 2.5) are also the counties in which N_2O emissions are more intense (Figure 2.4). Figure 5A in the appendix shows livestock concentrations in Ontario in 1990 and 2016; such maps can be useful first step in determining the drivers of N_2O emissions.

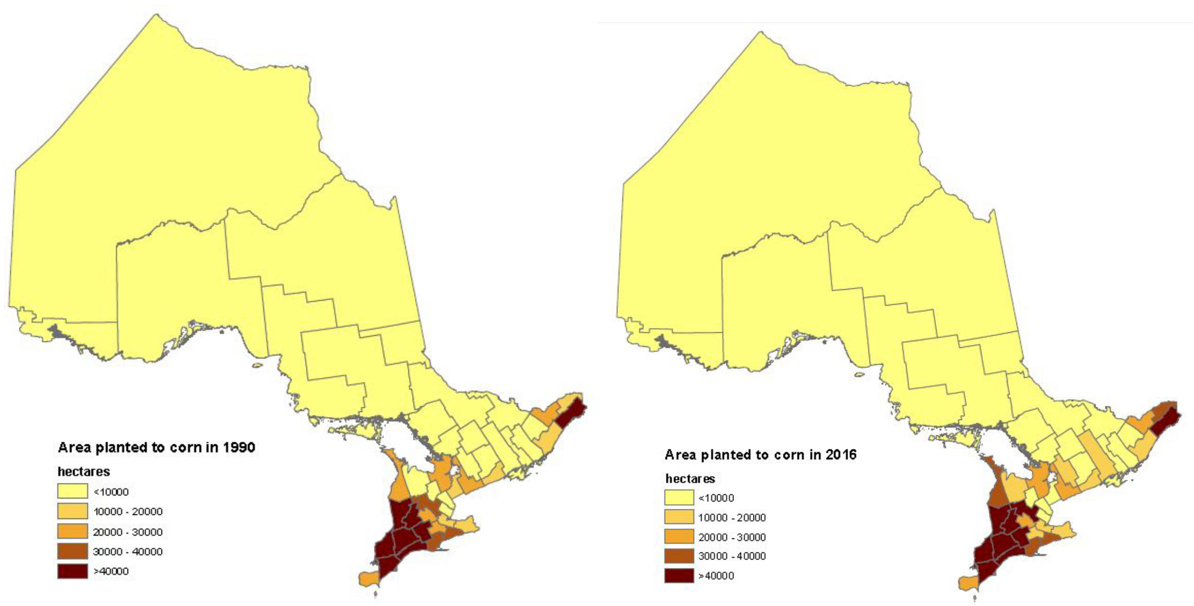

Figure 2.5. Area of land planted to corn (hectares) in Ontario counties in 1990 and 2016.

2.3. Conclusion

This section summarizes the updated county-scale inventory of N_2O emissions from Ontario's agricultural soils. The trends and changes in N_2O emissions since 1990 have been briefly discussed. Statistical analyses can be performed if necessary using the compiled data to determine the significance of these trends as well as correlation with agricultural factors of production. However, such analyses are beyond the scope of this report. The main purpose of this updated inventory is to serve as baseline for emission reduction scenarios that were developed as part of section 3.

3. GHG Mitigation Effectiveness of Soil BMPs

This section includes the results of the qualitative literature review of GHG mitigation of BMPs and the results of the quantitative meta-analysis focused on N_2O emission under Ontario conditions, when data were available. Ideally, the BMP efficacy would have been given for certain soil types and or certain landscapes or production systems. However, after the initial literature review, it became apparent that the data availability does not allow for such a categorization. Rather, we assessed the BMPs based on the available knowledge about soil processes and their regulatory factors (such as texture and pH), i.e. the assessments are not empirical. The BMPs that were considered are N management (rate, application method, type, use of inhibitors), crop management (CCs, diversification, perennial and biomass crops), soil management (tillage, variable rate fertilization, liming, use of biochar), manure management (timing, method, treated manure), and afforestation. These BMPS are related to the processes that could affect direct and indirect N_2O emissions, CO_2 emission and C sequestration.

3.1. Background on nitrogen transformations and cycling in the soil

Nitrogen is present in the soil as organic and inorganic N. For organic N to be used by plants or lost from the soil it has to first be transformed to inorganic N through the microbial dependent process of mineralization. The product of mineralization is ammonium (NH_4^+) which faces a number of different fates. The NH_4^+ not used by the plant directly, may bind to negatively charged parts of the soil and of organic matter (OM) (which protects it from loss), or it may undergo further transformations through the microbial-mediated processes of nitrification and denitrification or it may be converted to ammonia (NH_3) and volatilized. The balance between NH_4^+ & NH_3 is pH dependent with more volatilization occurring at higher pH. Figure 3.1 describes the different losses and pathways of N in the soil (this is not a complete N cycle). Direct N_2O emissions are those emitted from soil as a result of fertilizer and manure addition, organic matter mineralization, or disturbance activities such as tillage. Indirect N_2O emissions are those emitted from the leached and volatilized N and this occurs at some other location inside or outside of the farm/unit.

To assess indirect emissions from leaching and volatilization, the amount of N that is leached or volatilized is multiplied by the appropriate EF which is used by the IPCC (2006) and the NIR. The default EF is 0.010 (0.002-0.05) kg N_2O-N /kg N volatilized and 0.0075 (0.0005-0.025) kg N_2O-N /kg N leached.

If the amount of N leached/volatilized is unknown then the IPCC method assumes either default values or region-specific values to estimate the amount of N that is leached or volatilized from fertilizer-N and manure-N. The fraction of all applied-N that is leached (leaching and runoff into rivers and ground water) is based on the specific conditions related to the ratio of precipitation to potential evapotranspiration (P/PE). This fraction (N leached + runoff) is calculated as $F_{LEACH} = 0.3247 \times P/PE - 0.0247$. Therefore, if P/PE is 0.8, then $F_{LEACH} = 0.24$ kg N/kg N applied. Then, indirect N_2O-N is estimated as $F_{LEACH} \times 0.0075$.

The fraction of N volatilized from fertilizer is on average 0.10 (0.03-0.3) kg NH_3–N / kg N applied and the fraction volatilized from manure and organic fertilizer is 0.20 (0.05-0.5) kg NH_3–N / kg N applied. Then, indirect N_2O-N is estimated as N-volatilized x 0.010.

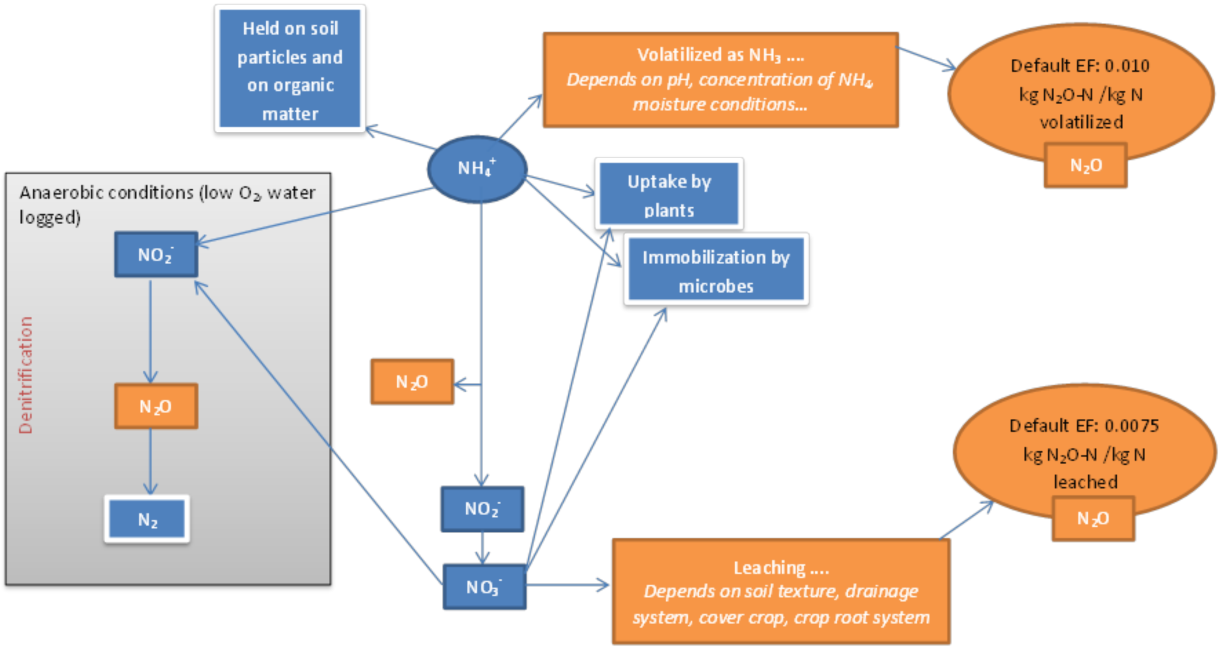

Figure 3.1. Schematic diagram of some N transformations in soil showing the sources of direct and indirect N_2O. Grey area represents anaerobic conditions. Orange highlighted sections are losses from the soil-plant system to the environment.

3.2. Methodology

- The selection of promising BMPs is based on consistent results showing reduction in GHG or gain in C stock or a combination of experimental results showing benefit to GHG mitigation in addition to a soil ecosystem process that is scientifically known to lead to mitigation.
- The literature search was conducted through Web of Science (Thompson Reuters) and Google Scholar (Google Inc.) as well as from personal communications with scientists and academics.
- The aim was to include only field studies. Laboratory incubations and greenhouse experiments were included if the literature from field studies for specific BMPs was scant.
- The preference for inclusion of studies was in the following order: Ontario and Eastern Canada and northeastern United States, followed by northern European countries, western Canada and in a few cases from Japan (because those from Japan were for full year including winter/spring). This was done so that the analysis covers climatic conditions that are similar to Ontario (snow cover in winter, wet conditions in spring and summer).
- Studies from other regions (e.g. Mediterranean irrigated studies) were included in the qualitative literature review (not in the meta-analysis) either because data from preferred

regions was unavailable or because a given study assessed long-term effects which was deemed important to mention.

- The data from the selected literature was manually mined.
- In many cases the data was extracted from figures/charts using the online software WebPlotDigitizer (https://automeris.io/WebPlotDigitizer/).
- In some cases values were calculated from published data to acquire relevant values (for example, to simply convert between units or to calculate cumulative emissions from different time frames, or to calculate intensity of emission per yield).
- If the published data was presented as averages of different treatments within an experiment and in the case that some treatments were not relevant to practices in Ontario (e.g. data averaged over irrigated and rainfed fields), then effort was made to generate a calculation for the separate treatments.
- Studies that reported cumulative emission data from short measurement periods were scrutinized and a subjective decision made whether to include or exclude the study based on the methodology and the details stated in the text of the specific study.

Meta-analysis is a statistical method that aggregates and contrasts results from different studies that address the same question (Glass, 1976). The analysis requires the input of an "effect size" measure and an "error" measure. In published studies, the first is usually easy to acquire (from tables and figures) whereas the error measure is not always available. Attempts were made to contact authors for data to improve our dataset; entries with missing measures of error were completed by generating a conservative error value equal to 150% of the average coefficient of variation of the available data. Therefore, because of associated methodical shortcomings, it is noted that assessments and initial results from the meta-analysis need to be examined closely by further probing and validation from the literature.

3.3. Nitrogen fertilizer management

3.3.1. N-Rate optimization to match crop need:

Reducing the rate of N applied to match crop requirements will result in less mineral N in the soil that is available for nitrification/denitrification losses and/or for leaching/volatilization losses. In theory this should lead to a reduction in GHGs but might also lead to a yield reduction. Therefore the selection of N rate should be based on a combination of factors: economical optimization, current availability of soil N (soil test), and reduction of surplus soil N at times when nitrification/denitrification potentials are high and crop uptake is low. Selection of N rate on the basis of a soil N test (e.g. Pre-Sidedress Nitrogen Test, PSNT) and on specific crop/soil recommendations aims to match crop needs with N availability and reduces surplus soil N (especially in NO_3^- form). In a meta-analysis on corn in North America, Eagle et al. (2017) found that the best relationship between N rate and N_2O emission was an exponential function where N_2O increased by 2.8-11.9% with each additional 10 kg N/ha of fertilizer. Additionally, these researchers used a hierarchical model to describe the relationship between application rates and emission and reported that N_2O emission increased by 2.7% with every additional 10 kg N/ha when fertilizer application rates were between 110 and 270 kg N/ha. A study in Ontario and Quebec by Ma et al. (2010) on corn showed that, across years and locations, the relationship between N fertilization rate and N_2O emission is described by an exponential function such that increasing the N rate from 90 to 150 kg N/ha resulted in doubling N_2O emission from 0.46 kg N_2O-N/ha to 1.04 kg N_2O-N/ha.

A study using the DeNitrification-DeComposition (DNDC) simulation model on corn systems in western Ontario by Anderson (2016) demonstrated a reduction in emissions when the N fertilizer rate was decreased from 170 to 150 kg/ha. Were this lower N rate adopted province wide, the reduction in N_2O emissions from corn would be 155342 Mg CO_{2e} per year (0.136 Mg CO_{2e}/ha/y; 0.29 kg N_2O-N/ha/y), equal to 10.6% reduction, while the effect on grain yield reduction would be either small or non-existent.

These results demonstrate that there is potential to reduce N_2O emissions by reducing the N application rate. Therefore, N applications based on soil tests is a first step in reducing direct and indirect N_2O emission without sacrificing yield. Optimizing N rate will also help mitigate indirect N losses through NO_3^- leaching because there is going to be less surplus N in the soil after plant uptake. The amount of N_2O emissions that can be reduced vary and are linked to fertilization timing, fertilizer placement, and the type of N fertilizer and are discussed below.

3.3.2. Timing of N application

Ontario has adopted a 4R nutrient stewardship strategy: the Right source at the Right rate, at the Right time and the Right place. Timing of fertilizer application, especially when the rate is reduced from the conventional rate, is an important consideration and is the basis for the concept of grouping N management options in the 4R plans. It is important that the fertilizer is applied when the crop is growing and taking up N from the soil. The relationship between N_2O emissions and the timing of N application is linked to weather conditions: temperature as well as the amount and timing of rainfall. Weather variability is one of the main reasons why there are conflicting results in the literature on this BMP. Results from a meta-analysis (Abalos et al., 2016b) as well as a few other studies were found to be inconclusive. The examples given below describe direct N_2O emissions and do not address the indirect losses of N which are affected by timing of N application. Indirect N losses are affected by precipitation and air/soil temperature on N transformations in the soil and on the degradation/dissolution of fertilizer coatings and inhibitors.

The fertilization options included in the many studies that examined the timing of fertilizer application are: fall one-time application, at-planting spring application, one-time sidedress application, and a split growing season application (split-application is discussed in the next section). A meta-analysis (Abalos et al., 2016b) showed that sidedress N applied mostly at the corn V6 stage was not statistically different than fertilizer applied at planting. The numerical average reduction in N_2O with sidedress was –18.5% and the range was +8 to –

38%. This study also reported a 4% increase in yield with N applied as a sidedress compared to all N applied at planting.

In a 3-year study in Ontario, Drury et al. (2012) showed that N_2O emissions were dependent on interactions between tillage, rainfall, and N application time. During some years, the application of N as a sidedress coincides with rainfall, and because of warmer conditions at the time of sidedress compared to planting, greater N_2O emission can occur. Warmer soil/air later in the growing season can also induce more N volatilization losses from broadcast or surface banded ammoniacal N. *Under conventional tillage (CT)*: over the 3 years, N at planting produced 26-40% more N_2O emissions than sidedress-N and using controlled release polymer-coated urea (PCU) did not affect this trend. Additionally, there was an 11% reduction in yield with sidedress-N in one year. *Under no-till (NT)*: the sidedress-N produced 53-83% more N_2O emissions in the 2 wet years whereas N_2O emissions were only slightly more from the N applied at planting in the dry year. Use of PCU generally caused an increase in N_2O emissions for both application times. *Under zone-till*: N_2O emissions were greater from sidedress-N than N at planting for 2 years (a wet year and a dry year) and using PCU in those instances was not effective. The only consistent effect across years and fertilizer types was that under CT, N as a sidedress reduced N_2O emissions compared to N applied at planting.

Another 2-year study in Ontario on corn production (Roy et al., 2014) showed that N applied at planting during a wet year resulted in more growing season N_2O emissions than when the same amount of N was applied at the V8 stage. However, in a second drier year, emissions from the V8 sidedress application were greater than when N was applied at planting.

In a simulation study using data from two field experiments in Ontario for calibration/validation of scenarios run in the DNDC model, Abalos et al. (2016c) found that sidedressed N (150 kg N/ha on corn) produced less N_2O emissions (av. 1.7 kg N_2O-N/ha) compared to N applied at planting (av. 2.3 kg N_2O-N/ha). This simulation was on a full year basis for growing and non-growing season emissions. However, the two treatments were not exactly identical which could have contributed to the change in emission; at planting, urea was incorporated and at sidedress urea ammonium nitrate (UAN) was injected. The simulations showed that yield was slightly reduced (av. 5%) with the sidedress compared to at-planting N applications which resulted in the yield-scaled emissions being greater for the sidedress application.

Fall application of N on corn was reported to be statistically similar to spring application in a meta-analysis including 11 side-by-side comparisons (Abalos et al., 2016b). This comparison however included both manure and mineral N sources. The data included in the analysis was highly variable and underscored the need for further research on this comparison. For mineral N fertilizers, Abalos et al. (2016c) used two field experiments in Ontario to calibrate/validate a DNDC (computer simulation model) scenario analysis and the results showed that fall broadcast urea produced more N_2O emissions (1.6-8.0 kg N_2O-

N/ha) than incorporated urea applied at planting (1.9-3.5 kg N_2O-N/ha) and injected UAN applied as a sidedress (1.1-2.6 kg N_2O-N/ha).

The three Ontario studies presented above show how the interaction between the different variables in the field affect emissions, the quantity and timing of rainfall being important factors, and why it is difficult to generalize the effect of N-application timing. In general, it can be said that if wet conditions are to be expected later in the season, especially true for NT fields, then sidedress N could cause a peak in emissions in that year.

Due to the variability in results, side-dressing and spring N application could not be selected with confidence as practices that can mitigate GHGs. Further research and more data collected under different conditions will help to identify conditions when these BMPs can be effective in GHG reduction.

3.3.3. *Split-N application:*

It is important here to distinguish between side-dressing and split N application: side-dressing is the application of N after crop emergence (which can be part of a split application or a single application of all N) whereas split-N is the splitting of the total fertilizer into at least two applications usually over the spring and summer and can include pre-panting/at-planting in addition to side-dressing. The side-dressing section 3.2.2 included only one-time side-dress applications so they differ from the split-N applications discussed in this section.

Split-N application into several applications based on crop demand potentially reduces the availability of unused NO_3^- in the soil early in the season but can also have an increase in overall emission resulting from application-induced pulses. Emission amounts depend on the same conditions stated above for sidedress N. It should be noted that split-application requires multiple applications on the same field resulting in a non-soil related increase in the C footprint. This point should then be considered from a full C-budget point of view.

There were two studies on N_2O emissions from split N application in Ontario from the literature search. Abalos et al. (2016c) showed that split application in three times: at planting (60%), V6 (20%) and V8 (20%) did not result in significantly less N_2O emissions (av. 2.1; 1.4-3.5 kg N_2O-N/ha) compared to all N as a sidedress (av. 1.6; 1.1-2.4 kg N_2O-N/ha) or N applied at planting (av. 2.3; 1.8-3.5 kg N_2O-N/ha) (all treatments used a UI +NI). This simulation showed that the yield in the split application was similar to that when N was applied at planting and was better than when N was sidedressed at the V6 stage. Compared to another simulation study in Western Ontario (Anderson, 2016), where split-N was applied as 70% pre-plant and 30% side-dress (at the V4-V6 stage), there were 21% less N_2O emissions compared to when all N was applied at planting. Further, there was an average 6.2% reduction in yield. This is contrary to the report from Abalos et al. (2016c). The difference between these two studies could be related to the use of UI and NI in the Abalos et al. (2016c) study as well as the difference in the split percentages and timings.

In a Minnesota corn study over two years (Venterea et al., 2016), N_2O emissions were similar from a split application of urea (split into three applications at 11 days after planting, at the V6 stage, and at the V14 stage) and from a single application (10-11 days after planting). The split application of urea+NI+UI reduced FIE by 12%, and when this same treatment was combined with a 15% reduction in N rate there was about 50% reduction in FIE. A statistical analysis of data from North America on corn production (Omonode et al., 2017) found that N_2O emissions with pre-plant N were lower than early sidedressed N (V8 stage) which was lower than split applications that included later sidedress application (V14 stage). The conclusion was that the application of the right amount of N at the right time (especially pre-plant and early sidedress) was more likely to reduce emissions compared to split application of N that involved N application at stages greater than the V12. The authors also noted that synchronizing plant uptake (which is usually high at the grain-filling stage) with N supply is likely to reduce overall N losses. The grain-filling stage of corn growth is a phase of large N uptake, however timing the split N application at this late stage is tricky because short-term excess N can lead to denitrification losses and can increase overall N_2O emission. The literature reports on emissions from split N are not consistent as shown by results from a hierarchal multi-level regression by Eagle et al. (2017), also on corn in North America, where N_2O emission reduction was estimated to be between 20-26% when at least a portion of fertilizer N is split applied as a side-dress compared to all pre-plant N (spring or fall).

Split application of N fertilizer needs further research to address the following three issues. First, there needs to be an assessment of its efficacy especially when a full C-budget is considered. In addition, more data is required to elucidate the interactions between emissions under this management and other interfering factors such as tillage and environmental conditions as they relate to crop yield. Finally, more studies on interactions with other management options such as the addition of inhibitors is needed especially since there is evidence from the literature that combinations of management options have potential for better control on emissions and other losses.

3.3.4. N-placement or application method:

There were few studies that compared emissions from different N fertilizer placement treatments. Placement options (broadcasting, broadcasting and incorporation, injection, and banding) can affect the form of N loss to some extent. For instance, broadcast N fertilizer which is left on the soil surface uncovered, especially urea-based N, is more likely to increase NH_3 volatilization leading to indirect N_2O emissions whereas placement in concentrated areas and at depth tends to increase direct N_2O losses. Although broadcast and incorporation of N fertilizer is thought to reduce N losses and is advocated as a BMP, the literature does not contain enough evidence to statistically confirm this (Abalos et al., 2016b; Decock, 2014). A meta-analysis on corn studies in the United States and Canada (Decock, 2014) found that there is no effect of broadcast or banding of N fertilizer on N_2O emissions. Injection of liquid N fertilizer and the effect of injection depth are also not well covered in the literature. It is generally thought that a hot-spot for denitrification is formed at the N injection site and can result in increased direct N_2O emissions. This is supported by a study by Eagle et al. (2017) from a multi-level regression model where broadcasting N fertilizer as opposed to injection or banding was found to reduce emissions by 25-33%.

Volatilization of NH_3 (indirect N_2O emission) is a function of availability of NH_3 near the soil surface, pH, water content and temperature (Rochette et al., 2014). A study in Ontario (Drury et al., 2017) comparing broadcast urea and injected UAN (130 kg N ha^{-1}), both sidedressed to corn, showed that NH_3 volatilization loss after urea application amounted to 68% (2013) and 40% (2014) of the applied urea-N. The 2013 year was wetter than 2014 (the growing season rain was 605 mm and 558 mm in 2013 and 2014, respectively and 349 mm and 186 mm over June and July in 2013 and 2014, respectively) and the fertilizer was applied at a later date (July 15) than in 2014 (June 30) which contributed to the larger volatilization in 2013. Additionally, 2.1% and 0.5% were lost as direct N_2O in the urea treatment. In comparison with UAN injection, NH_3 loss was 32% and 9% and N_2O loss was 1.7% and 0.7% in 2013 and 2014, respectively. Therefore, without any additional treatments, ammonia loss accounted for the majority of the N loss and the authors noted that the relatively high loss of ammonia with injection of UAN is related to soil type which was high in clay where the injection slot did not seal after application and resulted in exposed UAN. This suggests that in non-swelling clay soils or soils that have less clay content, injection of liquid urea-N sources would not result in as much NH_3 volatilization. Addition of a double inhibitor (UI + NI) to broadcast urea lowered NH_3 volatilization to levels similar to those that occurred with injected UAN (N_2O emissions were not much affected by the use of inhibitors in this case). It should be noted that grain yield was greater with injected UAN than with broadcast urea and this is a reflection of the amount of available N left in the soil for crop uptake (specific yield-weighted emissions were lower with injection for this reason). Banding of urea at 5-10 cm depths also reduced NH_3 volatilization loss compared to surface banding in a Quebec study on corn (Rochette et al., 2014).

Based on these studies, it is not clear if banding is preferred to broadcasting or if injection is going to reduce N losses when leaching is considered as well. The literature review did not reveal reasons to refute the recommendation of coverage/incorporation of broadcast fertilizer as a BMP. The placement of the fertilizer, which in some cases is dictated by the type of fertilizer and the timing of application, can be effectively managed to reduce N losses (especially indirect emissions) but the management is site specific and related to weather. More studies are needed for the assessment of the fertilizer application methods that account for all forms of N loss (N_2O, NH_3, NO_3^- leaching). Although it has potential, fertilizer placement was not selected as a mitigation BMP because more research is needed to cover the interactive effects and the relation to yield.

3.3.5. N fertilizer type:

Most comparisons of fertilizer types in the literature are between anhydrous ammonia (AA) and urea or ammonium nitrate (AN). Several studies in the United States on corn reported nearly doubled N_2O emissions from spring applied AA compared to spring applied urea (Fernández et al., 2015; Venterea et al., 2010) on silty loam to silty clay loam soils. On a loamy sand soil Fujinuma et al. (2011) also reported higher N_2O emissions from spring application of AA injected at 20 cm and at 10 cm compared to urea that was incorporated to 10 cm depth. While the use of AA in Ontario is not widespread, a shift in use from AA to urea would likely result in reduction in N_2O emissions from corn crops. A meta-analysis on corn in North America (Eagle et al., 2017) reported that a shift from AA to urea results in a 45% reduction in N_2O emissions, while a shift from urea to urea+NI+UI results in a 26% reduction and finally a shift from urea to PCU results in a 15% reduction.

From our collected dataset the studies reporting on fertilizer induced emission (FIE), only those from mineral fertilizer sources were selected and analysed in a meta-analysis. The dataset included the following N sources: ammonium+nitrate and ammonium forms of N, urea, UAN, and urea+UAN. The results showed that the type of fertilizer did not statistically affect FIE in the growing season (Figure 3.2). The included studies took place in Ontario, Quebec, New Brunswick, Iowa, Kentucky, and Pennsylvania. It is possible that including more studies from other locations in the dataset could have improved the analysis and resulted in smaller confidence intervals.

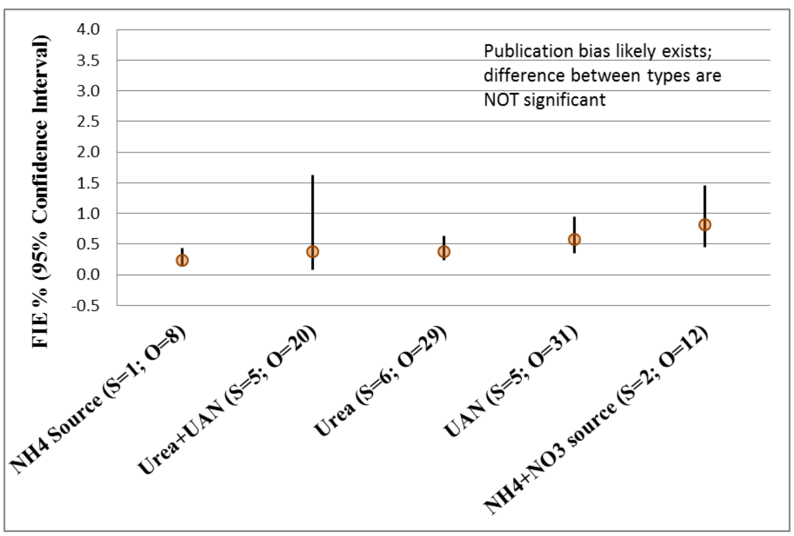

Figure 3.2. Meta-analysis results showing estimated fertilizer induced emission percent (FIE) for ammonium-source fertilizers, Urea+UAN, Urea, UAN, and ammonium nitrate-source fertilizers. The S and O indicate the number of studies and number of observations, respectively. Because all the error bars overlap, the differences are not significant.

Fertilizer type should theoretically have an effect on N_2O emissions. For example, having a nitrate-source fertilizer is expected to result in more N_2O emissions. However, the literature review did not produce enough data for our meta-analysis to allow for the categorization of fertilizer types for BMP.

3.3.6. *Mineral fertilizer vs manure N:*

The comparison between N_2O emissions from mineral fertilizers and manure is discussed in this report however the decision to use either source as a fertilizer does not just relate to emissions or soil benefits/improvement or even crop yield. There are additional considerations such as availability, convenience, and cost. Manure is usually produced and used on the farm, and if not used as a fertilizer, alternative uses or means of disposal of manure (or excess manure) need to be available. Overall, our findings seem to indicate that direct and indirect emissions from manure are greater than from mineral fertilizer. This means that manure use is not a best option for reduction of emissions, however as stated by Abalos et al. (2016b) there are socio-economic and environmental considerations that could outweigh the negative effects on emissions or yield. The benefit of manure as a C source in the soil is not considered in the analysis below but should be included when a full assessment of the environmental impact of manure use is made. Therefore, based on the available data, replacing mineral fertilizer with manure as an organic-source fertilizer is not selected as a BMP for emission mitigation.

Two meta-analysis on corn in North America reported different results when comparing emissions from manure versus mineral N. Abalos et al. (2016b) found no statistical difference when comparing 91 data points but the average was about 9% reduction under manure use. Decock et al. (2014) found that there is an average reduction in N_2O emissions of about 40% when using mineral fertilizers. They compared 73 observations, however in 30% of the studies the amount of manure-N applied was higher than the mineral-N applied. This is still a valid comparison because manure application usually aims for a higher N rate because it is expected that manure-N mineralization is slower. Additionally, Abalos et al. (2016b) found that there was an 11.3% reduction in corn yield when using manure compared to mineral fertilizers. This was attributed to the better synchrony of mineral fertilizer-N with crop demand. This effect of yield reduction under manure N might be found more in fine textured soils because N limitation is the main growth limiting factor in these soils whereas in other soil types N and moisture availability are limiting factors.

A meta-analysis was conducted on the dataset collected for this report. Studies that included side-by-side comparisons (80 data points) of emissions from manure versus mineral fertilizer showed that the application of manure results in 38% (95% confidence interval (CI): 18-62%) more N_2O emissions than when mineral fertilizer is applied. The amount of added N was very close in both treatments with an average of 128 and 130 kg N/ha for mineral fertilizer and manure, respectively. This is different than the study by Decock et al., (2014) where more N was applied with the manure than the mineral fertilizer.

The crops included in the meta-analysis were corn and timothy grass. The data set included mostly liquid manure sources except one with poultry litter and another with solid cattle manure and two studies with either anaerobically digested or otherwise treated manure which ranged in ammoniacal N from 2.3 to 3.8 kg N/m^3 and in dry matter from 13 to 46 kg/m^3. In our analysis, the timing and method of manure application had no effect on mitigating the increase in N_2O emissions stemming from manure use. However, there were differences in N_2O emissions related to manure use on different soil textures. Figure 3.3 shows that manure use results in between 50-80% more N_2O emissions than mineral fertilizer on coarse and medium textured soils. On fine textured soils, the difference in emissions between manure and mineral fertilizer is not significantly different. It should be noted that only direct N_2O emissions are considered here. This assessment does not take into account the effect on yield or the losses of N though volatilization and leaching. Ammonia volatilization is expected to be higher under manure use and yield could also be impacted but there were not enough data on these criteria to make a statistical assessment.

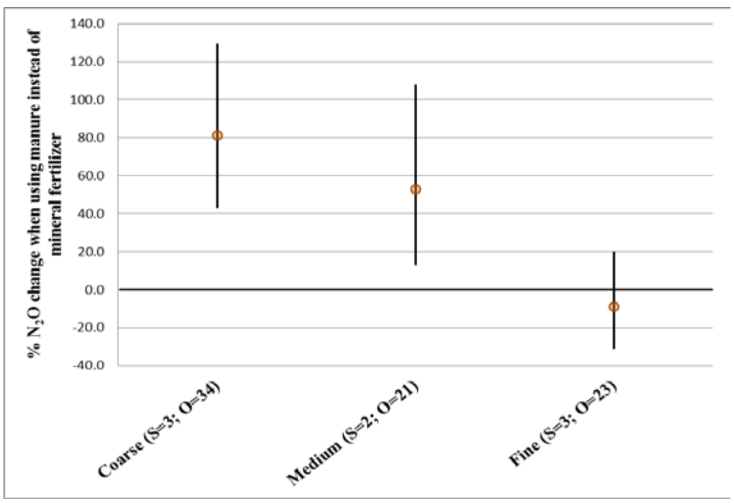

Figure 3.3. Change (%) in N_2O emission from using manure instead of mineral fertilizer on different types of soil. The numbers following the soil type are number of studies (S) and number of observations (O).

3.3.7. *Use of inhibitors, polymer coating, and slow release fertilizers*

NIs work with ammoniacal type fertilizers by reducing the rate of conversion from NH_4^+ to NO_3^- by inhibiting the activity of the bacteria responsible for the first step of nitrification (*Nitrosomonas*). Inhibitors are affected by soil moisture, temperature and soil type with the breakdown of NI faster in warm soils. The use of NIs keep the nitrogen in NH_4^+ form longer therefore they reduce the chances of NO_3^- leaching (because NH_4^+ is used by plants or is held on negatively charged soil particles).

UIs work with urea-type fertilizers through the inhibition of the microbial enzyme urease which converts urea to NH_4^+. Urea hydrolysis is thereby delayed or slowed down usually for 1-2 weeks. For surface applied urea, UI allows for more time for rain or another mechanical incorporation of urea into the soil therefore reducing the amount of ammonia volatilization during this time.

PCU is a controlled-release type of fertilizer where N from granular urea is released by diffusion through a polymer coating when in contact with water. Environmentally smart fertilizer (ESN) is a polymer coated urea. The rate and coating lifetime is affected by temperature and moisture and the chemical type of the coating. Volatilization and leaching are reduced because of the controlled N release rate. However, if the rate of release is slow the amount of urea-N available might not match crop needs and yield would be affected. Sulfur coated urea is a slow-release fertilizer and this type of coating is degraded by microbial activity or physical/mechanical damage or chemically which means that the rate is less controlled than PCU. Due to the variable nature of fertilizer release, the effectiveness of coated fertilizers and their effect on yield is more inconsistent than with NI and UI use, and this is reflected in the literature findings.

Overall, the literature review shows mixed results from PCU use but supports a reduction of N_2O emissions when using inhibitor-treated fertilizers. The most consistent reports of reduction in N_2O emissions were from NI and double inhibitor (NI+UI) use which is why this BMP was selected as a potential option for mitigation of GHGs.

<u>PCU:</u> From a study in Ontario, Drury et al. (2012) found that there was an inconsistent effect of PCU compared to urea that varied with tillage treatment and weather conditions. The authors reported that PCU was only effective in reducing emissions under wet conditions in conventionally tilled soils. The effect of PCU varied from an increase in N_2O (+8%) (Vyn et al., 2016; International Plant Nutrition Institute (IPNI) report) to a range of +7 to –18% difference in yield-scaled N_2O (Eagle et al., 2017; meta-analysis) when switching from urea to PCU. A meta-analysis including studies from the United States, Canada, Germany, China, Japan, India and other countries (Thapa et al., 2016) analyzed the effect of controlled release fertilizers (CRF), the most commonly used of which is PCU, on reducing emissions found that CRF (e.g. PCU) had a 19% (–28 to –12) reduction in N_2O emissions compared to conventional fertilizer however there was either no benefit or a negative effect on yield. Akiyama et al. (2010) reported meta-analysis results that polymer coated fertilizers did not significantly differ from regular fertilizers on upland soils but that there was publication bias with this result.

<u>Urea+NI+UI and double inhibitors:</u> The IPNI (Vyn et al., 2016) estimated 12-61% reduction in N_2O emission when urea+NI+UI was used compared to urea in rainfed systems. The big range in reduction seems to be related to the sources of data used. A study by Eagle et al. (2017) calculated yield-scaled N_2O emission reduction of 26% (range: 13-39%) when using urea+NI+UI instead of urea alone. The meta-analysis by Thapa et al., (2016) showed that double inhibitor (UI+NI) was not better than NI alone in reducing N_2O emission but that there could be an added benefit to double inhibitors which is the reduction of indirect emissions from NH_3 volatilization (effect from UI). Crop yield was better under

NI alone than under both UI+NI demonstrating that double inhibitors do not provide the same synchrony in N release and crop demand as NI. In alkaline soils, UI+NI might be a better option if urea is used as fertilizer as UI would delay the hydrolysis of urea to NH_4^+ (therefore reducing the likelihood of NH_3 loss) and NI, shown to have a positive effect on yield, would regulate the rate of denitrification. In Ontario, Drury et al. (2017) compared UAN with UAN+NI+UI (the NI dycandiamide (DCD) + the UI N-(N-butyl)-thiophosphoric triamide (NBPT)) (injected) and also compared urea with urea+NI+UI (broadcast). The two-year averages of N_2O emissions were not significantly different: 1.7 kg N/ha for urea and 1.5 kg N/ha for urea+NI+UI. The use of urea+NI+UI resulted in a significant reduction of NH_3 volatilization but not a significant yield increase compared to urea. Injected UAN (1.5 kg N_2O-N/ha) did not produce significantly more N_2O emissions than UAN with NI+UI (1.5 kg N/ha).

<u>NI:</u> The IPNI (Vyn et al., 2016) estimated a 27-56% reduction when UAN+NI (nitrapyrin) is used compared to UAN alone in rainfed systems. Thapa et al (2016) reported that over all data, NI reduced N_2O emissions by 38% (–44 to –31%) and there was a 7.1% increase in grain yield (wheat). They also reported that NI worked well in acidic soils to reduce N_2O emissions (av. 59% reduction when NI was used). The authors noted that DCD NI residue was found in animal feed and its use was suspended in New Zealand. Similar results were reported by another meta-analysis (Akiyama et al., 2010) where the reduction of N_2O emissions with NI use ranged from –43 to –26% compared to regular fertilizer alone. As well, Eagle et al. (2017) found that NI reduce N_2O emissions from corn fields by an average 31% as estimated from a multi-level hierarchical regression model.

<u>UI:</u> From a study in Ontario, Drury et al. (2017) compared UAN with UAN+UI (NBPT) (injected) and urea with urea+UI (broadcast). Compared to urea alone (1.7 kg N_2O-N/ha), there were more N_2O emissions when urea+UI was used (2.2 kg N_2O-N /ha). This was likely because UI reduced NH_3 volatilization which resulted in a bigger concentration of mineral N in the soil leading to greater N_2O-N compared to urea use alone. Injected UAN (1.5 kg N_2O-N/ha) did not produce significantly more N_2O emissions than UAN+ UI (2.0 kg N/ha) however UAN+UI tended to produce more N_2O emissions when averaged over two years and about 3-9% more grain yield (yield not significantly different between UAN and UAN+UI). The meta-analysis by Akiyama et al (2010) found that N_2O emission from UI was not significantly different than from fertilizer alone. Thapa et al. (2016) also found that UI use did not reduce N_2O emissions when all the data was analyzed together (range –42 to +2%) however UI use was effective when the data was separated by crop or soil type. Example: for corn, N_2O emissions were reduced by an average of 36% (–55 to –17%) with UI use compared to conventional fertilizers and in coarse-textured soils N_2O emissions were reduced by 28% (–55 to –4%) with UI use. It should be noted that there may bias in this analysis as the number of studies included in these specific tests is smaller and spread over a wider range of geography.

There was only one study from Ontario (Drury et al., 2017) that compared N_2O emissions from fertilizers alone and fertilizers with inhibitors in the dataset we compiled. We therefore included studies from Germany (Weiske et al., 2001) and the United States (Dell et al., 2014; Parkin and Hatfield, 2014; Sistani et al., 2011). The dataset contained side-by-side comparisons of urea versus urea+UI (NBPT), urea versus urea+NI+UI, UAN versus UAN+UI+NI (Agrotain®Plus), and ammonium sulphate nitrate (ASN) versus ASN+NI (two types of NI, DCD and 3,4-dimethylpyrazol phosphate (DMPP)). We did not include PCU in this analysis. Meta-analysis comparing untreated fertilizer versus inhibitor-treated fertilizer showed that there is a reduction of ca. 8% (+5 to –19%) of N_2O when using the inhibitor-treated fertilizers but since the range crosses zero, this effect is considered non-significant. However, the type of inhibitor (double, NI, UI) had an effect on the results such that the only significant effect that consistently reduced emissions was ASN+NI and the reduction ranged from 28 to 46% (Figure 3.4), which is similar to the result by Thapa et al. (2016). However, since the literature shows that UI reduce NH_3, it is recommended to apply both NI+UI to help reduce both direct and indirect N_2O emissions.

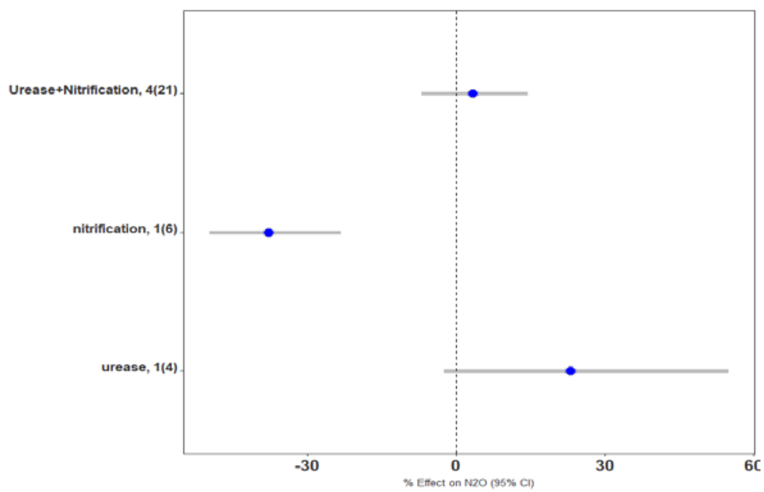

Figure 3.4. Meta-analysis results showing estimated % effect of using fertilizers with inhibitors instead of fertilizer alone. The numbers following the inhibitor type are number of studies (number of comparisons). Confidence intervals were bootstrapped with a 1000 iterations. Studies included were from Canada, United States and Germany only and results are for growing season emissions.

3.4. Crop management

3.4.1. Cover crops

Cover crops (CC) provide many benefits to soil health and to farmers. These benefits include but are not limited to the addition of organic matter into the soil providing N for later crops, the opportunity for an extra crop, and the reduction of soil erosion. However, the potential increase in N_2O emissions from soil due to CC is unclear. Literature findings on the influence of CC on GHGs are inconclusive mainly due to when measurements are made. It makes a difference if measurements are made during the growing season of the CC, after CC incorporation/termination, during the growing season of the main crop, or for the whole year. Additionally, because CCs affect NO_3^- levels in the soil usually by reducing them, they will have an effect on indirect N_2O emissions related to NO_3^- losses, and this is not always accounted for when reporting GHG emissions from CC experiments. The main potential benefits of CCs in relation to GHG mitigation are: C input to the soil; mitigation of indirect N_2O emission through the capture of excess NO_3^- after the main crop harvest; and reduction in the N rate of applied fertilizer through the provision of organic N for the following crop.

C sequestration with CC:

A modelling experiment in California showed that CC decrease Global Warming Potential (GWP) mainly due to an increase in the SOC storage from CC (De Gryze et al. 2010). A study by Poeplau and Don (2015) modelled C sequestration under CC systems from widespread data (73% from temperate regions) and reported a SOC sequestration potential of 0.32 ± 0.08 Mg C/ha/y which was not affected by the type of CC or the tillage system. This is in agreement with the estimate given by Eagle et al., (2012) of a sequestration potential in SOC of 1.34 (–0.07 to +3.22) Mg CO_{2e}/ha/y (0.37 Mg C/ha/y) and when all process and upstream mitigation is included the total potential is estimated at 1.92 (0.51-3.81) Mg CO_{2e}/ha/y (0.52 Mg C/ha/y).

N_2O emission reduction with CC:

A meta-analysis by Basche et al. (2014) using data from many countries found that in general CC increase N_2O emissions but that there are many factors that play a role in the emission rate. Their analysis showed that when full-year measurements were made, systems with CC produced N_2O emissions similar in amounts to systems without CC. There was an interaction between the rate of fertilizer N applied and the type of CC, legume or non-legume. When no additional N was applied, the legume CC induced high N_2O emissions. This is because the low C:N ratio (<25) induces mineralization of crop residues and increases NO_3^- levels – an indirect contribution to N_2O emissions. However, soils that received high N rates (> 200 kg N/ha), the non-legume CC produced more N_2O emissions. There is also an interaction of CC with tillage. Fertilizing a CC that is grown on a NT system might result in greater biomass being produced which creates a mulching effect on soil and retains more soil moisture thus increasing denitrification and N_2O emission. An important finding from this study is that the incorporation of CC into the soil results in more N_2O emissions compared to non-incorporated residue. This finding is corroborated by others studies. For example, Petersen et al. (2011) in a study in Denmark, reported that CC (fodder radish) might be an option for mitigation of N_2O emissions in a reduced tillage or NT management but that emissions increased in a CT field after radish incorporation. This meta-analysis seems to indicate that non-legume CC without incorporation and with reduced N application rate for the following crop results in lower

N_2O emissions. Legume CC use would produce lower N_2O emissions if the following crop is fertilized to regulate the soil C:N ratio.

From the meta-analysis by Basche et al. (2014) it is clear that there is an interaction between the amount of available C in a soil, the CC, and N_2O emissions. The addition of CC increases the amount of SOC which increases microbial respiration. Increased respiration can deplete O_2 levels in the soil leading to denitrification and increased N_2O (Petersen et al., 2011) i.e. the background SOC and the C:N ratio of the CC residue should in theory have an effect on the N_2O emission from CC. However, a general consensus is lacking on this subject and there is a need for further research.

With respect to non-growing season effects, a study in Harford, New York (Dietzel et al., 2011) analyzed the effect of winter rye CC seeded into corn to replace winter fallow. The results of the two-season September to April/May study showed that winter/spring N_2O emission fluxes were smaller in the cover cropped soil compared to fallow fields when snow was not present. With climate change and more thaw events expected in winter, having a CC could mean capture of more NO^-_3 and less direct and indirect N_2O emissions. For Ontario, Wagner-Riddle and Thurtell (1998) observed that the presence of plants over winter (alfalfa or grass) can result in negligible emissions during thaw and suggested over-wintering CCs could reduce N_2O emissions. In a 10-year field study in central Iowa Parkin et al. (2016) compared NT corn-soybean rotations with and without a winter rye CC on a clay loam soil. The results showed that the cumulative full-year direct N_2O emissions were not different in the presence or absence of the CC when averaged over 10 years (noting that the CC was terminated chemically without plowing).

<u>Effect of CC on indirect N_2O emissions through the reduction of NO_3^-:</u>

From the Iowa study by Parkin et al. (2016) mentioned above, NO_3^- leaching was significantly reduced with the rye CC with 10-year cumulative losses of 359 ± 151 kg NO_3-N /ha for the control and 167 ± 27.5 for the CC treatment. This indicates that there is a benefit of reduction of indirect N_2O emissions from systems with CC, in this case winter rye, even when there is no difference in direct emissions.

Indirect N_2O emissions from NO_3^- leaching was investigated in clay loam fields with controlled tile drainage and unrestricted tile drainage with and without a winter wheat CC in Woodslee, ON (Drury et al., 2014b). Nitrate leaching was monitored over five years in corn-soybean rotations. The presence of winter wheat significantly reduced the 5-year average NO_3^- concentration (mg N / L) by 21% under the unrestricted tile drain system and by 38% under the controlled tile drain system. Total NO_3^- loss (kg NO_3-N /ha) was reduced by 14 to 16% in the presence of the winter wheat CC. There was an increase in the 3-year average of the soybean yield by 8 to 15% with the presence of CC. Therefore, depending on the drainage system in similar soils, the 21 to 38% reduction in NO_3^- leaching related to CC, means that more N is available in the soil for the following crops. There are then two benefits the environment: reduction of both water contamination and indirect N_2O emissions.

Reducing corn N fertilizer rate by using CC (Ontario studies):

N credit from CC replaces part of the fertilizer needs and, although N_2O emissions might be similar, using less fertilizer is indirectly beneficial. Less N is lost through volatilization and leaching, and therefore the need for N fertilizer application is diminished. The following studies show the potential for reducing/eliminating N rates when CCs are included before corn. Nevertheless, there is a need for more tests to be done to make precise predictions on different soils and types of CC.

The amount of available N for corn following legume CCs was studies over two years in Ontario (Coombs et al., 2017). The study findings suggest that spring as opposed to fall termination of alfalfa and red clover is better for providing N for the following corn crop. There was an improvement of corn grain yield in one of two years when corn followed CC without the addition of any N fertilizer. Over two years and two soil types (sandy loam and loam) alfalfa provided 37-63% and red clover provided 46-65% of plant available N compared to soil that was fertilized at 224 kg N/ha.

Another Ontario study, (Vyn et al., 2000), studied the N availability for corn following winter wheat with different CC on NT and fall tillage systems. Red clover, rye, oilseed radish, and oat were tested as CCs. The legume red clover was as effective as the non-legume CC in reducing NO_3^- levels in the fall and in the spring. Nitrate levels with the red clover CC were 35 to 49% lower than the no-CC plots as an average of both fall and spring sampling dates. When no additional N fertilizer was applied to corn, grain yield was generally better in the CT than the NT under almost all CC types and grain yield was consistently higher after red clover compared to other CC types regardless of tillage system. When 150kg N/ha was applied to corn after the CC, the effects of CC type and tillage system were minimal. It was noted that the cereal CC (rye and oat) resulted in lower corn grain yield compared to a no-CC control suggesting that these CC cause N immobilization, which is why they responded better to the addition of 150 kg N/ha to the corn crop.

Using winter wheat as a CC in corn-soybean rotations was shown to improve yield of both crops in a 4-year study in Ridgetown, Ontario (Gaudin et al., 2015a). Four-year averages showed that corn yield was improved by 16.6% and 18.8% respectively in zone-till and till systems noting that the yield improvement varied between years and tillage systems depending on weather conditions. The N rate that maximized grain-N did not decrease with the inclusion of winter wheat in the corn-soybean rotation under till systems however it decreased from 192 to 150 kg N/ha in zone-till systems. Addition of a red clover CC frost seeded to winter wheat reduced N rate that maximized grain-N yield to 101 and 136 kg N/ha in tilled and zone-tilled systems, respectively.

Therefore, the use of CC as a BMP was selected as an option that has potential to mitigate GHG emissions because it consistently showed at least a net-zero or reduction in N_2O emissions when a full year is considered. Additionally, the use of CCs has potential to reduce indirect N_2O emissions and increase C sequestration. The negative interaction with NT systems should be noted before implementation.

3.4.2. *Crop rotations and diversification*

There are very few studies that measured impact of rotation and crop diversification on GHGs. The timing of GHGs measurements in crop rotation studies is important because GHG quantification from full year or full cycle versus that from the cash crop growth season would be different. One long-term field experiment in Ontario compared continuous corn to corn in rotation (corn-oat-alfalfa-alfalfa) over three years (Drury et al., 2014a). Corn yield increased when planted in rotation (average 10 t/ha) compared to continuous corn (average 5.5 t/ha). Even when corn was not fertilized, it still produced 3.93 t/ha when in rotation because it benefited from the alfalfa residue from the previous year (N supplied by alfalfa plow down was estimated at an average 242 kg N/ha). On a 3-yr average, emissions were greater under continuous corn (7.4 kg N_2O-N/ha) compared to corn in rotation (6.5 kg N_2O-N /ha) and yield-scaled emissions were even lower in favour of corn in rotation. However, this study does not account for the full rotation effects. It is not known if emissions from the alfalfa growing years or the plow down of alfalfa would have emissions that are larger than continuous corn.

Inclusion of a perennial crop in rotation was reported in Ontario by Gregorich et al., (2001). Continuous corn was compared to corn-oat-alfalfa-alfalfa rotation from a 35-year experiment. The amount of SOC was about 20 Mg C/ha greater in the rotation than the continuous corn. This translates to a rough estimate of C storage of 2.1 Mg CO_{2e}/ha/y under the diversified system that included alfalfa. It should be noted that when perennial crops are included in a rotation there is automatically a reduction of tillage frequency compared to the corn system.

A long-term experiment in Ridgetown, ON, while not measuring GHG mitigation, examined the impact of crop rotations on soil health based the Cornell soil health assessment (CSHA) (Van Eerd et al., 2014). Soil was assessed after 11 years of establishment and compared between the following crop rotations. Continuous corn (C-C) was compared to continuous soybean (S-S), corn-soybean (C-S), soybean-winter wheat (S-W), and soybean-winter wheat-corn (S-W-C). The soil from the S-W rotation, which includes the highest frequency of winter wheat, scored the best followed by S-W-C. The inclusion of winter wheat especially improved the soil aggregate stability. The organic C content was also the highest in the top 5 cm of soil in the S-W rotation (21.5 Mg C/ha) under the CT treatments compared to all other rotations which had an average of 14.1 Mg C/ha. All the rotations were found to have similar SOC when managed as NT systems with an average of 20.7 Mg C/ha. No differences in SOC were found between rotations in deeper soil layers down to 120 cm. This research points to the general benefits to soil health and the improvement of soil C sequestration by increasing the frequency of winter wheat in rotations, especially under CT where winter wheat seemed to have offset the effects of tillage and made the SOC comparable to that under NT.

A report by Eagle et al. (2012) showed that the estimated SOC sequestration from crop diversification is zero Mg CO_{2e}/ha/y (–1.7 to + 1.7) from 90 comparisons with an estimated reduction in N_2O emission given as 0.17 Mg CO_{2e}/ha/y. The data used for the calculation of these estimates were from diverse regions and the ranges might not apply to Ontario conditions. Therefore, research is still required to examine the total effects of crop diversification and intensification of rotations. Due to the paucity of data and therefore the uncertainty about the full-year impact from the non-cash crop phases of rotations, diversification and long rotations that include perennials was not selected as a potential BMP for GHG mitigation although it is acknowledged that there are benefits to these practices.

3.4.3. Inclusion of long-term perennial and biomass crops

Perennial crops in agriculture could be grown as hay, used for pasture, as biomass crops for biofuel or as a land conservation/restoration practice on degraded and marginal land. There is a special interest in switchgrass and silvergrass (miscanthus) as perennial biomass crops because of their C4 plant efficiency in C assimilation and the large root system, the latter being especially true in switchgrass which has a dense and deep root system adding a relatively large input of C to the soil.

The impact of perennial biomass crops on GHG mitigation is through the replacement of fossil fuel use, which is beyond the scope of this report, and also through N_2O and CO_2 emission reduction and C sequestration when compared to annual cropland. The effects on GHG emissions are a result of the reduction in tillage activity, lower fuel consumption from machinery, lower amounts of fertilizer application, and more efficient N cycling. Perennial deep-rooted crops can also be beneficial in reducing indirect N_2O emission because they can capture NO_3^- and require less fertilizer inputs (Gopalakrishnan et al., 2011; McIsaac et al., 2010).

The potential to sequester soil C is dependent on the land-use change. If perennial crops replace cropland there is a potential for C sequestration however, if they are established on grassland there is likely going to be a C loss. Conant et al., (2001) reported that conversion of cultivated land to pasture including grasslands has an average C sequestration rate of 1.01 Mg C /ha/y. The accuracy of this rate depends on the amount of C initially in the soil and has a finite time range which is reached when the sequestration potential of the soil is reached. Therefore this rate is not fixed but varies depending on conditions and region. There are other benefits that are not directly related to GHG mitigation but are important for soil health and have an environmental impact. For example, warm-season grasses (WSG) for biomass production as an alternative to corn residue removal (which is removed for biofuel) on marginal land in Nebraska showed improvement in wind and water erosion of topsoil and improvement in plant-available water (Blanco-Canqui et al., 2017).

Some examples of specific crops from the literature search are given below. These studies indicate that there are spatial and temporal differences in C sequestration by biomass crops that should be taken into account.

<u>Studies that calculated net ecosystem C balance (NECB) or GHG balance:</u>

NECB accounts for CO_2 flux, the C input from biomass and manure, and C output from harvest but does not account for N_2O emissions. GHG balance is NECB plus N_2O emission flux.

Sulaiman et al., (2017) compared NECB and GHG balance of corn to perennial hay (timothy grass and alfalfa mixture) in side-by-side silt loam soil fields in Ontario. The hay was established about four years prior to the start of the experiment, and was cut once in the first year of the study period and twice per year in the following two years. They found that N_2O emissions were smaller by about 6 times in hay than in corn. The exception being in the year when the hay is plowed down at which time N_2O emissions from hay became only about 1.5 times less than that from corn. Further, it was found that the GHG balance (less is better) was consistently greater in corn than in the hay system. Both were net producers of CO_{2e} but hay (1.3 Mg CO_{2e}/ha/y) produced less than corn (8.0 Mg CO_{2e}/ha/y). NECB was negative (C sink) in only one out of three years for both corn and hay but hay was a bigger C sink. The 3-year average NECB was +0.07 ± 0.5 Mg C/ha/y for hay and +1.5 ± 0.79 Mg C/ha/y for corn, indicating hay was C neutral but corn was a C source. The authors compared their results to other published research and reported that the range of NECB for *hay* was –3.2 to + 1.8 (– is a C sink) with an average of –0.47 Mg C/ha/y. This study does not account for emissions from equipment use.

Carbon dioxide emissions from a switchgrass crop grown for biomass production as a biofuel feedstock was measured for two years in southern Ontario (Eichelmann et al., 2016b). The non-growing season CO_2 emissions resulted in a total loss of C equal to an average of 1.4 Mg C/ha. Growing season respiration (CO_2) loss was equal to an average of 8.5 Mg C/ha. Because of different condition and growing season lengths in the two years of the experiment, the NECB (assumes biomass removed for bioenergy is released as CO_2 again) was a net source (+1.1 Mg C/ha) and a net sink (–0.6 Mg C/ha) in 2012 and 2013, respectively.

Switchgrass and corn for biomass feedstock production were also compared in a southern Ontario study (Eichelmann et al., 2016a). The switchgrass crop was established about eight years before the experiment and so had already reached full production capacity. Total growing season respiration (CO_2 flux) was smaller for switchgrass (9.6 Mg C/ha) compared to corn (12.2 Mg C/ha). Harvested switchgrass biomass was smaller than that of corn, therefore the amount of C removed from the system with harvest was smaller for switchgrass with an NECB –0.66 ± 0.6 Mg C/ha for switchgrass and +7.0 ± 0.5 Mg C/ha for corn (when both corn grain and stover were removed for biofuel production). As corn and switchgrass have different types of canopies and growing season lengths their evapotranspiration and water balance are different. The authors suggest that because of these differences the two crops have to be compared on a common basis in terms of productivity, ecosystem water use efficiency (EWUE). This was calculated as 13.0 and 13.2 g CO_2 / kg H_2O for switchgrass and corn, respectively. Both have similar ecosystem water use efficiency. These results were reported from one year of field data and should be supplemented with more research.

A similar assessment to that conducted by the Eichelmann et al. studies, was carried out in Pennsylvania (Skinner and Adler, 2010). The switchgrass crop was a net CO_2 sink for the first three years of the experiment (av. NECB = –2.5 Mg CO_{2e}/ha/y) and a net source in the fourth year (1.8 Mg CO_{2e}/ha/y) which was due to the large harvest removed from the system. The C output was greater than C input in the fourth year. Differences between years was also observed in the Eichelmann et al., (2016a,b) studies indicative of the variablility related to crop growth and weather conditions. Again, it is important to note that this type of assessment is from the agroecosystem side and does not include the fuel displacement offset associated with the larger biomass removal. Therefore, comparing these results with life cycle assessment (LCA) results would provide a bigger picture on sinks and sources.

A life-cycle assessment was conducted to assess the GHG mitigation potential of biomass cropping systems including hybrid poplar (HP, harvested every 10 years), reed canarygrass, switchgrass, corn-soybean rotations, and corn-soybean-alfalfa rotations in Pennsylvania (Adler et al., 2007). Alfalfa stems, corn stover, corn grain, reed canarygrass, switchgrass, and HP were used for ethanol production, soybean grain for biodiesel, and reed canarygrass, switchgrass, and HP for electricity from gasification. The study was conducted using the DayCent model to calculate GHG fluxes and simulations conducted for 30 years. All the crops in these systems were found to be GHG sinks. Net GHG (including fossil fuel displacement and all farm related operations) was estimated at –2.6 Mg CO_{2e}-C/ha/y for HP and at –2.1 Mg CO_{2e}-C/ha/y for switchgrass. The other crops were as follows: reed canarygrass (–1.2), corn-soybean (av. –0.7) and corn-soybean-alfalfa (av. –0.5). This estimation was done without the assumption that SOC change has stabilized at a new equilibrium. When the SOC is assumed to reach a new equilibrium in the long term, the net GHG estimates become –1.8 and –1.7 Mg CO_{2e}-C/ha/y for HP and switchgrass respectively. Emission of N_2O was found to be the biggest GHG source in this life-cyle assessment with the average for all the corn rotations of ~3.7 kg N_2O/ha/y, ~3.4 kg N_2O/ha/y for reed canarygrass, and ~2 kg N_2O/ha/y for switchgrass and HP.

A modelling study compared the effect of corn-soybean rotations, *miscanthus* and switchgrass crops, and prairie grass on a silty-loam soil on GHG in Illinois, United States using the DayCent model (Hudiburg et al., 2015). The calculation of GHG balance included N_2O and CO_2 fluxes, as well as above and below ground plant biomass. The results showed a benefit in GHG reduction compared to corn-corn-soybean, emissions summed over 30 years. The total GHG difference (benefit or sink) was –287 Mg CO_{2e}/ha for miscanthus, –132 Mg CO_{2e}/ha for switchgrass, and –100 Mg CO_{2e}/ha for native prairie. The largest benefits were mostly from C stock differences: –242, –99, –58 Mg CO_{2e}/ha for miscanthus, switchgrass, and native prairie, respectively. The benefits from reduction in N_2O efflux were –44, –31, –42 Mg CO_{2e}/ha for miscanthus, switchgrass, and native prairie, respectively and the reduction in NO_3^- leaching were –1.1, –1.2, –1.0 Mg CO_{2e}/ha for miscanthus, switchgrass, and native prairie, respectively.

Smith et al., (2013) compared direct and indirect N losses from corn-corn-soybean (C-C-S) rotation and biomass crops switchgrass, miscanthus, and prairie grass in Illinois, United States in a 4-year field study. The C-C-S was fertilized at about 180 kg N/ha every year and switchgrass was fertilized only in two years (56 kg N/ha). Miscanthus did not establish well in the first year, therefore comparisons were made in the 2nd to 4th years. Nitrate leaching was calculated in the soil at 50 cm and in the tile drain. In the soil, the level of NO_3^- was consistent in the C-C-S rotation at an average of 44 kg N/ha/y, decreased from 30 to 17 to 2 kg N/ha/y in miscanthus in year 2, 3, 4, respectively, and was consistently low in switchgrass (av. 7) and prairie grass (av. 2.8). Similarly NO_3^- in the drain was lower under the biomass crops than from the corn and soybean crops in the 4th year. Therefore, after establishment, the biomass crops performed best in reducing N leaching. The biomass crops were better in reducing N_2O emissions with the 3-year average N_2O emissions at 4.5 kg N/ha/y for corn, 1.1 kg N/ha/y for switchgrass and miscanthus, and 0.6 kg N/ha/y for prairie. A 3-year N balance was calculated including N inputs from fertilizer and mineralization and N losses. The C-C-S rotation balance was 22 kg N/ha/y and that of switchgrass was also positive at 9.7 kg N/ha/y and therefore these were net sources. This was likely due to the addition of fertilizer. In contrast, miscanthus and prairie grass were sinks with N balances of –29 and –18 kg N/ha/y, respectively. These calculated balances are subject to change depending on the input of fertilizer N and the export of N in the harvested matter and therefore should not be generalized or used indiscriminately.

The effect of N fertilization rate on switchgrass and reed canary grass (RCG) yield and GHG emissions was tested in two years in Nova Scotia (Wile et al., 2013). Three rates of N fertilizer were tested: 0, 40, 120 kg N/ha. While the yield of both crops did not show a significant response to N addition, N_2O emissions did. Nitrous oxide emissions were lower from the RCG plots than the switchgrass and N addition, especially at the highest rate, resulted in increased emissions in one year in switchgrass and both years in RCG. At all N rates both crops were CH_4 sinks. On average the total growing season GHG emissions (N_2O and CH_4) in the 1st year were 197, 153, and 432 kg CO_{2e}/ha for switchgrass and 16, 40, 113 kg CO_{2e}/ha for RCG at 0, 40, 120 kg N/ha, respectively. In the 2nd year, the total growing season GHG emissions were 93, 37, and 69 kg CO_{2e}/ha for switchgrass and 2, –9, and 98 kg CO_{2e}/ha for RCG at 0, 40, 120 kg N/ha, respectively. Accounting for biomass yield, the GHG emission intensity ranged between 8 and 60 kg CO_{2e}/tonne switchgrass and between –2 and 20 kg CO_{2e}/tonne RCG. Therefore, addition of fertilizer especially at the highest level of 120 kg N/ha resulted in higher GHG emissions which were not offset by an increase in yield. The balance between yield benefit from fertilization and GHG emissions needs further site-specific testing.

Studies that calculated SOC storage:

In terms of soil C storage, the study by Adler et al., (2007) (detailed above) reported that HP resulted in the biggest gain of SOC estimated at 0.8 Mg C /ha/y (this value is the change from initial soil C), and switchgrass and reed canary grass resulted in an estimated increase of 0.4 Mg C /ha/y. VandenBygaart et al., (2010) reported that replacement of continuous corn with alfalfa in a field in southern Ontario increased SOC by 8 ± 4 Mg C/ha in 25 years (about 0.3 Mg C/ha/y).

Liu et al., (2017) assessed the impact of biomass production on marginal land in Canada that is not currently in crop production (on land classes 4 to 6, which amounts to an area of 328,000 ha in Ontario). The conversion of un-cropped marginal land to biomass production which mostly includes a land use change from unmanaged to perennial production was estimated to result in a loss of stored soil C equivalent to 2.13 million tonnes CO_{2e}/year under switchgrass and 66.56 million tonnes CO_{2e}/year under HP. This was only as a direct effect of land use change. The benefits to GHG mitigation were realized when accounting for the use of the biomass for the replacement of coal and gasoline for energy production. If the land use change was from annual crop to biomass crop there likely would not have been a loss of soil C stock. The authors noted that these numbers should be refined for specific regions in Canada. This study was based on land classes 4-6 which have low productivity or high competition from weeds and therefore, the establishment of biomass crops on these land classes requires a few years before the full potential of the crop is achieved.

Four years after the establishment of switchgrass in an experiment in Michigan, United States the SOC increase rate to 60 cm depth was calculated to be 2.1 Mg C/ha/y for unfertilized crops (Valdez et al., 2017). Jungers et al., (2017) studied the root biomass of different perennial biomass crops in the 0-90 cm of the soil in Minnesota, United States. They noted that switchgrass root biomass was greater in the unfertilized treatment than when N fertilized but other grass species were similar under both fertilization treatments. Switchgrass roots had about 40.5 ± 0.6% C in the roots from the top 30 cm of soil, greater than other species that had a mean of 37.8%. SOC had not changed from the initial measurements in the six to seven years since the start of the experiment. Averaged over fertilized and non-fertilized treatments, the GHG mitigation potential from root biomass C (without accounting for farm management GHG sources and fossil fuel offsets) was –3.6 ± 0.2 Mg CO_{2e}/ha/y for switchgrass, –3.2 ± 0.3 for 4-species grass mix, –2.6 ± 0.2 for grass+legume mix, and –1.8 ± 0.1 for both a 12-species mix and a high diversity 24-species mix.

In Georgia, United States energy cane and elephant grass were grown for four years as biomass crops with and without crimson clover cover at two rates of N fertilization with an unfertilized control (0, 100, 200 kg N/ha) (Sainju et al., 2017). The elephant grass increased the SOC stock more than the energy cane especially when it was combined with the CC and N fertilization at 100 kg N/ha. An increase in N fertilization rate to 200 kg N/ha did not improve SOC. In the top 0-5 cm soil layer, the SOC sequestration rate was estimated at 0.5-1.0 Mg C/ha/y for the elephant grass with crimson clover and 100 kg N/ha fertilization and at 0.1-0.5 Mg C/ha/y for the other treatments.

An environmental modelling assessment of biomass crop production on a conservation land program in Iowa, United States was done by LeDuc et al., (2017). No-Till corn for cellulosic ethanol was compared to switchgrass and a C3/C4 grass mix. Switchgrass was predicted to produce the most cellulosic ethanol per hectare but corn would produce more when both grain and cellulosic ethanol were considered. Soil erosion was greatest under corn and SOC stock was greatest under switchgrass. The simulated buildup of SOC with switchgrass was 13.5 Mg C/ha in 30 years compared to 6.5 Mg C/ha with the C3/C4 grass mixture, a loss of 15.5 Mg C/ha with NT corn. The authors concluded that switchgrass resulted in the best outcome environmentally and on a per liter ethanol basis.

SOC change after conversion to switchgrass from either cropland/pastureland or marginal land was modelled using CENTURY Soil Organic Carbon model which simulates long-term C and N dynamics for soil-plant systems (Emery et al., 2016). Change from cropland to switchgrass resulted in an increase in SOC by 0.13, 0.19, 0.29 t C/ha/y in Iowa, Kansas and Missouri, respectively. Change from cropland to switchgrass on marginal land resulted in similar increases in SOC (0.14, 0.21, 0.27 t C/ha/y) in those regions. However, a change from grassland to switchgrass on marginal land resulted in varying SOC stock changes from a loss of 0.02 to a gain of 0.06 t C/ha/y.

In Bologna Italy, switchgrass and giant reed replaced a 30-year poplar stand and SOC was assessed 10 years after grass establishment (Nocentini and Monti, 2017). The crops were not fertilized with N and were harvested annually. The annualized LCA of bioethanol production was also performed. The authors calculated that switchgrass contributed 9% to the total SOC stock in 10 years. Accounting for varying bulk density, a range of SOC stock increase was estimated at 0.75-2.18 and 0.37-1.58 Mg C/ha/y with giant reed and switchgrass, respectively. The giant reed produced a larger aboveground biomass which is the reason for the bigger SOC contribution. The LCA accounting for biofuel production, showed that there was a reduction of GHGs with both crops of about 2.8 and 1.9 Mg C/ha/y with giant reed and switchgrass, respectively.

The literature review suggests that perennial crops (grasses) for biofuel production or for restoration of degraded land generally increase SOC. Such land conversion can potentially result in GHG mitigation especially if a full C budget is considered. The GHG mitigation is chiefly a net sink of CO_2 when accounting for fuel displacement. The full C budget considers all on-farm sinks and sources from soil and management. There is also evidence that NO_3^- leaching (an indirect source of N_2O emissions) when perennial crops are grown is reduced compared to annual corn and corn rotation systems. For this reason growing perennial/biomass crops was selected as a potential BMP for GHG mitigation.

3.5. Soil management and practices

3.5.1. Tillage

Choice of tillage management affects both N and C dynamics in soils and is related to the amount of organic matter input (removing or keeping residues), the moisture regime, compaction, and soil physical effects. More C input is expected to accumulate in the top soil of NT soils providing a C sequestration potential in NT systems. It should be noted that when the whole plow layer is considered, C concentration might not be much different in CT and NT systems. This is because with CT the top soil, along with the crop residues, is turned under and buried at depths close to the plow layer which means that some amount of residue-C is re-distributed to a deeper soil layer. However, Six et al. (2002) reported from results of a modelling study, which included experiments from tropical and temperate climates, that the net C gain in the surface soil layer under NT seems to offset the C sequestration in deeper soil layers in CT. There are several factors that affect the amount of residue-C that is buried and how much of it is stabilized such as soil type, cropping systems, root depth, frequency and intensity of tillage amongst others.

<u>C sequestration:</u>

Two long-term studies in Ridgetown, Ontario compared the SOC content after moldboard plow (PT), chisel plow (CHT) and NT in a corn-soybean rotation after 15 years. Moldboard plows turnover the soil where chisel plows provide deep tillage with less soil disturbance. The second study examined the effects of CT to NT after 11 years in four types of rotations varying in diversity (Van Eerd et al., 2014). NT had the highest SOC content down to 80 cm. This was higher than with PT and either higher or statistically similar to the CHT. In the second experiment, NT also had greater SOC content (pooled over all the crop rotation plots) than the CT down to 100 cm. In the top 0-10 cm the SOC content was 36 Mg/ha for NT and 29 Mg/ha for CT soils. NT had 36% more SOC content (and concentration) compared to CT in the top 0-5 cm, 26% more in the 0-10 cm, and 16% more in the 0-100 cm profile. The study also reported on the general soil health of the CT and NT systems finding a better CSHA score for the NT including statistically better scores for soil aggregate stability and mineralizable N content in the NT system. These studies point to the benefits of NT in terms of C sequestration potential and improvement to soil health. This then describes another benefit to NT in addition to N_2O emission reduction demonstrated in other studies earlier in this section.

From five long-term studies in eastern Canada, VandenBygaart et al., (2010) reported a range of SOC difference between NT and CT from –2.7 to 2.2 Mg C/ha (positive is a gain in SOC under NT) in the top 30 cm of soils where study durations ranged from 12 to 25 years. On average, there was an increase of 0.22 ± 0.83 Mg C/ha in the top 30 cm of soil from switching from CT to NT (about 0.02 Mg C/ha/y).

Long-term field experiments in Michigan, Ohio and Kentucky in the United States found the difference in SOC resulting from NT and CT in the 0-20 cm soil layer of loamy and fine soils ranged between 2-6 Mg C/ha or 10-20% more C in NT (Six et al., 1999). Yearly sequestration rate, calculated from estimates based on the year of establishment of the treatments, were ~0.1-0.3 Mg C/ha/y. In a regression modelling study using data from humid regions, Six et al. (2004) estimated a rate of sequestration in NT compared to CT equal to 0.22 Mg C/ha/y (0.81 Mg CO_{2e}/ha/y) after 20 years of NT. Accounting for N_2O and CH_4 emissions resulted in a reduction of global warming potential under NT by 0.5 Mg CO_{2e}/ha/y after 10 years of conversion and by 2.1 Mg CO_{2e}/ha/y after 20 years. Therefore, even when/if N_2O emissions are increased in some cases with NT, the total GWP is reduced in the long run primarily as a result of C sequestration.

Puget and Lal (2005) compared C sequestration in corn-soybean systems under chisel plow (CHT), moldboard plow (PT) and NT in Ohio after eight years of establishment. The SOC was similar in the CHT and PT treatments. Differences in C stock were not statistically significant but the trend was for greater C was NT > PT ≥ CHT. The difference between NT and the average of CHT and PT was 19% in the top 0-5 cm layer, 13% in the top 0-10 cm and 9% in the top 0-30 cm. In terms of C stock, NT had 3.0, 4.0, and 8.0 Mg/ha more C in those layers, respectively.

Two studies from Minnesota reported no SOC differences between NT and CT when deeper layers in the profile were considered (Dolan et al., 2006; Venterea et al., 2006). Dolan et al. (2006) compared NT with CHT and PT on corn and soybean fields after 23 years. Although the top 0-15 cm had greater C in NT than in CHT and PT treatments, the soil layer from 25 to 45 cm had more C in the conventional treatments. Considering the whole soil profile down to 45 cm showed that C stocks were similar between NT, CHT, and PT. Venterea et al. (2006) compared NT to CT and to biennial tillage with a chisel plow (BT) performed during the soybean phase of a soybean-corn rotation. After 15 years of the treatment implementation, the SOC stock in the 0-60 cm profile was greatest in BT followed by similar amounts in NT and CT. The advantage of BT seemed to be from providing some tillage that helps to regulate moisture and promote growth compared to NT.

An 18 year study in Michigan reported differences in SOC between CT and NT change over time in a corn-soybean-wheat rotation (Senthilkumar et al., 2009). Over the course of the study, soil under CT lost 1.15 g C/kg whereas NT soil lost 0.71 g C/kg. In the top 15 cm of the soil, the difference between CT (8.2 g C/kg) and NT (9.9 g C/kg) was 21%. Interestingly, a certified organic treatment using CT that included a winter CC had SOC similar to the NT. In a second location, NT soil that had initially less SOC that the NT in the first site gained 0.31 g SOC/kg in 20 years.

The stratification of C in the profile under CT and NT in corn and soybean long-term experiments over three decades is supported by a study in Indiana (Vyn et al., 2006). There was greater C under NT in the soil down to 30 cm but the trend reversed in the 30-50 cm layer where C was greater in CT, however, the soil profile to 100 cm still had 3% more C under NT. There was 9, 17, & 23 Mg C/ha more under NT than CT in the 0-5 cm, 0-15 cm, and 0-30 cm layers.

The above examples show that there are some variations in potential C sequestration and C stratifications in different tillage systems but in general there is a consistent finding of an SOC increase with NT especially in the long term.

N_2O emissions:

For N_2O emissions, the current equations used in the NIR adjust N_2O emissions from agricultural soils in Eastern Canada by a factor (F_{TILL}) of 1.1 for NT management (based on study by Rochette et al. (2008b). This is because more emissions are assumed from reduced tillage soils in Eastern Canada. In our literature survey, we found reports that suggest N_2O emissions from NT soils might not be statistically greater than those from CT and that there are some variable factors that play a role in this determination. For example, when whole year emissions are considered (including spring/winter thaw) versus growing season emissions, NT become either comparable to those from CT or less. Another factor that plays a role in the intensity of emissions from NT systems is precipitation variability. When precipitation coincides with fertilizer application events, more N_2O emissions can be expected from NT than CT systems. At the same time there seems to be a yield advantage under CT which translates to lower yield-scaled emissions in Eastern Canada. There is a need for further research to determine the conditions when there is a yield-scaled increase in N_2O emissions under NT.

A study comparing tillage and crop residue removal in the U.S. Corn Belt at 9 different sites showed that, across sites, the differences in N_2O emissions were not statistically significant but did tend to be higher under reduced tillage (Jin et al., 2014). When residues were removed from the soil surface under reduced or NT, N_2O emissions were lower than when the residue was left on the surface. However, CO_2 was higher under CT soils. When total GWP of all three GHGs (N_2O, CO_2, CH_4) were combined, GWP was numerically but not statistically greater for CT compared to reduced till or NT and not affected by residue removal.

A long-term study in southern Quebec on an artificially-drained clay loam in a corn-soybean rotation showed no advantage in N_2O emission reduction using conservation tillage (ridge-till for five years and NT for 15 years) compared to PT (Pelster et al., 2011; Ziadi et al., 2014). Under conservation tillage there was generally more NO_3^- in the 0-15 cm soil depth especially at spring thaw. However, overall the higher NO_3^- did not result in greater N_2O emission under conservation tillage over the whole rotation. However, there was a yield advantage using PT (10% and 13% for corn and soybean respectively).

In an Ontario over winter study, CT systems had higher N_2O emissions than NT and that was related to temperature and moisture dynamics (Congreves et al. 2017). CT results in more soil freezing in winter and this results in more N_2O emissions associated with this system. NT tends to be more thermally insulated with reduced fluctuations in temperature in winter. Over winter, CT had 1.4 - 6.3 times higher N_2O fluxes than NT. Full year emission averages across years and 3 types of crops were 1.1 and 1.3 kg N/ha/y for NT and CT, respectively. Over winter and thaw, the average was 0.5 and 0.7 kg N/ha/y for NT and CT, respectively. This study also found that, contrary to the IPCC assumption, residue removal from the soil surface caused higher emissions.

In Ontario, a study by Drury et al. (2012) reported that N_2O emissions were lower in zone-tillage (ZT) compared to NT and CT. ZT was prepared in the fall by creating a zone, 21 cm wide by 17 cm deep, where corn is planted in the following spring. Compared to CT averaged over 3 years, N_2O emissions from NT were 16.6% lower (ranging from 5% greater to 41% less) and ZT were 43.9% lower (ranging from 36 to 54% lower) compared to CT. Yield was 4% lower under ZT and 11% lower under NT. ZT still produced 43.5% less N_2O emissions when weighted on yield because the three year differences in yield between CT and NT were similar. It seems that ZT provided better soil conditions (lower bulk density and higher aeration) than CT and NT and the clearing of residue from the corn zone could have also contributed to reducing N_2O emissions by limiting the readily available C source to denitrifiers.

Because of different hydrological properties under CT and NT, NO_3^- leaching and surface runoff might differ between these systems. Daryanto et al. (2017) compared indirect emissions from leaching and runoff of NO_3^- between CT and NT in a meta-analysis. Although the concentration of NO_3^- in runoff was greater in NT than CT, the NO_3^- loads were similar, i.e. there should be no effect of management system on runoff. However, leachate NO_3^- load (but not concentration) was greater in NT compared to CT. The main factor that affects this finding is soil texture. The higher NO_3^- load in NT fields comes from medium and fine textured soils and not from coarse textured soil. Therefore, the authors proposed occasional soil harrowing every about 10 years on NT to help overcome soil compaction, nutrient stratification, and continuous macro-pore formation in fine textured soils. Additionally, other management options on NT such as cover cropping and rotation crops were suggested as they would reduce the NO_3^- concentration in the soil under NT.

From the data that was compiled for this study, there were nine studies that compared growing season N_2O emissions from CT with NT and two studies that compared CT with ZT, from Ontario and Quebec. We conducted a meta-analysis on this data to compare CT with conservation tillage (NT plus ZT). N_2O emissions from reduced tillage were significantly different than CT (with an average reduction of 16%, range –26 to –4 %). Further examination of the data revealed that this finding was influenced by the CT versus ZT comparison (Figure 3.5). Average reduction with ZT was 36% (range of –49 to –19%). We also found a 6% (2-10%) reduction in yield with conservation tillage versus CT, which is not affected by the type of conservation tillage, whether NT or ZT. For studies that had experiments with variable N fertilizer rates including a control, we were able to calculate FIE, and found that tillage has a significant effect on emissions. FIE from CT soils was 0.5-1.1% (12 studies/142 observations) and that from NT soils was 0.1-0.43% (5 studies/76 observations) (ZT studies were not included in the FIE analysis). This analysis did not show a likely publication bias.

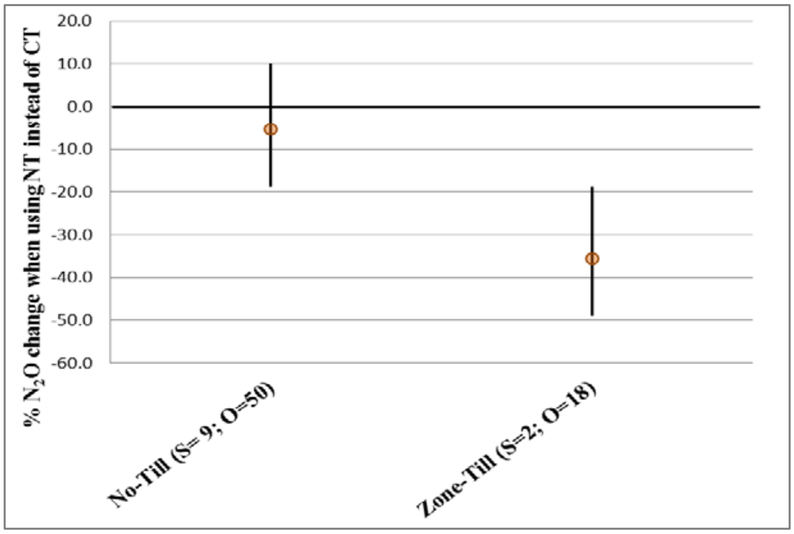

Figure 3.5. Meta-analysis results showing estimated % effect of using no-till (NT) and zone-till (ZT) compared to CT. The numbers following the type are S = number of study and O = number of comparisons.

Reduced tillage, including ZT and NT, was selected as a potential BMP to mitigate GHGs because there is evidence showing increased C storage and reduction in GWP under reduced tillage. Additionally, there is at least no increase in N_2O emission, and possibly a reduction in N_2O emission in winter/spring thaw, from reduced tillage soils. Economical assessment shows additional benefits to reduced tillage intensity through cost reduction and, on well drained soils, no adverse effects on yield. Although the potential for mitigation exists with reduced- or NT it is important to note that occasional tillage is necessary which is to be decided on a site-basis and soil-type basis. There is a relation with precipitation and fertilizer timing which is also to be kept in mind. Therefore, for reduced tillage to be effective it should be tailored to field conditions.

3.5.2. Variable rate fertilization and precision agriculture

Described in Zebarth et al. (2009), there are two ways that site-specific N can be managed in the fields: either as a fixed management-zone based rate or as a variable rate based on the spatial distribution of a measured parameter. The literature is scant on the efficacy of variable rate fertilization (VRF) in reducing the impact of N fertilization on the environment. Some studies were conducted in the United States (see Snyder et al., 2014) and showed the benefits of sensor-based N assessments and in-season N application but also noted that research is still needed and that application of this technology needs both a technical and economical assessment.

An LCA of sensor-based variable N rate on corn in Missouri was conducted by Li et al. (2016) where emissions (and indirect N losses) were estimated using the DNDC model. The use of variable rate N fertilization reduced N application by 11% without decreasing grain corn yield. Reduction of N_2O emissions was predicted to be 10%, in addition to reducing NH_3 volatilization by 23% and NO_3^- leaching by 16%. Total GWP was reduced by 10%. A field study over two years in Ottawa, Ontario was conducted to compare variable rate strategy with uniform N application on grain yield and soil N concentration (Ma et al., 2014). Information gathered from a normalized difference vegetation index (NDVI) handheld sensor was used to determine the N rate for sub-plots in the field for sidedress application at V6-V8 corn stages and this was compared to a uniform application of the same amount of N. Grain yield was found to be similar in both treatments (93 kg N/ha) and similar to the control of 180 kg N/ha applied at pre-plant. Soil mineral N was much higher in the variable rate plots and as such the full environmental impact would not have been reduced in this case with VRF because leaching of the excess N could have occurred. It was noted that the algorithm used to calculate N requirements based on the NDVI needs to be improved.

From the above, it seems that although the potential for the reduction of N application rates does exist with VRF, there is still a need for more research in order to assess the full impacts on production (applicability and economical assessments) and on the environment.

3.5.3. Liming

The effect of pH on N_2O emission is multifaceted. It is thought to play different opposing roles in nitrification and denitrification. Increasing the pH has been reported to increase denitrification and therefore decrease nitrification N_2O losses. pH plays a role in NH_3 volatilization as well with increasing pH increasing volatilization. The production of the enzyme that converts N_2O to N_2 (N_2O reductase) produced by denitrifying bacteria is thought to be limited by low pH. Increasing the pH might facilitate complete denitrification to N_2. Liming to regulate pH of acidic soils also adds an inorganic carbon source to the soil which can play a role in CO_2 emissions. In theory, one mole of CO_2 is produced for every mole of $CaCO_3$ dissolved but in the field there are many variables that control this equation.

It was reported (Thapa et al., 2016) that NIs work well in acidic soils to reduce N_2O emissions (av. 59% reduction if using NI instead of a regular fertilizer).

Liming to raise the pH of soil to ≥6.3 reduced N_2O emissions from pig slurry manure application but increased NH_3. Because of the uncertainty of the response of overall GHG dynamics from pH related manipulations, VanderZaag et al. (2011) reported that liming cannot be considered as a mitigation option before more research is conducted.

3.5.4. *Biochar application*

Biochar affects both N_2O and CO_2 production in soil and also can have an effect on soil C storage. However, the mode of action of biochar that leads to an N_2O effect is not clear yet. Biochar can have an effect on N_2O through its interaction with pH, NO_3^-, organic C, microbial activity, and possibly in other ways. In a lab experiment testing 15 different soil types, Cayuela et al. (2013) found that biochar decreased the ratio of $N_2O/(N_2O + N_2)$ in fine textured soil (i.e. it promoted the complete conversion of NO_3^- to N_2 the last step of denitrification). It was suggested that this effect is related to the ability of biochar to reduce the availability of NO_3^-. The pH and the C:N ratio did not correlate well with N_2O mitigation. The authors presented two hypotheses for the action of biochar that need to be tested further: 1) that biochar acts as a reducing agent and an "electron shuttle" thus competing with NO_3^- as an electron sink therefore resulting in less N denitrified or 2) it might facilitate N_2O reductase activity through its liming effects facilitating the final step in denitrification. There was a link between the buffer capacity of biochar (rather than the biochar pH or its effect on soil pH) and the decrease in N_2O emissions during denitrification. The authors reported that biochars produced at 500^oC through slow pyrolysis decreased N_2O production through denitrification in a short-term lab experiment.

In another lab experiment on Ontario soil, Zhang et al. (2015) tested the application of slow pyrolysis biochar at 200, 400, and 600^oC and activated biochar (activated by CO_2, steam, and raising the temperature to 800^oC). Biochar was added at a rate of 15 Mg/ha, and incubated with soil for 28 days. Biochars at 600^oC and activated biochars did not differ in N_2O emissions from un-amended soil. Cumulative N_2O emissions from biochar produced at 200^oC or 400^oC amended soils were increased by three fold compared to un-amended.

A mesocosm experiment in Ontario studied the effect of biochar on retention of soil inorganic N after freeze-thaw cycles (Zhou et al., 2017). Biochar (applied at a rate of 20 Mg/ha), especially those that are produced at the highest temperature (600^oC), increased N retention after winter and increased plant N uptake. However, the presence of biochar after the freeze-thaw cycle also increased the emission of N_2O.

Two meta-analyses in 2017 reported on the effects of biochar on GHG mitigation. Verhoeven et al. (2017) reported that there is a ~7% increase in yield in upland soils but that effect on N_2O emissions were inconsistent ranging from –18.1 to + 0.8%. He et al. (2017) reported that there is an increase in CO_2 (by 22%, although the majority of studies were from incubations and the field studies did not show the same trend) and a decrease in N_2O emissions (by 31%) with the addition of biochar (no difference for CH_4). However there were moderators: 1) CO_2 was not increased when biochar was added to fertilized fields and this is likely because the added fertilizer (NH_4^+ and NO_3^-) is adsorbed/immobilized by the biochar which leads to a suppression of C mineralization, 2) CO_2 was reduced as the soil was fine textured and 3) CO_2 was reduced when pyrolysis temperature was higher.

In a 99-day pot study in China by Jia et al., (2012), urea or urea+manure were applied at a rate of 400 kg N/ha to vegetables (choy sum and amaranth) and biochar (from corn stalk produced at 400°C) was applied at a rate of 20, 30, or 40 Mg /ha. The results showed that all the biochar amended treatments reduced N_2O emissions with the reduction ranging between 9 and 60% compared to urea application alone.

The effect of adding biochar to manure or to grazed pastures that receive excreta/urine from animals was evaluated in the following studies. One field study by Taghizadeh-Toosi et al., (2011) in New Zealand incorporated to a 10 cm depth (by roto-cultivation) biochar on a ryegrass pasture soil at rates of 15 and 30 t/ha. There was less N_2O produced with the 30 t/ha biochar rate compared to the zero biochar and the 15 t/ha biochar. N_2O emission was 0.15, 0.16, and 0.07% of the urine-N applied for the 0, 15, and 30 t/ha biochar rates. Another study that examined the effect of biochar when mixed with anaerobically digested cattle slurry during a 55-day incubation (Bruun et al., 2011). Wheat straw biochar produced at 525°C was added to soil at a rate of 1% and 3% (roughly equal to 20 and 60 t/ha assuming a bulk density of 1.4 g/cm^3 to a soil depth of 15 cm) and slurry was added to give 150 kg N/ha and water added to a water-filled pore space of 40% or 80%. The results showed that N_2O emissions were reduced by about 82% in the 3% biochar treatment that had 80% water-filled pore space compared to the non-amended slurry but that the 1% biochar rate and the 3% at 40% water were not different from non-amended slurry. This study shows that the potential effect of biochar on emissions from liquid manure should be examined further.

In general, biochar seems like it could provide some benefits to soil and could possibly reduce N_2O emissions but more research is needed on the types of biochars, their production temperatures, application rate and the repetition of application over time, specifically for the different soil types and conditions in Ontario. The effect of different biochar types (and pyrolysis temperature) on NH_4^+ concentration in soil and NO_3^- leaching from soil should also be assessed because biochar could bind NH_4^+ which affects N cycling and direct/indirect N_2O emission. The production process of biochar is a source of GHG emission as well. However, a review by Qambrani et al. (2017) reports that the net LCA of biochar is a net sink of GHGs.

3.6. Manure management

The manure source for most of the studies reviewed was liquid manure (dairy or swine) with only a few examining solid dairy manure and poultry manure application.

3.6.1. Spring vs fall application

In Ontario, Cambareri et al. (2017a) reported on N_2O emissions and yield from fall and spring applied liquid dairy manure (target rate 150 kg N/ha) over three years. Emissions from fall application were lower than spring application only in one out of three years when precipitation was low in winter and spring. There was no difference in emissions between application times when precipitation was normal. The intensity of NH_4^+ and NO_3^- (intensity here is a criterion defining the exposure of the soil to NH_4^+ and NO_3^-) was in general found to be higher in the field where manure was applied in the fall compared to spring-applied suggesting possible leaching losses from fall application of NO_3^-. Similarly, Abalos et al. (2016a) reported no difference in N_2O emissions from fall and spring applied liquid dairy manure when quantified over the full year. Although the spring application avoided the peak emission from the spring thaw, there were larger peaks associated with rainfall events in the spring. Therefore, no consensus can be drawn from the literature review to support spring application of manure over fall application as a BMP. However, a full assessment of leaching and volatilization as well as N_2O emissions is still needed before a final verdict on this matter can be made.

3.6.2. Application method of manure

Broadcasting and incorporation of liquid dairy manure produced less N_2O emissions than injection of manure and numerically but not statistically less N_2O emissions than broadcasting without incorporation (Cambareri et al., 2017a). Yield was in general greater when the manure was injected. The authors noted that injection could potentially produce yields with low N_2O emission intensity but for current practices broadcasting and coverage was the best option for GHG mitigation. Similar results were reported by Abalos et al. (2016a) where in two out of three years injection (av. 2.5 kg N/ha) caused greater N_2O emissions than broadcasting and incorporation (2.0 kg N/ha) of liquid dairy manure (yield scaled emissions were similar on average). Injection produced less N_2O emissions in the year when conditions were dry. Additionally, 30-60% of broadcast manure is lost as volatilized NH_3 (Meisinger and Jokela, 2000), and incorporation or coverage of surface applied liquid manure reduces volatilization (Rochette et al., 2001). While the injection of liquid manure reduces volatilization considerably, it likely results in increased N_2O loss with the possible exception being when the liquid manure is injected into dry soils which does not cause increase in N_2O (references in (VanderZaag et al., 2011)). Injection on clay soils are especially prone to increased NH_3 loss because injection slots can seal the surface of wet clay soil and delay infiltration. Comparing the trade-off between N_2O emissions and NH_3 volatilization from injection of liquid manure Wulf et al. (2002) found that N_2O emissions after injection might be as high, and in some cases even higher, than volatilization and concluded that immediate coverage of surface applied manure was the best option for controlling emissions. On pasture and perennials where harrowing or incorporation might not be an option, using drag-hose and the addition of trailing shoe instead of broadcasting might be a better option to reduce NH_3 volatilization. An exception might be on clay soils where using a trailing shoe can cause surface sealing of soil thus preventing good infiltration. Based on the available literature, immediate coverage of manure is the only application method that more consistently showed reduction in N losses. Quantification of amount of N loss reduction with this method was not possible from the available data because these lack information on both N leaching and volatilization which are required for a good assessment.

3.6.3. Anaerobically digested manure and composted manure

The focus of this report is GHG soil emissions however there are other storage-related GHG benefits to manure treatments that could add to their value as BMPs. Anaerobic digestion of manure, which produces biogas, produces a liquid digestate that is applied to soil as a fertilizer. It has less carbon than untreated manure and a narrower C:N ratio. A review by VanderZaag et al. (2011) noted that although there are consistent reports from experiments that application of digestate reduces NH_3 and N_2O emissions, most of these measurements were made a few weeks following application. Therefore, further research is needed to make better estimates of losses from this application by collecting data on long-term measurements and measurements done immediately after application. A recent study on Ontario loam soil compared N_2O emissions from anaerobically digested dairy manure and raw dairy manure that were injected, broadcast, or broadcast and incorporated in the fall (Cambareri et al., 2017b). It was found that digestate produced lower N_2O emissions compared to raw manure only when it was injected (2.5 kg N/ha) but not when it was broadcast (6.4 kg N/ha) or broadcast and incorporated (5.4 kg N/ha). The lower emissions from injected digestate may be related to its properties such as low viscosity which allow for better infiltration. As digestate use was associated with relatively high emissions compared to raw manure across application methods, the authors proposed that injection of digestate might be a good mitigation practice, at least for this type of soil. Further research is needed to assess the full impact of digested manure on GHG emissions.

Composted manure product is applied to soil as a fertilizer. Composted manure is expected to have less reactive N and less available C and a larger C:N ratio than untreated liquid slurry. Therefore, it is expected that less soil denitrification occurs after application of composted manure than raw manure. Not many studies were found that compared GHG emissions from composted versus raw manure in the field. A modelling study calculated LCA analysis of a farm manure separator and composter on an Ontario farm and emissions from soil were modelled using the DNDC model (Guest et al., 2017). A comparison between raw manure and composted manure was made for alfalfa production for dry hay and for haylage. Different amounts of liquid and composted manure were used to provide the same amount of N to the crops. The modelled N_2O emission factors were 39% and 45% lower for the composted manure in dry hay production and haylage production, respectively.

An Ontario study compared the non-growing season N_2O emissions after fall application of liquid swine manure (LSM) and composted LSM over two years (Kariyapperuma et al., 2012). On a dry matter basis, the LSM had ca.160 mg N/g (117 mg NH_4^+/g) and the composted LSM had ca. 21 mg N/g (0.7 mg NH_4^+/g). The results varied by year depending on winter conditions related to the extent of soil freezing. The plots received 70 kg N/ha in manure in the fall and 84 kg N/ha mineral fertilizer at planting. The first winter of the experiment was colder and experienced 115 days of soil freezing compared to 73 days in the second winter. The N_2O emissions were only significantly different between LSM and composted LSM in the first year. Differences were related to two flux events that were larger in the LSM compared to the composted LSM, the first was after manure application in the fall of the first year and the second was during spring thaw of the first year. The cumulative flux between February and April of the first year was about 57% less from the composted manure plot (0.53 kg N/ha) than the LSM plot (1.23 kg N/ha). Warmer conditions before the spring in the second year likely contributed to the loss of mineral N (through leaching or volatilization) before the thaw event which resulted in similar N_2O fluxes between treatments. Indirect N_2O emissions would then necessarily be different between the two treatments but these were not measured. The overall non-growing season N_2O emissions (November to April) averaged for two years were 0.62 kg N/ha for the control (no manure, only inorganic fertilizer), 1.41 kg N/ha for the LSM, and 0.98 kg N/ha for the composted LSM.

A study in Alberta compared soil denitrification as affected by long-term application of composted versus feedlot manure (both solid manure containing straw bedding) (Miller et al., 2012). Manure was applied in the fall for 11 years starting in 1998 in barley crop fields. Denitrification was measured in the growing season on soil cores every two weeks for four years. Denitrification in the composted manure was found to be statistically similar to the no-manure control. Compared to the feedlot manure (3.2*, 5.1*, 8.6, and 10.5 kg N/ha), the composted manure (0.7, 1.4, 2.6, and 4.9 kg N/ha) had significantly less cumulative denitrification losses in two of four years (noting that this is only denitrification N_2O-N loss not accounting for nitrification and other losses; * is significantly different between treatments). Many factors related to manure type and soil conditions/heterogeneity could have affected the inconsistency between years especially the variability in water content and anaerobic conditions between sampling times/years.

Some options for reducing emissions from manure application include addition of compost or biochar to the soil-applied manure or addition of biochar as a bulking agent during the composting of manure. Such methods are relatively new and only a few studies were found that compare emissions from these treatments. Examples include the following: A study in Australia (Dalal et al., 2010) compared feedlot manure (6-12 month stockpiled beef manure) and manure + green waste compost (composted for 12 months). Cumulative over a year, the feedlot manure (10 t/ha; 187 kg N/ha) produced 5.1 ± 0.3 kg N_2O-N/ha and the manure + compost (10 + 10 t/ha; 267 kg N/ha) produced 4.3 ± 0.1 kg N_2O-N/ha. For comparison, urea (150 kg N/ha) produced 5.0 ± 0.3 kg N_2O-N/ha. The most likely reason for the reduction in emissions is that the soil mineral N was less under the manure+compost treatment. The addition of biochar (hardwood sawdust + rice hull biochar) during the composting of chicken manure was compared to untreated chicken manure in a study by Yuan et al., (2017). A 120-day microcosm incubation compared CO_2 and N_2O emissions from soils. Cumulative N_2O emissions from the untreated manure treatment was 6.7 mg N/g N added and 5.0 mg N/g N added from the biochar-composted chicken manure. Cumulative CO_2 emission from the untreated manure treatment was 1.0 g C/g C added and

0.7 g C/g C added from the biochar-composted chicken manure. In a 55-day incubation study, wheat straw biochar (525°C) was added to anaerobically digested cattle slurry and incubated with soil (Bruun et al., 2011). Compared to anaerobically digested cattle slurry alone, the addition of biochar increased the CO_2 emissions and did not significantly affect N_2O emissions. The results from this experiment are not conclusive because it was not possible to calculate the emissions in terms of the different amounts of C and N that were added with the slurry + biochar, and there was no crops grown. Therefore the use of biochar and other organic amendments during composting of manure or with the manure in the field are options to be explored further with future research.

3.7. Afforestation and tree barriers

Afforestation is defined by IPCC as the planting of trees on land that, historically, did not have a forest. Other definitions include that of the United Nations Framework Convention on Climate Change (UNFCCC; Kyoto Protocol) as “the human-induced conversion to forest of land that has been non-forested for at least 50 years at the time of conversion”. This can include retired cropland, marginal or degraded land. The 2010 projected C stock in Ontario’s forests and wood products, modelled using FORest CARBon Budget Model adapted for Ontario (FORCARB-ON), is 6399 million tonnes equivalent to 23463 million tonnes of CO_{2e} (Colombo et al., 2007). Guo and Gifford (2002) studied the effect of land use change on soil C stocks in a meta-analysis from 74 publications. They reported that conversion of crop land to a secondary forest or a managed plantation has potential SOC gain over the long-term by 38-65% for a forest and 10-30% for a plantation. Afforestation on pastureland or natural grasslands results in a net loss of C. It was noted that because the available data is not large enough to be conclusive, the results are to be considered as a working hypothesis for more targeted studies. In another meta-analysis, Laganiere et al. (2010) reported that in temperate climates the potential for C sequestration from afforestation is in the range of –5 to +20% (av. +7%; results from 49 comparisons). It was found that clay soils (with clay >33%) had the biggest potential for C sequestration and that broadleaf (excluding eucalyptus) trees also offer the highest SOC stock increase of on average 25%. Coniferous plantations are usually found to either produce less SOC stock or even to have a SOC loss after afforestation. However, when accounting for the organic layer and considering long-term sequestration, coniferous trees are shown to sequester SOC.

Eagle et al. (2012) reported results from the United States from four studies including windbreaks, riparian buffers and alley cropping and from 35 comparisons on woody SRCs. Soil C sequestration ranged between 0.8-6.9 Mg CO_{2e}/ha/y (0.2-1.9 Mg C/ha/y) for agroforestry and equal to an average of 2.5 (–7 to +13) Mg CO_{2e}/ha/y (~0.7 Mg C/ha/y) for SRC. Additionally there was an estimate of 0.8 Mg CO_{2e}/ha/y (~2.6 kg N_2O/ha/y) from N_2O emission reduction.

Two studies in Quebec on short-rotation intensive cultures and intercropping reported on afforestation with willow and HP. The management of the willow cultures were based on 3-4 year cycles after the trees are coppiced. The study by Lafleur et al. (2015) measured C storage in the 0-10 cm soil layer on five sites two to six years after establishment and compared them to adjacent reference fields that are either agricultural or abandoned fields. Percent change between the concentration of organic C in the willow fields and the reference fields ranged from 0 to 40% greater in the willow with an average of 25%. Carbon stock differences were smaller and ranged from 3.7 to 60% with an average of 25%. The soil C accumulation rate in the top soil ranged from 0.4 to 4.5 Mg C/ha/y. In this case standing biomass is not considered because the trees are harvested in short cycles. Fine root production was measured in the top 20 cm of the soil at these sites and contributed about 0.6 to 1.2 Mg C/ha/y. The greatest potential C increase was found in the soils with higher sand and lower clay contents because of the initially low SOC concentrations in their reference soils.

The intercropping study took place in southern Quebec (Winans et al., 2015) and compared HP grid plantation and HP intercropping with hay, grain corn, and hay production. The HP plantation had 1,111 trees/ha and the HP-hay intercropping had 111 trees/ha. Carbon sequestration potential was calculated based on assumptions of yield of the different crops and HP (based on a rotation age of 20 years) from local and published information. The assumption was made that 12-20% of the C input to the soil is integrated into the stable SOC pool. The potential for C sequestration in soil was then estimated at 1.85-4.72 Mg C/ha/y for HP plantation, and at 0.7-1.38 for HP-hay intercropping compared to 0.55-1.31 for corn and 0.28-0.72 Mg C/ha/y for hay systems. Economic return was best for corn production and the authors noted that economic valuation of the C sequestration potential of HP would make the system more attractive for producers.

In Ontario, Peichl et al. (2006) compared a sole barley crop with intercropping systems of Norway spruce or HP. Trees were intercropped with barley at 111 trees/ha (13 years after establishment of the system). Above and below ground C in trees, soil C, soil respiration and C leaching were determined. In 13 years, HP had sequestered more than twice as much C in biomass than spruce. There was about 20% more C in the soil in the HP-barley system than the sole barley or the spruce-barley systems. The SOC stock in the 0-20 cm soil layer was 65, 66, and 78.5 Mg C/ha for barley, spruce-barley, and HP-barley, respectively. Total net C flux into each system (not including SOC pool) were estimated as +13.2, +1.1, and –2.9 Mg C/ha/y for HP-barley, spruce-barley, and barley, respectively.

Also in Ontario, Wotherspoon (2014) assessed intercropping with different tree species and found that the carbon sequestration potential (in biomass C stock only) of the fast growing trees (HP) is greater than slow-growing (e.g. spruce). Soil organic C difference from soybean sole crop (71 Mg C/ha) after 25 years in the 0-40 cm soil layer was +16 Mg C/ha for HP, +13 Mg C/ha for oak, +12 Mg C/ha for cedar, +7 Mg C/ha for spruce and +6 Mg C/ha for walnut. Accounting for all crop-C inputs and outputs from these systems (not considering SOC), the total net C balance after 25 years was best for the intercropping systems with spruce (2.37 Mg C/ha/y), poplar (2.07 Mg C/ha/y), Oak (1.63 Mg C/ha/y), Cedar (1.35 Mg C/ha/y), walnut (1.10 Mg C/ha/y), all better than the comparison with sole soybean (–1.4 Mg C/ha/y) which was a depletion of C stocks in the same time frame.

Niu and Duiker (2006) modelled the SOC accumulation in marginal land under four scenarios of coniferous and deciduous forest types and two management practices short-rotation versus permanent forest in the Midwestern United States. The C sequestration capacity (SOC and biomass) was predicted to be 78 and 82 Mg C/ha over 20 years for deciduous and coniferous forests trees, respectively. Over 50 years, averages were 157 and 162 Mg C/ha for deciduous and coniferous trees, respectively. As an average of C sequestration rate in the three states that more closely resemble Ontario conditions, specifically Michigan, Ohio, and Pennsylvania, the rate was estimated as 3.9 and 3.7 Mg C/ha/y for coniferous and deciduous forests for 20 years, respectively. From the 50-year simulations, the rate was similar between coniferous and deciduous trees estimated at 3.1 and 3.0 Mg C/ha/y in permanent and short-rotation forests, respectively (short-rotation are assumed to be harvested every 20 years). Carbon stock in SOC were similar under deciduous and conifer trees and estimated at ca. 0.6 Mg C/ha/y and 0.4 Mg C/ha/y for 20 and 50 year simulations, respectively. Morris et al., (2007) compared soil C stocks in coniferous (red and white pine) and deciduous (maple, oak, tulip poplar) afforested land 50-60 years after establishment to adjacent native forest and agricultural soils in Michigan, United States. Compared to the agricultural soil, the soil C storage was calculated as 18 Mg C/ha (0.35 Mg C/ha/y) in the deciduous afforested area over 53 years and 13 Mg C/ha (0.26 Mg C/ha/y) in the coniferous afforested area after 50 years. Total ecosystem C storage including aboveground, belowground, and litter C, was estimated at 2.4 and 2.5 Mg C/ha/y for deciduous and coniferous afforestation sites, respectively.

Studies of shelterbelts were few and mainly from Alberta and Saskatchewan. For example, Amadi et al. (2016) reported a substantial potential of shelterbelts to increase C sequestration (although increased CO_2 compared to cropland), reduce N_2O emissions and increase CH_4 oxidation (C sink). Soil OC accumulation compared to adjacent cropland was estimated at 0.7 to 1.5 Mg C/ha/y. The average emission from the shelterbelts was 4.1 Mg CO_2-C /ha/y compared to 2.1 for adjacent cropland whereas N_2O emissions were greater in the cropland (2.5 kg N_2O-N/ha/y) than the shelterbelt (0.65 kg N_2O-N/ha/y) likely as a result of fertilization. The total mitigation of N_2O+CH_4 emissions in shelterbelts compared to cropland was estimated at 0.55 Mg CO_{2e}/ha/y.

The literature consistently shows potential for C sequestration, and in some cases GHG mitigation, from afforestation and intercropping. The land availability for afforestation and appeal of intercropping to farmers is a factor that plays a role in these BMPs. That said, the potential for mitigation has been consistently demonstrated and warrants the selection of this category as a potential BMP for GHG mitigation.

3.8. Summary of literature review

Results from the literature review are summarized in Table 3.1 and in appendix Table 3A which shows the conditions where the values apply. Note that Table 3A summarizes all practices reviewed in this section, while Table 3.1 summarizes selected BMPs for which a range and/or mean for GHG reduction could be derived based on the minimum requirements outlined in the methodology. The values given in Table 3.1 are ranges collected from published literature and are the results from specific experiments that are usually related to specific conditions and specific soil types. Therefore, the below numbers can be used to get an idea of possible reduction ranges within the context of Ontario conditions. More details about the conditions where the values were generated can found in the detailed discussion below or by referring to the published articles.

Table 3.1 Summary of GHG mitigation ranges from the literature (conditions from which these ranges were acquired are given in the Appendix Table 3A). Where possible median or mean values are given. Note that N fertilizer type, manure N, crop rotations and diversification, variable rate fertilization and precision agriculture, liming, biochar application, spring vs fall application and anaerobically digested manure and composted manure are not listed below due to lack of data. The impact of these practices is summarized in Table 3A.

Management	Literature review calculated/acquired ranges	Notes
Match N rate to crop demand	0.1 to 0.5 kg N_2O-N/ha	N application rate versus N_2O emissions is linear or exponential i.e. more application beyond crop potential/needs will lead to increased specific (yield weighted) emission For comparison: IPCC Tier I applies a 1% N_2O emission factor to fertilizer lost as N_2O-N
Sidedress (spring application after crop emergence) vs planting	<u>N_2O emission change</u>: +68 to –38%	Needs further research. Effect of sidedress N application on yield is variable
Split N application (split into at least two spring applications)	<u>N_2O emission change</u>: • Split vs sidedress +13 to +52% • Split vs at planting +0 to –26%	Still to be considered: increased energy use with split application Further assessment of effect on yield Interaction with rain and temperature
N placement	<u>N_2O emission change</u>: • Broadcasting vs. injection or banding –25 to –33%	Question of trade-offs between N_2O and NH_3 Broadcasting followed by incorporation reduces NH_3 loss Banding will likely benefit yield and decrease NH_3 if subsurface-banded Injection of liquid N on swelling clay soils risks injection slots remain open and increase NH_3 volatilization Yield increase possibility with injection

		Injection and injection depth needs further research
Inhibitors	<u>N_2O emission change</u>: • NI+UI: range –12 to –83%; mean +2.1% with confidence interval of -10 to +16% (from Figure 3.4) • NI: range –26 to –56% mean -37.5% with confidence interval of -45.6 to -28.2% (from Figure 3.4) • UI +2 to –42%	Potential of UI to reduce emission is variable Results from NI are more consistent but the range is wide Double inhibitor might work to reduce both N_2O and NH_3 especially on alkaline soils More research could help increase certainty
Polymer coated urea	<u>N_2O emission change</u>: +8 to –28% for N_2O +7 to –18% yield-scaled N_2O	Inconsistent results about PCU
Cover crops	<u>SOC change:</u> Range –0.02 to +0.88 Mg C/ha/y with average +0.37 Mg C/ha/y <u>Indirect N_2O reduction:</u> –14 to –53% NO_3^- reduction roughly equivalent to indirect N_2O saving of 0.02 kg N_2O-N/ha/y	Research is still needed especially for effect of CC on N_2O Important to consider full benefit & account for all C and N fluxes Plowing or incorporation of CC results in emission peak i.e. might be affected by tillage system
Inclusion of long-term perennial and biomass crops	<u>SOC change</u> (switchgrass, grass mixes, pasture, giant reed): –1.8 to +2.2 Mg C/ha/y; average of 0.6 Mg C/ha/y and median and median is 0.4 Mg C/ha/y <u>GHG Fluxes</u>: –2.6 to +1.1 Mg C/ha/y (Net ecosystem C balance); average –0.76 Mg C/ha/y and median of –0.80 Mg C/ha/y –2 to –3.4 kg N_2O-N/ha/y (compared to annual crop)	Consistent increase in SOC stock after conversion from cropland to perennial/biomass crops Net flux accounts for GHG emission and C input from primary production (definition might vary depending on study)
Tillage	<u>C sequestration change for NT vs CT (depending on soil depth)</u>: Range of -0.6 to +1.1 Mg C/ha/y and mean of 0.44 Mg C/ha/y; +3% to +36% more SOC with average of 17.3% <u>N_2O fluxes from reduced tillage:</u>	C storage under NT could depend on the depth of soil layer assessed Intensity and frequency of tillage seems to have an effect on C storage and GHG emissions from reduced tillage systems Full-year N_2O vs growing season suggest similar emissions from till and NT

	NT: mean reduction of –5.4% with confidence interval of +10 to –18.7% ZT: mean reduction –35.6% and confidence interval is –18.8 to –48.9%	Yield is better under tilled system i.e. yield-scaled emissions are smaller
Afforestation	<u>SOC sequestration with afforestation:</u> Range –5 to +65% and +0.2 to +1.9 Mg C/ha/y; average of 0.7 Mg C/ha/y and median of 0.5 Mg C/ha/y <u>SOC sequestration with SRC:</u> Range +0.4 to +4.7 Mg C/ha/y; average of 2.2 Mg C/ha/y and median of 2.7 Mg C/ha/y <u>SOC sequestration with intercropping:</u> Range +0.08 to +1.4 Mg C/ha/y; average of 0.6 Mg C/ha/y and median of 0.5 Mg C/ha/y	Research is mostly from United States studies and shelterbelt studies from western Canada Questions remain about reversal after termination, soil C saturation and project timeframes

4. Cost Effectiveness of Mitigation BMPs

The intended outcome of this section is that it will inform any future needs assessments and program design. This section focuses on the BMPs identified in the previous section as the most promising in terms of their potential to reduce GHG emissions and assesses their economic costs and benefits. The most promising BMPs identified in the previous section and assessed here are: N-rate optimization to match crop need, N-placement and timing, mineral fertilizer vs manure N, use of inhibitors, polymer coating, and slow release fertilizers N application rate, cover crops, crop rotation and diversification, tillage, biochar application, and afforestation. N fertilizer type, liming and manure management (spring vs fall application, application method of manure and anaerobically digested manure and composted manure) are not included since they did not show clear evidence of GHG emission reduction. Available evidence for variable rate fertilization and precision agriculture is included under the section of 'N-rate optimization to match crop need' as there was not enough literature results to have a separate section for this topic. Studies on the cost-effectiveness of 'Inclusion of long-term perennial and biomass crops' were few and hence, this topic is not include here even though this is a promising BMP.

The aim of the section is to provide available estimates of the costs GHG mitigation for a BMP or BMP suite under different soil or climate conditions. If quantification is not possible, then an attempt is made to compare qualitatively cost-benefit and/or cost-effectiveness of a BMP compared to other BMPs. The remainder of the chapter provides an overview of the available estimates of the economic costs and co-benefits of soil GHG mitigating BMPs. Finally, it identifies some potential constraints to the adoption of the assessed BMPs.

4.1. Nitrogen fertilizer management

4.1.1. N-Rate optimization to match crop need:

Economically inefficient use of N results in higher emissions, lower yields, and higher levels of residual nitrate (Grant et al., 2006). The return on N fertilizer is maximized at the point where marginal revenue from the extra crop produced is equal to the marginal cost of N (Mussell et al., 2015). Deviations from this rate can lead to suboptimal economic performance. While there seems to be little research on the economic benefits of lower application rates, most research suggests that once this optimal level of N is exceeded, yields do not increase while N_2O emissions increase substantially.

Table 4.1 summarizes the costs and benefits associated with N-rate reduction. Current economic models are based on the assumption that the farmers' objective is to maximize the expected net value of yield as a function of fertilizer (Pannell, 2017). Above the profit-maximizing level of N, crop yields do not rise substantially (Grant et al., 2006). Wheat yield in particular is quadratically related to N fertilizer rate (Ma et al., 2010). This implies that an increase in N fertilizer initially causes a large increase in yield, but yields increase at a decreasing rate. Pannell (2017) also finds that, past the economically optimal rate of N, crop yields do not seem to increase substantially. This implies that increases in revenue resulting from higher N rates, when N rates are already relatively high, generally do not cover the additional fertilizer cost. In this case, fertilizer rate reductions are economically beneficial for farmers.

Economically optimal N fertilizer rates are case-specific (Pannell, 2017). In grassland in Southern Ireland, grass productivity was typically enhanced with the application of up to 400 kg/ha of N fertilizer (Kim et al., 2010). While some studies estimate the economically optimal level of N to be around 300-400 kg/ha, other researchers have found this rate to be much lower. In Atlantic Canada, fertilizer N rates predicted to result in maximum economic return ranged from 62 to 101 kg N/ha for barley (Zebarth et al., 2008). In Southwestern Ontario, optimal N rates for corn tend to be higher than this, usually in the range between 100 and 150 kg/ha (Rajsic et al., 2009; Rajsic and Weersink, 2008).

N_2O emissions increase linearly up to an optimal level of N in soil. Moving past the level of N where profits are maximized, N_2O emissions increase exponentially (Grant et al., 2006; Kim et al., 2013; Kim et al., 2010). N_2O emissions sharply increased after exceeding a certain threshold value of N input, around 320 kg N/ha for grassland (Kim et al., 2010), and around 150 Kg/ha for Ontario corn (Rajsic et al., 2016). Fertilizer N induced N_2O emissions in Atlantic Canada were approximately twice as high when fertilizer N rate was increased from 75 to 150 kg N/ha compared with when fertilizer N rate was increased from 0 to 75 kg N/ha (Zebarth et al., 2008).The overuse of N can result in diminishing returns as well as an increase N_2O emissions.

Output price risk may also influence optimal N rates. Increasing crop yield through N application increases the overall yield variability, which increases revenue risk. In that sense, N can be a risk-increasing input. Policies designed to reduce risk will likely increase N fertilizer use overall (Pannell, 2017).

Since farmers tend to apply more than the recommended or profit-maximizing rate of N, reducing N would appear to be a win-win scenario. A reduction in N use would increase on-farm profits and lower N_2O emissions. However, the payoff function for applying nitrogen is flat. For example, any rate between 24 to 88 kg/ha of N gives almost the same level of profit for wheat regardless the type of soil (Pannell, 2017). Rajsic et al. (2009) also found the financial cost to the farmer of over-applying N to be very small. While there is a cost, the benefit of over-application is a large yield boost in good growing conditions. The gain in returns from this ideal year offsets the cost of the extra fertilizer in the other years, thereby causing farmers to apply more than the optimal rate (Rajsic et al., 2009).

The digitalization of agriculture is generating a number of tools for applying the site-specific optimal levels of N. While precision agriculture technologies such as GPS autosteer, which prevent over-application while ensuring complete coverage, have been widely adopted, variable rate application technologies for inputs such as fertilizer have not been adopted to the same level (Mitchell et al., 2018). Pannell (2017) argues that the flat payoff function from fertilizer means the benefits of greater precision in the prediction and application of fertilizer are likely less than the cost of those enabling technologies. Weersink et al. (2018) argue that enhancing the adoption of non-GPS precision agriculture technologies for fertilizer application will require turning the vast amount of new data collected on crop production into manageable and valuable decisions for the farmer.

Table 4.1 Summary of results on the costs and benefits associated with N-rate reduction

Source	Key Findings	Implications
Kim et al. (2013)	- N_2O emissions increase exponentially after an economically optimal level is reached -Decreasing input could reduce costs and have environmental benefits	-N rates should be reduced if they exceed the economically optimal level
Ma et al. (2010)	-Soil mineral N availability, not N fertilization regulates N_2O flux from soils -Higher N_2O emissions in wet soils -Wheat yield response to N fertilizer is quadratic	-Too much N decreases yields
Kim et al. (2010)	- N_2O emissions sharply increase after exceeding a certain threshold value of N input (around 320 kg N/ha for grass) -Grass productivity was typically enhanced with the application of up to 400 kg/ha of N in fertilizer	-Using less N, particularly in dry periods, decreases cost of fertilizer without a significant yield loss
Zebarth et al. (2008)	-Highest economic returns were found between N application rate between 62-101 kg N/ha - N_2O emissions were twice as high when fertilizer N rate was increased from 75 to 150 kg N/ha compared with when fertilizer increased from 0 to 75 kg N/ha	Reducing N rates may have both economic and environmental benefits
Pannell (2017)	-Greater output price risk leads to lower optimal fertilizer rate. However, policies designed to reduce risk for farmers are likely to increase N rates, more risky N applied without bearing full consequences of those increased risks - Precision technologies for imaging standard rates are not likely to have large benefits	-Policies to reduce price-taking risks can increase N application rates - Variable rate technologies may be more too costly
Grant (2006)	- N_2O emissions rose exponentially with fertilizer application rates, particularly when rates exceeded the maximum economic rate of 110 kg N above which crop yields did not rise substantially	-Lower economic maximum N rate than previous sources - N_2O emission rates increase exponentially post optimal level of N fertilizer

4.1.2. N-placement and timing

Table 4.2 summarizes the key finding on the costs and benefits associated with N placement and timing. For corn production in Ontario, most growers apply all their N via spring broadcast and incorporate it prior to planting (McDonald, 2015). However, side-dressing allows a grower to better adjust N rates to yield expectations later in the season (McDonald, 2015). A model developed by Bontems and Thomas (2000) indicated that the value of information and risk premiums account for a significant amount of both fertilizer costs and profit per acre in the U.S. Midwest. The information provided through tools like soil/tissue N tests used to determine more accurate N application rates with side-dressing is worth approximately $14 per acre (Bontems and Thomas, 2000). Furthermore, the amount that a grower would pay to be absolved of risk, including production risk, is $2.50 per acre (Bontems and Thomas, 2000). These values combined make up 20% of profit per acre and about 30% of the fertilizer cost in the U.S. Midwest (Bontems and Thomas, 2000). Thus, there is significant benefit to side-dressing N given the yield and soil information it can provide in addition to the reduction in production risk.

Side-dressing, however, also has its drawbacks (Feinerman et al., 1990). Simpson and Williams (1985) produced a formula for calculating fertilizer application costs related to the equipment used. Variables included annual fixed costs (depreciation, interest, annual taxes and insurance, and annual maintenance and repair) and variable costs (fuel, oil and lubricants, and labour). The capital costs associated with fertilizer application were noted to be substantial. Additionally, Feinerman et al. found that risk-averse growers are more likely to apply more N earlier in the season and less as a side-dress due to concerns that wet conditions may prevent side-dressing operations while risk-neutral growers are more apt to apply more N as a side-dress (1990).This information suggests that growers may be restricted in their ability to adopt new fertilizer application strategies given the upfront cost of new equipment. If they do in fact have access to equipment required for side-dressing, they may still lean towards less intensive, one-time broadcast applications at planting.

The above literature review indicates that there are benefits and drawbacks to N fertilizer application strategies. Split N applications in corn can allow for the adjustment of N rates that better reflect crop requirements part way through the season. The fact that N is applied closer to when the corn crop requires it in large amounts lowers the chance of loss to the environment, which otherwise represents an internal cost to the producer and an external cost to the environment/surrounding community. However, equipment costs and potential weather-related inconveniences of side-dressing are noteworthy issues.

Table 4.2 Summary of results on the costs and benefits of N placement and timing

Source	Key Findings	Implications
Feinerman et al. (1990)	-Risk-averse growers are more likely to apply more N early in the season and less as a side-dress because of concerns with high precipitation levels impeding sidedressing operations -Risk-neutral growers are more apt to apply more N as a side-dress application	- Side-dressing provides a grower the flexibility to adjust N rates based on yield potential - Risk aversion affects whether a grower will side-dress or not
Bontems and Thomas (2000)	-Risk premiums and value of information account for 20 % of profit per acre in U.S. Midwest and about 30 % of the fertilizer cost -The value of information (ie. through N soil/tissue tests performed part way through the season to determine a more accurate sidedress application rate) is substantial (around $14 per acre). -The risk premium is less in magnitude at approximately $2.50 per acre	-Side-dress applications of nitrogen in corn allow growers to adjust N rates based on more accurate yield potential estimates later in the season

4.1.3. Mineral fertilizer vs manure N

There are both benefits and costs in assessing the economic viability of manure fertilizer, compared to mineral fertilizer. The largest benefit of using manure fertilizer its lower cost (Adhikari et al., 2005). However, manure application is associated with other costs such as has high fixed cost, labour costs, transportation costs, and could risk run-off into water resources (Huijsmans et al., 2004; Kaplan et al., 2004).

For manure application to be agronomically efficient, the rate at which it is applied should be no greater than the rate at which crops can assimilate the applied nutrients (Kaplan et al., 2004). There are certain conditions where manure fertilizer is most successful. Timing of manure application matters. The recommendation is to use most manure at the beginning of the growing season (Janzen et al., 1999). Manure management is considered ecologically and economically sustainable if manure is applied to a land area large enough to recycle manure macronutrients effectively.

The effectiveness of manure fertilizer depends on the land capability to recycle nutrients, price of crop, and price of manure products (Janzen et al., 1999). Models assessing the economic viability of manure fertilizer including both fixed and variable costs centred mostly around application time as well as the transportation cost of the manure (de Vos et al. 2006; Huijsmans et al., 2004). When high value crops are considered, benefits of using manure could be economically beyond the boundaries of production unit since revenues from high value crops can justify higher manure transportation costs (de Vos et al., 2006; Adhikari et al., 2005; Janzen et al., 1999). A break-even distribution for manure use on high value crops would permit a break-even distribution distances for some manure products of more than 300 km (Janzen et al., 1999). Other models have also predicted negative impacts on distance and transportation cost of manure to field in the use of manure fertilizer (Kaplan et al., 2004). In addition to transportation costs, labour costs are also associated with manure application. Average additional costs of manure application by a trailing foot or a shallow injector are decreased by 15% on small extensive farms to more than 50% on intensive farms (Huijsmans et al., 2004). Broadcast spreading appears to be the method with the lowest cost, when compared to trailing foot band application and shallow injection, although there is no significant cost change among methods (Huijsmans et al., 2004). Huijsmans et al. (2004) also find that the transportation cost tends to be a major factor determining the profitability of manure application. Differences in lowest cost application methods of manure are explained by the time to pump and transport manure load.

While there is some GHG benefit from the cost-effective use of manure as fertilizer, the major environment benefit is associated with a reduction in nutrient loading. Nationally in the United States, damages from the discharge of manure nutrients into ground and surface waters could approach $830 million (Kaplan et al., 2004).

Table 4.3 Summary of results on the costs associated with manure N use

Source	Key Findings	Implications
Adhikari et al., 2005	-in all cases, found fertilizer cost reduction using manure as opposed to chemical fertilizer	
Janzen et al., (1999)	- Important costs: transportation and processing costs; they limit how far it is profitable to transport manure (up to 300 km for high value crops) - One-time large quantity application most profitable, but this increases risks of run-off - Best conditions depend on land capability to recycle nutrients, crop response to macronutrients, and price of crop	-Timing of manure application matters for economic returns - There is a trade-off between profitability and environmental quality
Huijsmans et al., (2004)	- Major costs: Machine costs affect operating costs, determined by distance from storage, loading time and travel speed (road quality) - Broadcast spreading had a small cost advantage compared to trailing foot band application and shallow injection - Average additional costs of manure application by a trailing foot or a shallow injector decreased by 15% on small extensive farms to more than 50% on intensive farms, when fertiliser value of N is taken into account - Profitability depends on the quantity of manure applied (large quantities more profitable)	-Machine and transportation cost must be considered when evaluating the success and profitability of manure -Broadcast spreading is lowest cost method - Higher application rates reduce the fraction of time making up total cost

Kaplan et al., (2004)	- Costs associated with transportation costs, fixed costs, and additional fixed costs (for soil testing and fertilizer savings) - Land constraints, mean more intensive agriculture, which increases the potential for run-off - As willingness to substitute manure for chemical fertilizer increase, nitrogen leached into ground water may increase	- Transportation costs matter - Intensive agriculture can lead to higher potential run-off

4.1.4. *Use of inhibitors, polymer coating, and slow release fertilizers*

A more efficient and cost-effective fertilizer can play a role in increasing crop yields and can address malnutrition issues, as well as reduce the amount of fertilizer that farmers need to use, resulting in cost savings (Ag Innovation Ontario, 2015). Cost savings are expected from the reduction in fertilizer use, as well as a reduction in compliance costs for environmental regulations. However, little research has been conducted to demonstrate these economic benefits. While the environmental benefits have been explored, the economic implications of nitrification inhibitors, the use of polymer coating, and CRFs have not been examined in detail.

A meta-analysis conducted by Yang et al. (2016) found that the NI DCD consistently increased crop yields by 6.5% while the NI DMPP did not. A similar study by Abalos et al. (2014) found that DCD application resulted in a yield increase of 7.5% when used in comparison to conventional N fertilizers.

Different soil conditions result in varying levels of N inhibitor efficacy. The meta-analysis of 81 peer reviewed publications by Yang et al. (2016), found higher pH levels were associated with higher yields and higher N uptakes. The examined range of pH levels was between six and eight. The same study also conducted a financial analysis and found that the monetary benefit from DCD application outweighed the cost. DCD application resulted in an increase in revenue of \$109.49 USD ha^{-1} based on the mean revenue for a maize farm in the US, which translated into an increase of 6.02%. In contrast, DMPP lowered net returns to the average corn farmer by \$93.82 USD ha^{-1}.

Table 4.4 summarizes the costs associated with controlled-release fertilizers. Controlled-release fertilizer production and distribution costs are significantly higher than those of conventional fertilizers. This difference in cost restricts the use of CRFs to high-value crops, golf courses and private green spaces (Trenkel, 1997). Mixing the coated granules with non-coated granules may lower the cost of CRFs (Trenkel, 1997).

Only one study found thus far includes an economic analysis of ESN controlled-release fertilizers. Walsh and Girma used a partial budget to analyze and compare the financial impacts of fertilization using (1) urea, (2) ESN, and (3) a 50:50 blend of urea and ESN (2016). The authors found a slight revenue advantage was obtained with the urea and ESN blend, and no benefits were shown from applying ESN alone. The study confirms that ESN provides environmental benefits by allowing plants to take up more nutrients, but also identifies the need for caution when adopting ESN for improving financial returns.

Table 4.4 Summary of review on the costs associated with the use of inhibitors, polymer coating, and slow release fertilizers

Source	Key Findings	Implications
Trenkel (1997)	-The high cost associated with CRFs restricts its use to high-value crops and golf courses -Mixing the coated granules with non-coated granules may lower the cost of controlled-release fertilizers	-This BMP could be marketed to producers of high-value crops initially, without the need of a subsidy program
Walsh and Girma (2016)	-Government subsidy programs help reduce the cost of the fertilizers to farmers, thus increasing use -In the US, growers using ESN are eligible for payments through the Environmental Quality Incentives Program (EQIP) - A slight revenue advantage was obtained with the urea and ESN blend, and no benefits were shown from only applying ESN. The study confirms that ESN provides environmental benefits but indicates the need for caution when adopting ESN for improving economic returns	-A government subsidy program does not yet exist in Canada -The creation of one can help producers cover the increased costs to achieve the environmental benefits -The environmental benefits alone may not be large enough to encourage adoption of the BMP

4.2. Crop management

4.2.1. Cover crops

There are numerous economic benefits of CCs including: sustainability with increased yield, increased marketability of cash crop, reduction of fertilizer costs while yield is maintained, reduction in disease and pest cycles and the corresponding decrease in production costs associated with pesticides and fumigation (Morton et al., 2006).

The literature review of the economic benefits associated with CC use suggests that the type of CC used, and cost of inputs can impact the profitability. A study in Texas, United States assessed the profitability of rye and crimson clover CC and found economically viable levels of biomass can be obtained from the CCs (Morton et al., 2006). The level for rye, prior to cotton, was 4897 lbs per acre and for crimson clover, prior to corn, the minimum biomass level was 2680 lbs per acre. Despite the economic and environmental benefits, the initial cost of establishing CCs is high (Morton et al., 2006). The largest cost associated with CCs is the high opportunity cost of income forgone of cash crops. When the benefits of weed suppression and avoided cost of pre-emergence herbicide is taken into account, then a savings of $7.47 per acre can be realized for both crimson clover and rye. In considering planting a winter CC, a farmer will plant if the gain in revenue from planting and managing the CC to achieve a level of biomass is greater than or equal to the cost of the

CC minus savings. In terms of selecting the appropriate CC, there is a trade-off between CCs that produce low quality in large quantities, such as grasses, and those that produce moderate amounts at a higher quality, such as legumes. Establishment costs are also a consideration as the cost associated with legume establishment can be up to 10 times higher than that of grasses (Morton et al., 2006).

According to Schipanski et al., (2014), the financial benefits of CCs have not been publicized to farmers, which means adoption is lower because farmers are only aware of the costs from planting, establishing and removing CCs. Roth et al., (2018) conducted a study in Illinois over a two-year period on a corn and soybean rotation and included one system of fall dominated N application (FCC) and a second of spring dominated N application system (SCC), and quantified the economic returns and noted the environmental benefits provided by CCs. Adoption costs included CC establishment, termination, and resultant cash crop yield change and total cost included establishment costs such as seed cost and planting cost, termination costs such as cost of herbicide as well as the various opportunity costs like changes to equipment. The cost-benefit analysis for the 2014-2015 corn planting found that in the FCC system, the benefits provided by CC use recovered 87.9% of CC adoption costs and 86.1% of total CC costs. CCs in the SCC system recovered 66.2% of CC adoption costs and 33.4% of total CC costs. The costs in SCC were slightly higher because the N load was higher compared to the FCC. In the soybean trial, valuing environmental benefits resulted in CC adoption costs of 83.5% and CC total cost recovery of 57.4%. In the SCC treatments, CC benefits resulted in CC adoption cost recovery of 99.1% and total CC cost recovery of 64.7%.

O'Reilly et al., (2012) assessed the impacts of CCs on various biological indicators and economics of CCs in Bothwell and Ridgetown, Ontario. The profit margins over CC and N fertilizer costs were calculated as total revenues less costs associated with CCs. Revenues were determined based on yield for each plot and the provincial average price of fresh market sweet corn in 2007. The costs were determined by seed costs, cost of hiring a custom applicator to plant the CCs, as well as the cost of herbicide control and application of herbicide for treatments including rye. The various CCs included fall application of the following treatments: no CC, oat, cereal rye, oil seed radish (OSR) and a mixture of OSR and cereal rye.

At both sites, with or without N fertilizer, all CCs had profit margins that were as high as or higher than no CC. At the Bothwell site, oat had a statistically significant increase effect on profit margins by $450 ha-1 compared with no CC. At the Ridgetown site, OSR increased profit margins by $1300 ha-1 and OSR and rye increased profit margins by $760 ha-1 compared with no CCs. Furthermore, N fertilization application increased profit margins by $662 ha-1 at Bothwell and by $902 ha-1 at Ridgetown. Net returns varied less when fertilizer N was applied in combination with CCs as opposed to CCs alone. Depending on treatment, N fertilizer application and other factors, CCs have the potential to increase sweet corn yield.

The potential reduction in N fertilizer use can lead to input cost savings for the cash crop planted after a CC. While the nutrients provided by a given CC may not be in usable form, legume CCs can provide 45-224 kg ha^{-1} of available N for cash crop production, depending on availability of nutrients in the soil (Bergtold et al., 2017). Rigorous management of the cropping system is required if lower input costs are to be achieved because N application rates must be adjusted in accordance with the available legume-fixed N. If the producer does not reduce their N application rate, then the fertilizer benefits of the legume CC will be lost.

There is risk involved with CC adoption because prior to adoption, producers cannot know with certainty whether cash crop yields will increase or decrease, or whether input cost-savings will be realized (Bergtold et al., 2017). The potential for economic gains is dependent on input prices, seed costs, environmental factors as well as management practices. The cost-benefit analysis displays that the environmental benefits alone do not recover 100% of total costs. Producers in the Midwest United States have opted to enroll in cost-sharing programs which provide payments to cover a portion or all the costs associated with CC adoption (Roth et al., 2018). The combination of the payments received through EQIP and the quantified environmental benefits of CC adoption resulted in adoption cost recovery in the range of 186.4 to 212.8% and the recovery of total costs from 84.1to 203.7% (Roth et al., 2018). The creation of a government cost-sharing program, such as EQIP in the US, can provide incentives for producers to adopt the use of CCs. Bergtold et al., (2017) also found that just a 1%increase in the cost of using a CC would decrease probability of adoption by 14%.

Table 4.5 Summary of results on the costs and potential co-benefits associated with CCs

Source	Key Findings	Implications
Gabriel et al. (2013)	-The profitability was lower in the CC system because the increased input costs of tillage and planting operations and seed are greater than the benefits of a reduced N fertilizer inputs	-The authors state that a government run incentive program could help increase the adoption rates. A subsidy program could provide the economic incentive for farmers
Ward (2017)	-The lack of data on the economic effects of management practices is being addressed by a program called the Soil Health Partnership, out of Missouri, United States.	-There is a recognized need for more data to link the environmental benefits to the economic benefits
Roth et al. (2018)	- environmental benefits from CCs covered the majority of costs	- long term economic value of CCs not incorporated and may justify use

4.2.2. *Crop rotation and diversification*

The implementation of more diverse crop rotations has been proven to contribute to greater yield stability, higher yields, and greater profitability in Ontario growing conditions. Using data from a long-term crop rotation and tillage trial in Elora, Ontario, Meyer-Aurich et al. (2006b) found that corn-corn-soybean-wheat (under seeded to red clover) and corn-corn-soybean-wheat rotations provided the greatest net returns relative to other rotations while also generating substantially fewer GHG emissions than corn-corn-soybean-soybean and continuous corn rotations. Net returns were calculated at an average of $140 per hectare per year across tillage systems for a corn-corn-soybean-wheat (under seeded to red clover) rotation, $128.50 per hectare per year for a corn-corn-soybean-wheat rotation, $50.50 per hectare per year for a corn-corn-soybean-soybean rotation, and $47 per hectare per year for continuous corn (Meyer-Aurich et al., 2006b). Another study by Meyer-Aurich et al. (2006a) outlined similar profitability results. Furthermore, yield variability was significantly reduced for rotations with additional crops relative to a continuous corn scheme, especially in a reduced tillage system. Risk averse growers were found to favour more complex rotations and the profitability of more complex rotations was less susceptible to increased energy costs and decreased crop prices (Meyer-Aurich et al., 2006a).

More recent work has shown similar results. Gaudin et al. (2015b) used data collected from the Elora, Ontario, long-term rotation and tillage trial. They also found increased corn and soybean yields when other crops (ex. wheat, alfalfa) were incorporated into the rotation. This was particularly true in reduced tillage systems. Corn and soybean yields were found to be less affected by droughty and other abnormal conditions in diverse cropping systems (Gaudin et al., 2015b). Recent research has been conducted at the long-term tillage and rotation trial in Ridgetown, Ontario. The incorporation of wheat into a corn-soybean rotation was found to significantly increase corn and soybean yields (Gaudin et al., 2015a). For corn, N input requirements were reduced and N use efficiency was significantly augmented with the inclusion of wheat in the crop rotation (Gaudin et al., 2015a).

It is important to emphasize how all the studies mentioned above have put major emphasis on the addition of wheat and forage legumes (ex. alfalfa) to continuous corn and corn-soybean systems. This theme was also highlighted in the conclusions of a paper by Congreves et al. (2015), in which a soil health scoring system was developed for Ontario. It was determined that the inclusion of wheat and alfalfa in particular contributed to higher soil health scores in both Elora and Ridgetown (Congreves et al., 2015). All the work presented here stress the high potential of diverse crop rotations to deliver net economic benefits to producers through greater yields and stabilized income.

Table 4.6. Summary of results for economic potential of crop rotations

Source	Key Findings	Implications
Congreves et al. (2015)	-Outlines the development of the Ontario Soil Health Assessment (OSHA) from the CSHA using data from long-term rotation and tillage trials from across Ontario -At the Elora and Ridgetown trials, more diverse crop rotations with winter wheat or alfalfa generally displayed higher soil health scores	-The integration of wheat or alfalfa into simple rotations can positively influence soil health. This could lead to greater productivity and decreased yield variability
Gaudin et al. (2015a)	-Incorporating wheat into a corn and soybean-based cropping system can increase both soybean and corn yields significantly -Wheat also increases N use efficiency for the following corn crop, especially when red clover is under seeded into the wheat	-The application of wheat into a corn-soybean rotation can increase revenue from corn and soybean while also reducing optimum N rates/input costs in corn
Gaudin et al. (2015b)	-More complex rotations benefit corn and soybean yields relative to continuous corn and corn-soybean systems -The potential for corn yield failure is decreased in complex rotations -Corn and soybean yields are less affected by droughty and other abnormal conditions in diverse cropping systems	-Revenue from corn and soybean crops can be augmented in more diverse crop rotations -Yield and profitability are more consistent in more diverse crop rotations
Meyer-Aurich et al. (2006a)	-Net returns were greatest in crop rotations with wheat -The profitability of more complex rotations (particularly with wheat) was less susceptible to energy and crop price changes -More complex rotations are appealing to risk averse producers	-Diversification within a cropping system can enhance farm profitability while reducing economic risk
Meyer-Aurich et al. (2006b)	-The average net returns from a cropping system are greatest when wheat is added into the rotation -Corn-corn-soybean-wheat (under seeded red clover) and corn-corn-soybean-wheat rotations provide the greatest net returns per hectare	-Including wheat in a corn-soybean crop rotation can increase farm profitability while reducing GHG emissions

4.3. Soil management and practices

4.3.1. Tillage

Before a new tillage practice is adopted on a farm, the economic implications of the tillage scheme should be considered. These implications are summarized in Table 4.7. Our research has grouped the economic implications of tillage practices into three major categories: grain yield and quality, equipment costs and labour, and crop input costs.

The grain yield obtained from a particular tillage system translates into the gross revenue generated. Beyaert et al. (2002) found that corn grain yields in a corn, soybean, and winter wheat rotation on an Ontario loamy sand soil were not significantly different between spring plow, NT, and ZT (3 coulters) treatments over a three-year period. Similar results were found over an 11-year study by Dam et al., (2005) on continuous corn grain yields on a sandy loam soil in Quebec using NT, reduced tillage (fall and spring disc), and CT (fall plow and spring disc) treatments. However, Vetsch et al., (2007) reported that corn grain yields were increased in a corn-soybean rotation when fall ZT and fall strip-tillage (coulter and shank combination) were applied for corn relative to spring field cultivator and NT treatments.

Wetter years tend to result in substantial corn yield decreases in strict NT systems compared to rotational tillage systems (Vetsch et al., 2007). Soybean yields were also slightly higher with a fall chisel plow/spring field cultivator treatment relative to the NT treatment, with wet years causing for significant yield drags in the NT treatment (Vetsch et al., 2007).

It is important to note that the study by Vetsch et al. (2007) was conducted in a clay loam soil in Minnesota. On a silt-loam in Ontario, long-term rotation trials showed greater corn yields for a CT system compared to a NT system in simpler rotations (ex. continuous corn, corn-corn-soybean-soybean) (Munkholm et al., 2013). No yield difference was observed for more complex rotations (Munkholm et al., 2013).

Based on the outcomes of these experiments, the tillage system chosen on a farm does not generally affect grain yield. However, significant yield variability can be seen for NT systems, specifically on fine-textured soils in cool, wet seasons. For example, soybeans planted into corn residue using NT have generally been found to experience yield losses in cool, wet seasons and this is especially true for fine textured soils (Vanhie et al., 2015). Increased yield variability implies higher revenue variability and thus a greater risk to growers when reducing tillage intensity.

Grain quality is another significant factor to consider when choosing a tillage system. Grain moisture is one common measure assessed in the literature. High grain moisture can result in additional drying costs or can even represent reduced revenue if the grain is marketed directly from the field. Beyaert et al. (2002) found that corn grain moisture was significantly reduced for CT relative to NT, with ZT falling between these values. Another study, however, found no significant difference in corn grain moisture for various tillage treatments (Vetsch et al., 2007).

Although experiments assessing the effect of tillage systems on grain yield and quality have been conducted, there is little research that we found that examines the associated equipment/labour or crop input costs alone. A Danish project looking at interest, depreciation, maintenance, fuel consumption, and labour costs for various complements of tillage equipment found that the total of these costs was lowest for NT (approximately € 78 ($120) per hectare) and increased with increasing tillage intensity to full CT (approximately € 150 ($185) per hectare) [1] (Sørensen and Nielsen, 2005). Smaller equipment complements, lower power requirements, and lower labour costs for NT schemes may explain this result.

Aside from the literature reviewed above, crop input and equipment costs were listed as potential sources of cost differences across tillage systems in overall farm economic analyses of various tillage systems. Such studies may be particularly useful since they offer a systems approach to evaluating tillage systems rather than assessing individual cost components. Archer and Reicosky (2009) undertook a 7-year study of tillage system economics for a corn-soybean rotation in Minnesota on clay loam, loam, and silty clay loam soils. Yield, and thus revenue, was fairly constant between the NT, PT, CHT, and five strip-tillage treatments. Despite this and higher grain drying costs for the NT treatment, lower labour and diesel costs made NT significantly more profitable than the PT treatment. Overall, strip-tillage and NT treatments were found to maintain or increase profitability over CHT and PT systems (Archer and Reicosky, 2009). One issue with this study was that identical seed, herbicide, and fertilizer inputs were used for all the treatments. Seed, herbicide, and fertilizer expenses represented around 58% and 68% of the total costs for the PT and NT treatments, respectively. Thus, if any change in these inputs was made to more accurately reflect what would occur in varying tillage systems, the results of this study may have been altered (Archer and Reicosky, 2009). Weersink et al. (1992) assessed the production costs associated with conventional moldboard, chisel plow, strip till, and no-tillage systems in Ontario. They found that herbicide costs were lower for the conventional moldboard and chisel plow scenarios. However, like in the on-farm experiment performed by Sørensen and Nielsen (2005), the case-farm scenarios that utilized a moldboard plow and chisel plow system had greater variable and fixed machinery costs with higher labour requirements (Weersink et al., 1992). Overall, more intensive tillage systems like moldboard and chisel plowing showed an average total farm cost of $869.98 ha^{-1} across farm sizes compared to $791.77 ha^{-1} for less intensive tillage systems like NT and ridge-till (Weersink et al., 1992). Uri (1999) used National Agricultural Statistics Survey/ Economic Research (NASS/ERS) Cropping Practices Survey data from the United States to offer several insights into the economics of tillage practices. CT systems generally showed

[1] Exchange rate of $1.54 (Canadian) for €1 was used.

higher direct and opportunity labour use and costs. In addition, corn and soybean yields were reduced with decreasing tillage intensity on poorly drained soils. Better drained soils showed comparable yields between systems for corn and soybean in rotation, but continuous corn systems showed lower yields with decreased tillage intensity. The relationship between equipment costs and tillage system chosen was difficult to predict using the survey data (Uri, 1999). Finally, a general overview of tillage system economics was offered in a review article by Triplett and Dick (2008). They stated that purchased inputs like seed, fertilizer, and pesticides are often greater for NT, but this was offset by lower labour and equipment costs. Thus, total costs were similar between NT and CT. They put forward that yield is the factor that dictates which system is optimal in a given scenario (Triplett and Dick, 2008). Overall, there are mixed results on the farm-level economic impact of various tillage systems. What can be deduced, though, is that less intensive tillage systems like NT tend to be either neutral or beneficial in their effect on total farm economic viability.

An important consideration when deciding between different tillage systems is the machinery complement that a farm already possesses. The large upfront costs of machinery may make it difficult for a farm operator to obtain the equipment complement required for the adoption of a new tillage system. This is especially true if the operator is risk averse to changing management practices for tillage. Furthermore, a given farm may cover a range of soil types that could each benefit from differing tillage management practices. In this case, farms should choose the tillage system that makes the best economic sense on average.

Table 4.7 Summary of costs and benefits associated with alternative tillage practices

Source	Key Findings	Implications
Archer and Reicosky, (2009)	-No significant difference between grain yields for moldboard plow, chisel plow, NT, and strip tillage systems in a corn, soybean rotation in Minnesota -Strip tillage and NT systems generally showed lower costs relative to moldboard and chisel plow practices	-NT and strip tillage systems can maintain or increase profitability at the farm scale relative to moldboard and chisel plow systems
Beyaert et al., (2002)	-Corn grain yields not significantly different across tillage systems (CT, ZT, NT) for 3 yr. study in Ontario -NT yields are more variable across years	-There is more variability for NT revenues, although average yields are similar to CT
Dam et al., (2005)	-Significantly lower average grain yield in residue treatments compared to no residue for 11 years of continuous corn -No significant grain yield difference across tillage treatments (CT, NT, reduced tillage)	-Average grain yields are similar between NT and CT tillage systems in Southern Quebec
Munkholm et al., (2013)	-Significantly higher corn yields for CT for continuous corn and corn-soybean rotations compared to NT in Elora, Ontario	-Simpler rotations benefit from CT schemes. NT is more feasible (in terms of

	-Yields not significantly different between tillage systems for corn-barley(rc)-oats(rc) rotation	yield) with complex rotations
Sørensen and Nielsen, (2005)	-Costs increased with increasing tillage intensity, with direct seeding showing the lowest equipment and labour costs overall -Ploughing increased costs by up to 81%.	-Equipment and labour costs are significantly lower with NT relative to CT
Triplett and Dick, (2008)	-Purchased inputs are often greater for NT (seed, fertilizer, pesticides) -Labour and equipment costs generally lower for NT relative to tilled production systems -Total costs often similar in both systems -Crop yield drives profitability	- NT makes sense in areas where NT has greater yields than tilled systems. -Moreover, where NT has equal or lower yields, increased labour and machine efficiency may make NT the preferred choice
Uri, (1999)	-Yield effects of CT depend on location and the duration over which CT has been practiced -NT systems generally show lower labour and equipment costs but greater input costs	- The costs, benefits, and overall profitability of various tillage systems are very site and situation specific
(Vanhie et al., 2015)	-Various contemporary factors lead to higher corn residue levels being present in fields in the spring -Cool, wet seasons often cause reductions in soybean yield, especially on fine-textured soils	-Increasing corn residue at soybean planting is reducing the consistency of NT soybean yields
Vetsch et al., (2007)	-ZT and strip tillage for corn increased corn grain yields compared to shallow spring tillage and NT -Strict NT systems did not reduce yield compared to rotational tillage systems every year, but did substantially reduce corn grain yields in some years -NT soybean yield poorer in wetter years (clay loam soil)	-Although rotational tillage can maximize corn and soybean yields, increased economic returns are not consistently obtained on clay loam soils in Minnesota
Weersink et al., (1992)	-Herbicide costs lower for moldboard plow and chisel plow systems -Greater variable and fixed machinery costs for the moldboard plow and chisel plow systems (larger machinery complement) -Lower labour requirements for conservations tillage systems (aside from some scenarios using the chisel plow)	-Conservation tillage systems have a good potential to reduce overall farm costs in Ontario

4.3.2. Biochar application

Biochar applications have been shown to have several positive effects on soil health. A recent literature review found that biochar application improves soil structure, porosity, and water holding capacity, increases nutrient availability, acts as a liming agent, and positively influences soil microbial communities (Subedi et al., 2017). These findings indicate that biochar applications can improve overall yield in cropping systems. Subedi et al. (2017) offer a list of studies showing increased crop yields from biochar applications. Furthermore, a global meta-analysis by Crane-Droesch et al. (2013) showed a 10% average yield increase after one year of applying three t/ha of biochar. These sources point to the economic benefits of biochar applications for growers.

However, not all crops benefit from biochar application. Sorensen and Lamb (2016) found no yield effect for peanut, corn, or cotton in a one-time application at various rates of biochar. Similarly, Jay et al. (2015) found no yield advantages in strawberry, potato, or barley. Subedi et al. (2017) list other studies showing neutral to negative yield effects as well.

While the results on the effect of biochar application on yield are mixed, Subedi et al. (2017) argues that biochar functionality differs with how it is produced, what crops it is applied to, the soil to which it is applied, and with what other fertilizers/amendments it is applied. For soil type interaction in particular, Burrell et al. (2016) found that soil health response to biochar application tended to be the best on soils with coarser/sandier textures. In addition to this finding, Jay et al. (2015) linked well-managed, fertile soils to limited yield response to biochar. This point is interesting given that, in their meta-analysis, Crane-Droesch et al. (2013) suggest that biochar applications have the greatest potential for increasing crop productivity on poor, degraded soils. Soils with low cation exchange capacity, low SOC content, low pH, and fairly non-reactive clay content are specifically mentioned. Generally, this would encompass production regions in the tropics and subtropics. For the relatively productive growing conditions in the North American corn belt, including Ontario, Crane-Droesch et al. predicted low to negative yield response to biochar application. Given this information, it is unlikely that biochar applications will be profitable for Ontario growers if heightened crop productivity is the main source of revenue.

Another potential source of revenue for growers applying biochar is carbon credits. Sorensen and Lamb (2016) offer this as a potential economic strategy given the high C content of biochar and its resistance to breakdown in soils. For this to be effective, the revenue from carbon credits (if available) would have to be weighed against application costs, biochar costs, and crop yield effects (Sorensen and Lamb, 2016).

Table 4.8 Summary of results for the economic potential of biochar application

Source	Key Findings	Implications
Burrell et al. (2016)	-Biochar applications are more effective at improving soil physical characteristics on coarse textured soils	-Biochar applications are more likely to be profitable on coarse textured soils in Ontario
Crane-Droesch et al. (2013)	-Average crop yield increase from a 3 t/ha biochar application in the year of application is 10%	-Biochar applications are less likely to provide positive yield responses in Ontario growing conditions

	-Positive yield responses are more likely on soils with characteristics applicable to many agricultural areas in the humid tropics	
Jay et al. (2015)	-Variable rates of biochar had no effect on strawberry, potato, or spring barley yields in the United Kingdom.	-Fertile, well-maintained soils in temperate regions are less likely to provide economic response from biochar applications
Sorensen and Lamb (2016)	-Variable rates of biochar application had no effect on cotton, peanut, or corn yields in the Southeast United States -Obtaining carbon credits from biochar applications may act as a potential source of revenue	-Even without any yield or quality effects, biochar applications could be profitable where carbon credits are available

4.4. Afforestation and tree barriers

Afforestation can provide financial benefits to farmers if adopted under specific circumstances. Winans et al., (2015) compared the C sequestration potential as well as the costs and benefits of four cultivation systems in Southern Quebec. The cultivation systems included: a HP and hay intercropping system, a HP plantation, grain corn, and hay. The costs and benefits of the cultivation systems were measured using replacement chain[2] and equivalent annual annuity approaches with alternate discount rates. The economic benefits, without consideration of environmental benefits, was greatest for grain corn, followed by hay, HP-hay intercrop, and lastly HP grid plantation. When the C sequestration benefits were accounted for, the highest benefits were accrued for HP grid plantation, HP-hay intercrop, grain corn, and lastly hay. The authors suggest that the grain corn cultivation system is the only one expected to generate positive returns at three discount rates and three price options. They conclude that in order for tree-based cultivation systems to be viewed as financially viable for producers, a pricing system for C sequestration would need to be created and this could be achieved through C-trading programs and other government policies.

The necessity of a C valuation system was highlighted by Yemshanov et al., (2005). Through the simulation of three afforestation scenarios, the authors identified the investment attractiveness of afforestation in Canada by calculating net present values from plantation investments. The three scenarios include: HP, hardwoods, and softwoods with average growth of 14 and 6-7/ m^3/ha/year. The benefits included in the calculation were potential credits obtained from C sequestration, and net revenues from fibre, the present value of C offsets from burning wood instead of fossil fuel. The costs included potential agricultural production forgone. The results show that afforestation is not likely to be financially viable with a zero-carbon price.

[2] A capital budgeting method used to compare two or more mutually exclusive proposals with unequal lifespans.

The highest break-even values for C prices are found in certain areas due to the high agricultural land opportunity costs. The highest break-even values are found in Coastal British Columbia, Southern Ontario, and Southern Quebec. The lowest break-even carbon prices are found in Northern Prairies, along the boreal forest transition zone (Yemshanov et al., 2005). The lowest prices for conifers are in East-Central Ontario, along the agriculture-forest transition zone. Lastly, the best locations for hardwoods are found in the Northern Prairies and parts of the Peace River Regions, areas of Central Ontario and Northern Quebec. HPs have the lowest overall break-even prices. The model shows that very little land is attractive for afforestation at C prices less than \$7/t CO_2 and a price of \$5/t CO_2 is the minimum price that makes afforestation an attractive investment for landowners (Yemshanov et al., 2005). Higher C prices result in higher valued lands becoming more attractive to afforestation. These results suggest the importance of C prices and agricultural land opportunity costs in determining the economic viability of afforestation in Canada (Yemshanov et al., 2005).

Yemshanov et al., (2015) used a real option value-based framework to identify a land use change model that reflects the economic considerations of private land use decisions. The authors state that this model helps address potential irreversibility of decisions related to converting land to forestry. This study observes forestry and agricultural conversions in Alberta to estimate land use change patterns. The focus of the study is on private forestry land currently used for agriculture that may switch to HP forest plantations. The area of interest is a band of marginal agricultural land in the northern and western forested areas of the province. Annual land transfer values and Canada Land Inventory classifications determined the land values used. The authors found that the higher the land value, the less likely it is to be converted from agriculture to forestry land. Furthermore, land is more likely to be converted from forestry-to-agriculture than from agriculture to forestry. The results from this case study can be compared to other locations because the land use conversion elasticities are similar to other North-American estimates (Yemshanov et al., 2015). The key findings suggest that afforestation can be a good alternative to agriculture if the land is marginal. If the land is productive, an alternative could be inter-cropping of trees and agricultural crops or fruit trees (Yemshanov et al., 2015).

Afforestation is a potential form of profit for farmers. For afforestation to provide economic benefits to farmers, the payments for afforestation measures would need to be equal to or greater than the opportunity cost associated with the loss of land for production. It would not be economically desirable for a farmer to plant trees on productive land, but planting trees on marginal land prone to erosion or drought could be profitable (Yemshanov et al., 2015). There are also opportunities for profit if compensation is provided to farmers for planting trees as a way to offset their C emissions (Yemshanov et al., 2005).

Table 4.9. Summary of results of literature review of afforestation

Source	Key Finding	Implications

Balooni (2002)	inter-cropping or farm forestry is a better form of afforestation from the cost benefit ratio view	In order for trees to act as a C sink, they should be left standing for 60 years (Alberta Government, 2017). This restricts the options of afforestation to provide economic benefits. Inter-cropping allows land to be used for agriculture, while still planting trees
Balooni (2002) and Yemshanov (2015)	Marginal agricultural land can be converted to forestry	Unproductive, lower-value agricultural land could be put to better use, and provide a higher economic return, if it is converted to forestry. The economic returns depend on the compensation received by the farmer for use of the land as a C sink
Yemshanov (2015)	Planting hybrid trees as opposed to native species can increase financial return for conversion to forestry	Hybrid trees grow faster and can provide a higher payback when harvested. If the plantations are to be used for forestry industry purposes, the trees must be left standing for the appropriate period of time, in order to act as a C sink. This may restrict economic opportunities available to farmers

4.5. Summary of farm-level costs and benefits of the selected BMPs

N-Rate optimization to match crop need

- For Ontario conditions, the economically optimal range is bewteen 100 and 150 kg/ha of N for corn (Rajsic and Weersink, 2008). Beyond the economically optimal level of N application N_2O emissions increase at an increasing rate (Grant et al., 2006; Kim et al., 2013; Kim et al., 2010).

- The optimal level of N is case specific. Technologies to find this optimal level exist, but have high impletementation costs that tend to outweigh the small potential benefits (Pannell 2017). The profit losses from overapplication tend to be less than 20/ha (Rajsic et al., 2009).

- Weather and price risk may influence N application rates. Weather risk may motivate application above the recommended rates. A reduction in price risk also may result in higher application rates (Rajsic et al., 2009).

N-placement and timing

- Side-dressing allows N rates to be adjusted based on crop appearance part way through the season (McDonald, 2015). N is applied closer to when corn begins to require it the most. This allows for fewer losses and reduced costs for growers relative to broadcast applications.

- The information provided through tools like soil/tissue N tests used to determine more accurate N application rates with side-dressing is worth approximately $14 per acre (Bontems and Thomas, 2000). The amount that a grower would pay to be absolved of risk, including production risk, is $2.50 per acre (Bontems and Thomas, 2000). These values combined make up 20% of profit per acre and about 30% of the fertilizer cost in the U.S. Midwest (Bontems and Thomas, 2000). Thus, there is significant benefit to side-dressing N given the yield and soil information it can provide plus the reduction in production risk.

Mineral fertilizer vs manure N

- The largest benefit of using manure fertilizer is lower cost of the actual fertilizer of manure than mineral fertilizer (Adhikari et al., 2005; Huijsmans et al., 2004).
- Manure application is associated with other costs such as has high fixed cost, labour costs, transportation costs, and could risk environmental damage from run-off into water sources (Huijsmans et al., 2004; Kaplan et al., 2004).
- The effectiveness of manure fertilizer depends on the land capability to recycle nutrients, crop price, and price of manure products (Janzen et al., 1999). When high value crops are considered, the benefits could be greater (Adhikari et al., 2005; Janzen et al., 1999).
- Potential improvements in management to reduce GHGs needs to be considered against other environmental considerations such as water quality (and perhaps, operational considerations).

Use of inhibitors, polymer coating, and slow release fertilizers

- DCD consistently increased crop yields by 6.5% to 7.5%, while DMPP did not (Yang et al., 2016).

- A slight revenue advantage was obtained with the urea and ESN blend, and no benefits were shown from applying ESN alone (Walsh and Girma, 2016). The study confirms that ESN provides environmental benefits but points out the need for hesitancy when adopting ESN for improving economic returns.

- While the environmental benefits have been explored, the economic implications of CRFs have not been examined in detail.

Cover Crops

- The benefits of CCs include increased yield sustainability, marketability of cash crop, reduced disease and pest cycles, reduced fertilizer cost, and weed suppression (Morton, et al., 2006).
- The economically viability of a biomass from a CC is specific to the CC.
- There is a trade-off between CCs that produce low quality in large quantities, such as grasses, and those that produce moderate amounts at a higher quality, such as legumes.

The establishment costs of legumes can be up to 10 times higher than that of grasses (Morton et al., 2006).

- The largest cost in planting CC is the high initial cost as well as the high opportunity cost of income forgone of cash crops (Morton et al., 2006).
- The cost of CC adoption in combination with economic returns that have not been consistently quantified deters producers from adopting the practice of cover cropping (Schipanski et al., 2014).
- The environmental benefits provided by CC use can offset CC adoption costs and total costs (Roth et al., 2018).
- The potential, for economic gains is highly dependent on input prices, seed costs, environmental factors as well as management practices (Bergtold et al., 2017).

Crop Rotation

- The incorporation of additional crops into the continuous corn and corn-soybean rotations that dominate Ontario's cropping landscape can contribute to higher corn and soybean yields (Congreves et al., 2015; Gaudin et al., 2015a; Gaudin et al., 2015b; Meyer-Aurich et al., 2006a; Meyer-Aurich et al., 2006b).
- Yield variability is reduced with more complex rotations (Gaudin et al., 2015b; Meyer-Aurich et al., 2006a).
- Net returns are greatest with rotations including wheat (Meyer-Aurich et al., 2006a; Meyer-Aurich et al., 2006b).
- All articles presented stress the particular importance of alfalfa and/or wheat being included in rotation to maximize the benefits described above.

Tillage

- Reducing tillage intensity generally does not translate into higher yields and can, in fact, lead to higher yield variability, with the latter point being particularly true in finer textured soils (Beyaert et al., 2002; Dam et al., 2005; Munkholm et al., 2013; Vanhie et al., 2015; Vetsch et al., 2007).
- NT offers the benefit of lower equipment/labour costs relative to other tillage systems (Sørensen and Nielsen, 2005).
- For whole-farm economics, less intensive tillage systems like NT tend to be neutral to beneficial (Archer and Reicosky, 2009; Triplett and Dick, 2008; Uri, 1999; Weersink et al., 1992).

Biochar

- Biochar has the potential to improve soil health and stability despite some drawbacks including toxic compounds and heavy metals potentially being added to the soil (Subedi et al., 2017).
- Yield benefits vary with crop, soil, and management practices (Crane-Droesch et al., 2013; Subedi et al., 2017).

- For Ontario conditions, there is unlikely to be significant yield benefits from biochar applications (Crane-Droesch et al., 2013). However, C credits should be explored as another potential source of revenue for biochar (Sorensen and Lamb, 2016).

Afforestation

- The key findings suggest that afforestation can be a good alternative to agriculture if the land is marginal. If the land is productive, an alternative could be inter-cropping of trees and agricultural crops or fruit trees. The polyculture model could also provide benefits of reducing erosion (Yemshanov et al., 2015).
- Very little land is attractive for afforestation at C prices less than \$7/tCO_2 and a price of \$5/t CO_2 is the minimum price that makes afforestation an attractive investment for landowners (Yemshanov et al., 2005). Higher C prices result in higher valued lands becoming more attractive to afforestation and this suggests the importance of C prices and agricultural land opportunity costs in determining the economic viability of afforestation in Canada (Yemshanov et al., 2005).

5. Environmental and Economic Assessment Summary

5.1. Summary of findings of environmental and economic literature review

Table 5.1 summarizes environmental benefits in terms of GHG reduction potential and farm-level costs and benefits. The environmental benefits are derived in the most part from Chapter 3, while the farm-level costs are benefits are drawn from Chapter 4.

The major categories of on-farm benefits include additional outputs/revenues, increased yield, reduced input costs, and soil quality improvement. Additional outputs or revenues may result from outputs that result from the adoption of a particular BMP (i.e., biomass output and revenue from a CC). Reduced input costs may result from a reduced quantity of input used (i.e, less fertilizer used) or a complete absence of an input that was previously used (i.e., some pieces of plowing equipment in NT system). Soil quality improvement may be an improvement in soil structure, reduction in erosion, or an increase in organic matter.

Major on-farm costs include implementation costs, increases in input costs, and reductions in yields. For some BMPs, there may be both reductions in some input costs and increases in other input costs. For example, pesticide costs may increase for NT compared to CT, while fuel costs may be reduced for NT. However, since the relative sizes of these changes are uncertain, it is hard to predict the net effect.

5.2. Knowledge gaps

- More research is needed to discuss profitable strategies for finding the optimal level of N
- Side-dressing and split N fertilizer application needs more research to link emissions to environmental factors and soil types and tillage practices. Crop yield responses to split and sidedressed N needs more research as well. Accounting for increased passes on the field with split application to be assessed economically and environmentally.
- Injection of liquid N fertilizer and its combined effect on yield and N_2O and NH_3 emissions needs further research. The injection depth and application on different types of soils also needs more testing.
- More research on applicability and potential N savings with using sensor-based variable rate application linked to economical assessment.
- Effects of manure application time and manure application methods on direct and indirect N emission and the interaction with soil type.
- More information is needed on the anaerobically digested manure effect on leaching, volatilization, and emission of N_2O.
- Priming of SOC with the addition of organic amendments and the long-term effects on N dynamics and relation to crop growth.

Table 5.1 Summary of the major categories of benefits and costs associated with the reviewed BMPs. A checkmark means there was enough evidence or a general consensus from the literature that a BMP contributes to the specific mitigation. No checkmark means that evidence is lacking to generalize based on the literature reviewed.

BMP	Major GHG Reduction Benefits			Major On-Farm Benefits				Major On-Farm Costs		
	Reduced direct N_2O emissions	Reduced indirect N_2O emissions	Increased soil C storage	Additional Outputs/ Revenue	Increased Yield	Reduced input costs	Soil quality improvement	Implementation costs	Increased input costs	Possible yield reduction
Optimizing Nitrogen Application rate	✓	✓			✓	✓		✓		
N-Placement and Timing						✓		✓		✓
Manure Nitrogen						✓	✓		✓	
Nitrification Inhibitors	✓	✓			✓	✓			✓	
Controlled-Release Fertilizers									✓	✓
Cover Crops		✓	✓		✓			✓		
Biomass Crops	✓	✓	✓	✓	✓		✓	✓	✓	
Crop Rotation					✓		✓			
Reduced Tillage	✓		✓			✓	✓	✓	✓	✓
Biochar				✓			✓		✓	
Afforestation			✓	✓				✓		

- More research needed on the effects of compost and composted manure on both direct and indirect GHG emissions. The factors driving emissions (e.g. availability of labile C source, NH_4^+/ NO_3^- concentrations, soil conditions, soil water conditions) need to be identified.
- More research on the effects of slow release fertilizers and NI and UI use on different types of soils and in different tillage systems so that the ranges of estimated benefits can be narrowed down. As well, research is scant on the economic assessment of NI and UI use in agriculture warranting experiments to specifically test the economic viability of these additives. There are significant gaps in the research surrounding the economic benefits of CRFs and claims of 'reductions in application costs' are made without a source. The perception that economic benefits of CRFs exist could mean that there are cost-savings that research has yet to illuminate.
- More research is needed to determine the conditions when yield is reduced under NT compared to CT (and impact on yield-scaled N_2O emissions) and the factors that control this relationship.
- Although experiments assessing the effect of tillage systems on grain yield and quality have been conducted, there is little research that we found that looks at the associated equipment/labour or crop input costs alone.
- Emissions from the non-growing season including the winter and early spring and thaw events. Some BMPs such as reduced tillage and use of CCs are expected to have different emission patterns in the growing and non-growing seasons. This is because of the effects of factors such as soil cover and temperature and moisture conditions. Other BMPs that cause a change in yield (residue return; C:N ratio effects) or that cause a change to the soil physical properties can also have different emissions patterns across seasons. More research is warranted here.
- Cover crop effects:
    - Effect of CC type (legume, non-legume, mixed species)
    - Interaction of CC with tillage
    - Termination time and termination method
    - Full effect on GWP i.e. C sequestration, N_2O emission, NO_3^- leaching (under different soil types) and CO_2 emission
    - Amounts of N credit that can be counted for following crops from legumes and non-legumes. As concluded in Roth et al. (2018) study "The ability to estimate CC residue N content could allow producers to estimate the potential N cycling benefits of CC, and help them consider and recognize short-term valued benefits of CC."
    - Economic estimates of benefits of CCs not well understood

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Appendix

1. Methodology for Baseline Emission Calculations

1.1. **Model inputs**

1.1.1.Agricultural census data

To conduct the county-scale inventory of N_2O and methane emissions from Ontario agriculture, Jayasundara and Wagner-Riddle (2010) used data from the Census of Agriculture. As Statistics Canada conducts the census every five years, census data was only available for the years 1991, 1996, 2001, and 2006. Linear interpolation was used to estimate data in the inter-census years. The current report used the 2011 and 2016 censuses to update the inventory for those two years. Due to time restrictions, data was not estimated for the inter-census years, although this can be done in future studies.

Emissions were estimated for each county or census division individually, as well as in total for each of the five Ontario regions – Southern, Western, Central, Eastern, and Northern. The counties or census divisions included in each region by Statistics Canada are listed in Table 1A. The census data used to estimate N_2O emissions included the total area of farms, land use, irrigation, tillage, area of various field crops, tree fruits and berries, vegetables, and sod. To estimate N_2O emissions resulting from manure application to soils, data on the number of livestock, such as cattle, pigs, sheep, horses, goats, and poultry was also obtained from the census. Table 1A in the Appendix explains in detail the types of data that were used. Most of the land use, crop, and livestock categories were the same as the categories used by Statistics Canada in the Census of Agriculture, with the specific adjustments described below:

- The total area of farms was calculated by summing the following land use categories: land in crops, summerfallow land, tame or seeded pasture, and all other lands. All other lands included the "all other land" category from the Census of Agriculture as well as the area in Christmas trees, woodlands and wetlands. For the estimation of N_2O emissions resulting from reduced tillage practices, the areas of NT, zero-till, and tillage retaining most crop residue on the surface were combined into one category.
- In the field crop data, the areas of land planted to dry white beans and other dry beans were combined into one category. Data on the areas planted to tobacco and caraway seed was only available until 2001 because these categories were terminated by Statistics Canada in the subsequent censuses. Thus, tobacco and caraway seed were not included in N_2O emission estimation calculations after 2001. All other field crop categories were the same as in the Census of Agriculture.
- Total area of tree fruits was a summation of the areas planted to apples, pears, plums and prunes, sweet cherries, sour cherries, peaches, and apricots. The area of berries was the total of strawberries, raspberries, cranberries, blueberries, and saskatoons.

Table 1A. Ontario counties and census divisions		
Region	County or Census Division	
Southern Ontario	Hamilton Division Niagara Regional Municipality Haldimand-Norfolk Regional Municipality Brant County Oxford County	Elgin County Chatham-Kent Division Essex County Lambton County Middlesex County
Western Ontario	Peel Regional Municipality Dufferin County Wellington County Halton Regional Municipality Waterloo Regional Municipality	Perth County Huron County Bruce County Grey County Simcoe County
Central Ontario	Hastings County Prince Edward Division Northumberland County Peterborough County Kawartha Lakes Division Durham Regional Municipality	York Regional Municipality City of Toronto[a] Muskoka District Municipality Haliburton County Parry Sound District
Eastern Ontario	Stormont, Dundas and Glengarry Counties Prescott and Russell United Counties Ottawa Division Leeds and Grenville United Counties	Lanark County Frontenac County Lennox and Addington County Renfrew County
Northern Ontario	Nipissing District Manitoulin District Sudbury District Greater Sudbury Division Timiskaming District	Cochrane District Algoma District Thunder Bay District Rainy River District Kenora District

[a] N_2O emissions were not estimated for the City of Toronto because agricultural activity in this area was considered to be significantly small

- In the earlier report, Jayasundara and Wagner-Riddle (2010) included the following three weight categories for pigs other than boars and sows – less than 20 kg, 20-60 kg, and over 60 kg. However, the Census of Agriculture has the following categories: nursing pigs, weaner pigs, and grower and finishing pigs. Using data available from the OMAFRA Livestock Statistics (2017), it was determined that the sum of nursing pigs and weaner pigs is approximately equivalent to the number of pigs in the >20 kg weight category. Also, animals weighing 20-60 kg comprise about 52.6% of the grower and finishing pigs category, while pigs weighting over 60 kg comprise 47.4% of that category. By applying these adjustments, the census data was recalculated to obtain the approximate number of pigs in each of the three weight categories defined by Jayasundara and Wagner-Riddle (2010).
- Total sheep was computed as the difference between the “total sheep and lambs” category in the Census of Agriculture and the number of lambs. Number of hens was the difference between “total hens and chickens” and “broilers, roasters and Cornish”.

1.1.2. *N excretion rates*

In order to estimate N_2O emissions resulting from manure application to soils, N excretion rates from livestock had to be calculated. For pigs, sheep, horses, goats, buffalo, llamas and alpacas, chickens, and turkeys, the N excretions rates were assumed to not have changed over time, so the same values as those calculated by Jayasundara and Wagner-Riddle (2010) could be used. However, cattle N excretion rates were adjusted according to current feeding practices for 2011 and 2016. This was done because the cattle category is a major source of N excreted.

For dairy cows, dairy heifers, calves, and bulls, N excretion rates were obtained from Chai et al. (2016). In that study, N excretion rates were estimated for housing and grazing cows for the four ecoregions in Ontario (Boreal Shield, St. Lawrence Lowlands, Manitoulin-Lake Simcoe-Frontenac, and Lake Erie Lowland). For the purposes of this report, the values estimated by Chai et al. (2016) were averaged over the four ecoregions. This resulted in the following N excretion rates: 138.0 kg N yr^{-1} for dairy cows; 62.9 kg N yr^{-1} for dairy heifers; 142.3 kg N yr^{-1} for bulls; and 41.5 kg N yr^{-1} for calves.

N excretion rates for beef cattle were calculated using IPCC (2006) Tier 2 methodology as follows:

$$N_{ex} = N_{intake} \times (1 - N_{retention(T)})$$

where N_{ex} is the annual N excretion rate (kg N $animal^{-1}$ yr^{-1}), N_{intake} is the annual N intake per animal (kg N $animal^{-1}$ yr^{-1}), and $N_{retention(T)}$ is the fraction of N intake that is retained. From Table 10.20 in IPCC (2006), the $N_{retention(T)}$ for cattle was determined to be 0.07. N_{intake} was calculated as follows based on Chai et al. (2016):

$$N_{intake} = \frac{DMI \times CP}{6.25}$$

where DMI is dry matter intake (kg $animal^{-1}$ $year^{-1}$) and CP is crude protein content of the cattle diet (%).

Using Table A9 in Jayasundara and Wagner-Riddle (2010), average crude protein content of feed was obtained for beef cows, replacement heifers, slaughter heifers, steers, and calves. These values are shown in Table 2A. Following the methods of Little et al. (2008), DMI was calculated as

for beef cows, heifers, and steers: $DMI = \frac{NE_m intake}{Feed\_NE_m}$

for calves: $DMI = \frac{(average\_weight \times 0.4)}{2} \times 0.01$

In the formulas above, $Feed\_NE_m$ is the net energy in feed for maintenance (Mcal $animal^{-1}$ day^{-1}) and is computed as $Feed\_NE_m = (0.0305 \times DE) - 0.5058$, where DE is the percent digestible energy and should be entered into the calculation as a percentage (e.g. 66.5 not 0.665). It was estimated from Table A4-9 in Little et al. (2008), and is given below in Table 2A. $NE_m intake$ is the net energy intake for maintenance (Mcal $animal^{-1}$ day^{-1}). For mature beef cows, it is equal to

$$NE_m intake = (average\_weight)^{0.75} \times [(0.04997)(Feed\_NE_m)^2 + 0.04631]$$

For growing beef cattle (heifers and steers), the calculation is slightly different:

$$NE_m intake = (average\_weight)^{0.75} \times [(0.2435)(Feed\_NE_m) - (0.0466)(Feed\_NE_m)^2 - 0.0869]$$

The average weights of the different categories of cattle are also given in Table 2A. By applying the methodology described above, N excretion rates were computed for beef cows, replacement and slaughter heifers, steers, and cows. These values are shown in Table 2A. Since steers and slaughter heifers are not kept for an entire year but are slaughtered at the age of about 150 days, the N excretion rates for these animal categories in Table 2A are for 150 days, not a full year.

For the purposes of N_2O emission estimation, N excretion rates for dairy calves and beef calves were averaged to obtain a single value of 23.1 kg N yr^{-1}.

Table 2A. Values of parameters used to calculate nitrogen excretion rates for beef cattle and the estimated N excretion rates

	Crude protein content of diet (%)	Average weight (kg)	Percent digestible energy (%)	N excretion rate (kg N yr^{-1})
Beef cows	13.3	622.9	66.5	95.9
Replacement heifers	13.1	499.0	66.5	86.7
Slaughter heifers	12.8	525.6	66.5	36.2[a]
Steers	11.2	542.3	66.5	32.4[a]
Calves	15.3	288.2	66.5	4.8

[a] excretion rate for 150 days, not a full year

1.2. Estimation of N_2O emissions

Following the methods of Jayasundara and Wagner-Riddle (2010), this study applied the methodology of Rochette et al (2008) to estimate N_2O emissions from agricultural soils. The N_2O emission estimation methods developed by Rochette et al (2008) are used by ECCC in the National Inventory Report, but were adjusted here for county-scale calculation. The estimated N_2O emissions were multiplied by a factor of 298 to obtain CO_{2e} emissions.

2. Results from Baseline Chapter 2

Figure 1A shows the trends in major field crops since 1990. Area planted to soybeans, as well as winter wheat, has increased substantially. Corn area has increased slightly, while the area of alfalfa/alfalfa mixtures and forage crops (tame hay and fodder crops) has experienced a decrease.

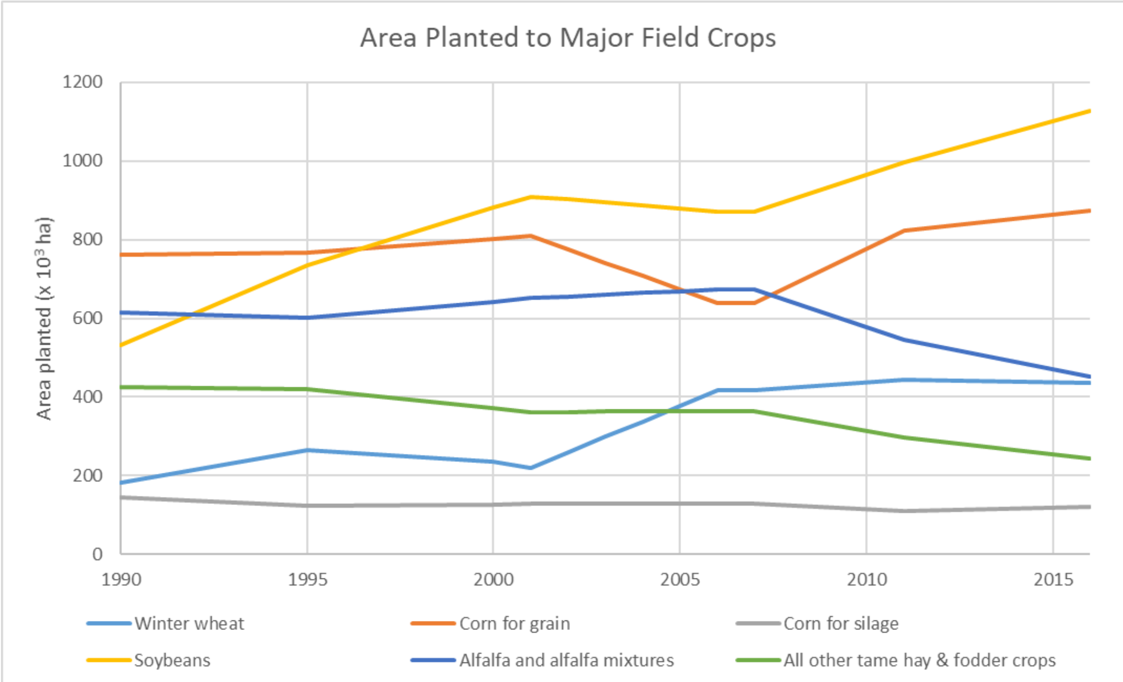

Figure 1A. Area planted to major field crops in 1990-2016

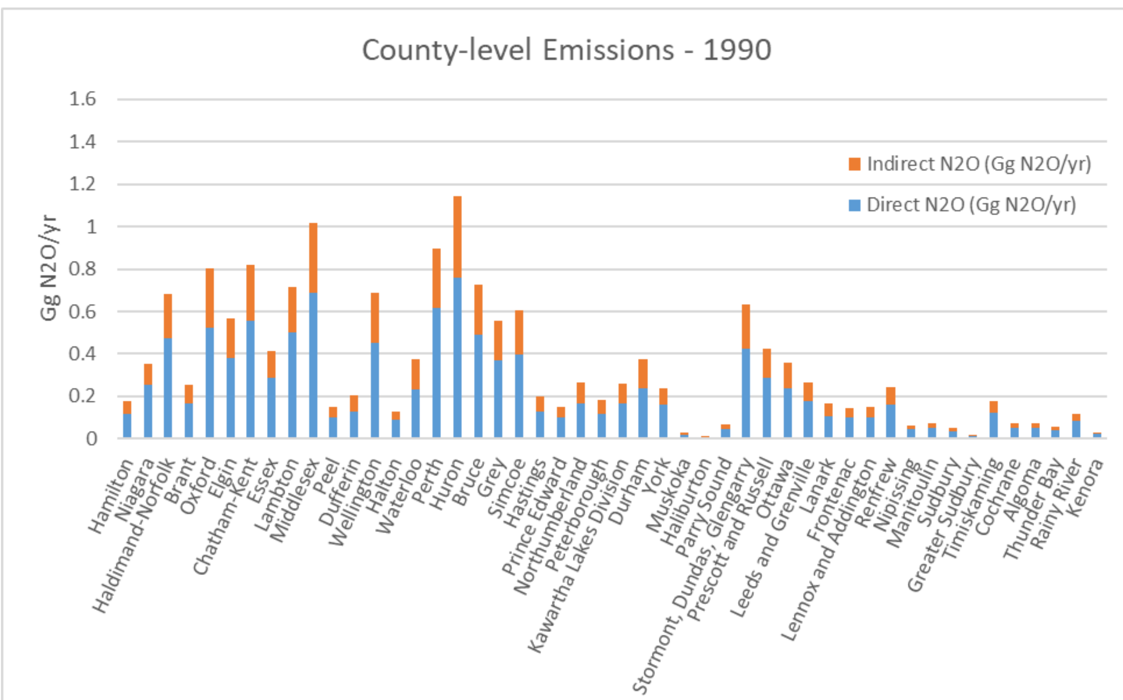

Figure 2A. Total N_2O emissions per county in Ontario in 1990

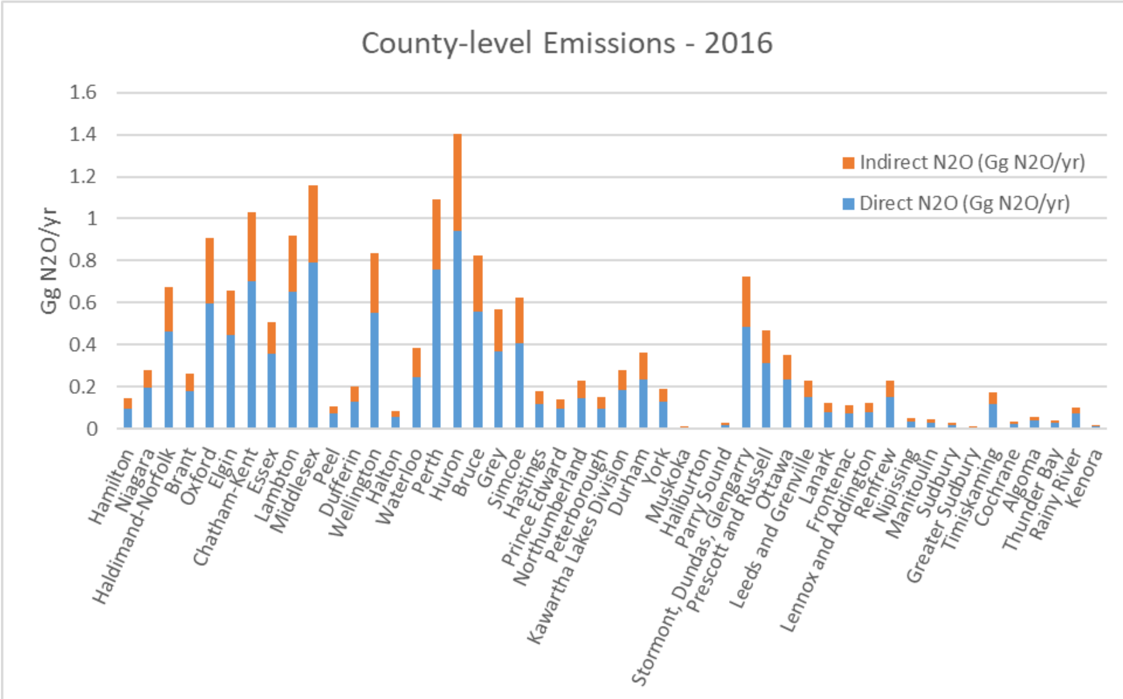

Figure 3A. Total N_2O emissions per county in Ontario in 2016

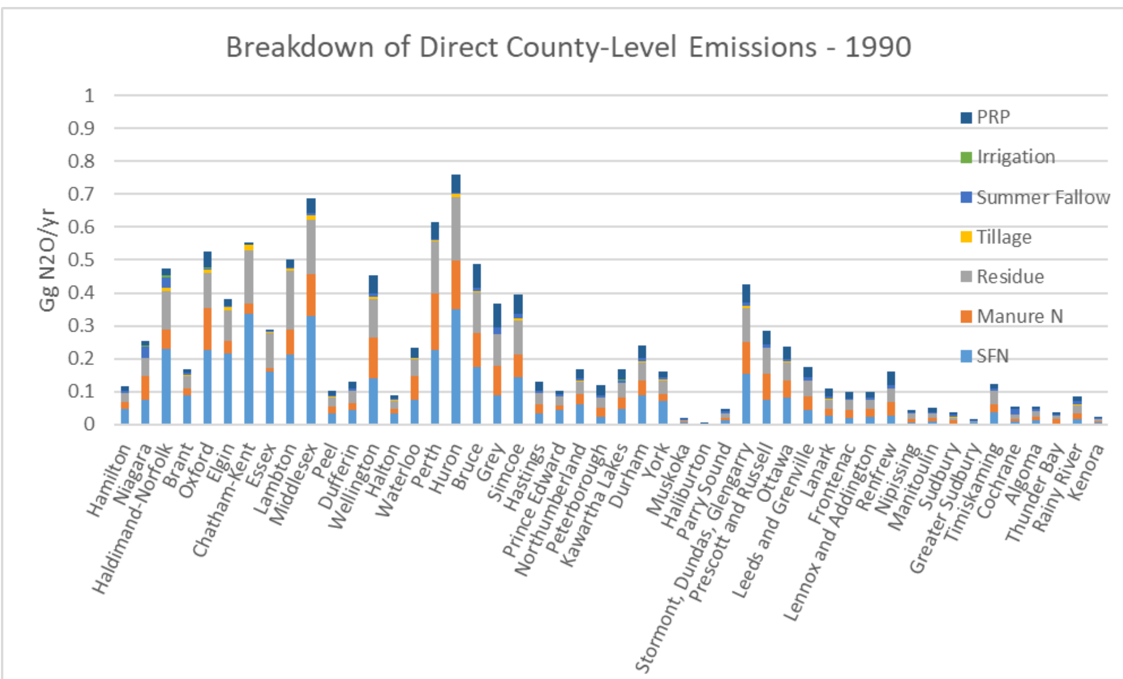

Figure 4A. Breakdown of direct N_2O emissions per county in Ontario in 1990

Livestock are also a major driver of N_2O emissions due to manure production. Figure 5A compares livestock concentration in different counties of Ontario in 1990 and 2016. Livestock concentration was computed as the total number of animals in each county divided by the area of farmland. Although this approach is very simple and may not be appropriate for more in-depth analyses, the maps developed from this calculation can be a useful first step in determining the drivers of N_2O emissions.

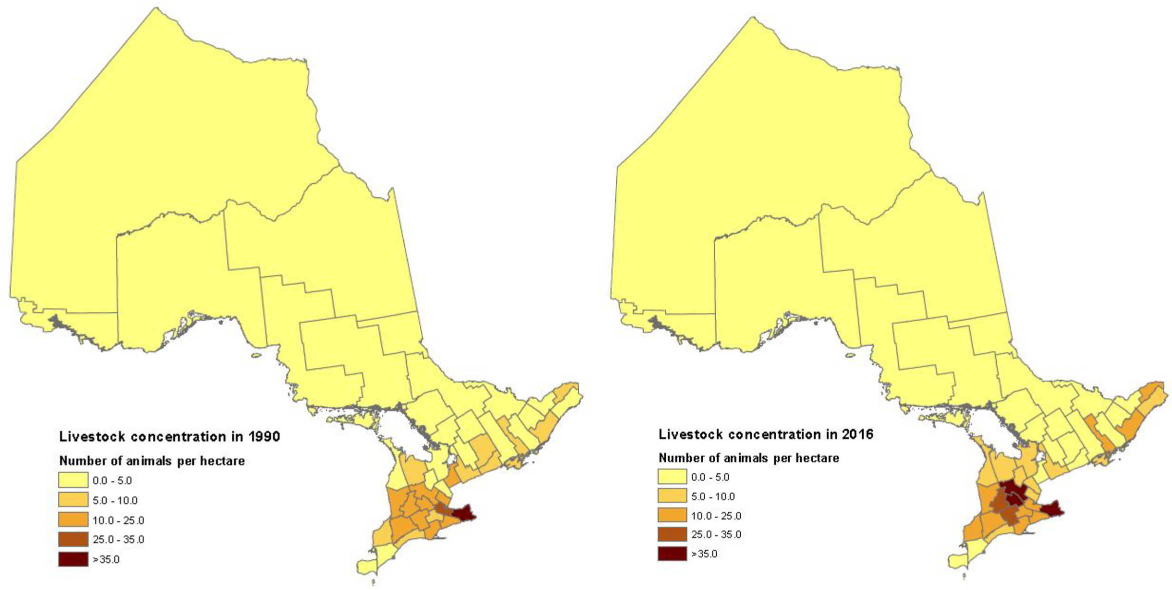

Figure 5A. Livestock concentration (number of animals per hectare) in Ontario counties in 1990 and 2016.

3. Summary of GHG mitigation ranges from the literature with details about conditions where values apply

Table 3A Summary of GHG mitigation ranges from the literature

Management	Range from literature or meta-analysis	Notes
Match N rate to crop demand	From exponential function (Ma et al. 2010) on corn growing season emission only: Reduce N rate from 150 to 90 kg N/ha: save 47-67 g N_2O-N/Mg grain or 0.3-0.5 kg N_2O-N/ha Reduce N rate from 170 to 150 kg N/ha: save 23-33 g N_2O-N/Mg grain or 0.1-0.2 kg N_2O-N/ha From DNDC model (Anderson, 2016) full province wide reduction potential if rate is reduced from 170 to 150 kg N/ha: 0.29 kg N_2O-N/ha/y	N application rate versus N_2O emissions is linear or exponential i.e. more application beyond crop potential/needs will lead to increased specific (yield weighted) emission For comparison: IPCC Tier I applies a 1% N_2O emission factor to fertilizer lost as N_2O-N
Timing of N application	Sidedress versus at-planting: variable results N_2O emission change: • +8 to –38% (Abalos et al., 2016b) • ca. –25% under conventional till (Drury et al., 2012). 10% yield reduction in 1/3 years. • ca. +68% under no-till in wet years (Drury et al., 2012) • ca. –26% full year basis but there was ~5% yield reduction (Abalos et al., 2016c)	Needs further research. Effect of sidedress N application on yield is variable.

	Spring versus fall: Not enough data	
Split N application	Split application: variable results • +13 to +52% more N_2O compared to sidedress (not significantly different) (Abalos et al., 2016c) • –25 to +10% compared to N at planting (Abalos et al., 2016c) • Av. –21% less N_2O compared to N at planting (Anderson 2016) & ~6% less yield • Similar to N applied 10 days after planting (Venterea et al. 2016, Minnesota) • –12% from Split+urea+NI+UI compared to N applied 10 days after planting (Venterea et al. 2016) • –20 to –26% compared to N applied in Fall or spring pre-plant (Eagle et al., 2017)	Still to be considered: increased energy use with split application Full effect on yield Interaction with rain and temperature
N placement	Broadcasting vs. injection or banding found to reduce N_2O by 25-33% (Eagle et al. 2017) 40-68% of broadcast urea lost as NH_3 and the addition of NI+UI inhibitor reduced this loss considerable (Drury et al. 2017) 0.5-2.1% of broadcast urea-N lost as N_2O and 0.7-1.7% of injected UAN-N was lost as N_2O (Drury et al. 2017)	Question of trade-offs between N_2O and NH_3 Broadcasting followed by incorporation reduces NH_3 loss Banding will likely benefit yield and decrease NH_3 if subsurface-banded Injection of liquid N on swelling clay soils risks injection slots remain open and increase NH_3 volatilization Yield increase possibility with injection Injection and injection depth needs further research
N fertilizer type	Shift from anhydrous ammonium to urea expected to result in 45% reduction in N_2O (Eagle et al. 2017)	Not enough research available on this subject
Inhibitors	Urea+NI+UI: • –12 to –61% N_2O vs. urea (Vyn et al., 2016) • –13 to –39% N_2O vs. urea (Eagle et al., 2017) • –0.2 kg N_2O-N/ha vs urea (av. –12%) (Drury et al., 2017) • Similar N_2O from UAN+NI+UI vs UAN (Drury et al., 2017)	Potential of UI to reduce emission is variable. Results from NI are more consistent but the range is wide.

	NI: • –27 to –56% N_2O UAN+NI vs. UAN (Vyn et al., 2016) (+ yield benefit) • –31 to –44% N_2O with NI (+ 7.1% yield increase) (Thapa et al., 2016) • –26 to –43% N_2O with NI (Akiyama et al., 2010) • Av. –31% N_2O with NI (Eagle et al., 2017) • –28 to –46 N_2O with NI (this report meta-analysis) UI: • +0.5 kg N_2O-N/ha vs urea (Drury et al. 2017) • +0.5 kg N_2O-N/ha vs UAN (Drury et al. 2017) & +3-9% more grain yield • +2 to –42% N_2O (Thapa et al. 2016)	Double inhibitor might work to reduce both N_2O and NH_3 especially on alkaline soils More research could help increase certainty.
Polymer coated and slow release N fertilizer	+8% N_2O with PCU vs. Urea (Vyn et al., 2016) +7 to –18% in yield-scaled N_2O emission with PCU vs. Urea (Eagle et al., 2017) –12 to –28% N_2O but possible negative effect on yield (Thapa et al., 2016) No change (Akiyama et al., 2010)	Inconsistent results about PCU
CCs	SOC sequestration potential: • 0.32 ± 0.08 Mg C/ha/y (1.2 ± 0.3 Mg CO_{2e}/ha/y) (Poeplau and Don, 2015) • 1.34 (–0.07 to +3.22) Mg CO_{2e}/ha/y (Eagle et al., 2017) Direct N_2O reduction: • Results dependent on interaction between fertilizer level and C:N ratio of CC (Basche et al., 2014) • CC possible option for mitigation in NT systems (Petersen et al., 2011) • CC provide option for reduced N_2O over warm winters with small snow covers and thaw events (Dietzel et al., 2011) Indirect N_2O reduction: • Winter wheat CC: 14-16% reduction in NO_3^- leaching (Drury et al. 2014b) (23-38% less concentration of NO_3^- in drain) • ca. 53% reduction in NO_3^- leaching with winter rye (Parkin et al., 2016)	Research is still needed especially for effect of CC on N_2O Important to consider full benefit & account for all C and N fluxes Plowing or incorporation of CC results in emission peak i.e. might be affected by tillage system

	• Alfalfa provided 37-63% and red clover provided 46-65% of plant available N for corn compared to control 224 kg N/ha fertilizer (Ontario; Coombs et al. 2017) • Red clover frost seeded to wheat (in corn-soybean-wheat) reduced N rate at maximum grain yield by 47% in tilled and 9% in zone-tilled systems (Ontario; Gaudin et al. 2015).	
Crop diversification Inclusion of perennials	Not enough studies found to provide ranges Diversified rotation including perennials (hay & alfalfa) estimated to add 0.6 Mg C/ha/y as SOC (Gregorich et al. 2001) Replacement of corn with alfalfa in Ontario increased SOC by 8 ± 4 Mg C/ha in 25 years (about 0.3 Mg C/ha/y) (VandenBygaart et al., 2010)	Benefits of longer rotations and diversification are many but not enough emissions data is available
Biomass Crops	SOC: (+ is addition & – is loss) • +1.0 Mg C/ha/y from cropland to pasture (Conant et al., 2001) • –1.77 Mg C/ha/y for switchgrass on land classes 4 to 6 (Liu et al., 2017) • –55 Mg C/ha/y loss of SOC for hybrid poplar on land classes 4 to 6 (Liu et al., 2017) • +2.1 Mg C/ha/y in 60 cm soil layer under unfertilized switchgrass (Valdez et al., 2017) • +0.75-2.18 Mg C/ha/y from giant reed and +0.37-1.58 Mg C/ha/y from switchgrass (Nocentini and Monti, 2017) • +13.5 Mg C/ha/30y from switchgrass and +6.5 Mg C/ha/30y from grass mix (LeDuc et al., 2017) • +0.13-0.29 Mg C/ha/y for switchgrass replacing cropland (Emery et al., 2016) GHG fluxes: (– is a sink of GHG) • av. 84% less CO_{2e} from hay than corn (GHG balance) (Sulaiman et al., 2017) • –4.1 ± 0.3 Mg C/ha/y from switchgrass (net CO_2 flux) and 0.24 Mg C/ha/y (net ecosystem C balance) (Eichelmann et al., 2016b) • –3.4 ± 0.4 Mg C/ha/y from switchgrass (net CO_2 flux) and –0.66 Mg C/ha/y (net ecosystem C balance) (Eichelmann et al., 2016a) • –0.3 to –2.5 Mg C/ha/y for switchgrass (net flux) (Skinner and Adler 2010)	Consistent increase in SOC stock after conversion from cropland to perennial/biomass crops Net flux accounts for GHG emission and C input from primary production (definition might vary depending on study).

	• –2.6 to –1.8 Mg C/ha/y for HP; –2.1 to –1.7 Mg C/ha/y for switchgrass (Adler et al., 2007) (net flux) • –78 Mg CO_2-C/ha/30y for miscanthus and –36 Mg CO_2-C/ha/30y for switchgrass (total GHG compared to corn-soybean) (Hudiburg et al., 2015) • –94 kg N_2O-N/ha/30y for miscanthus and –66 kg N_2O-N /ha/30y for switchgrass (compared to corn-soybean) (Hudiburg et al., 2015) • –3.4 kg N_2O-N/ha/y for switchgrass and miscanthus (compared to corn) (Smith et al., 2013) • 6 times less N_2O emission from hay than corn fields (or 1.5 times less in the hay plow down year) (Sulaiman et al., 2017)	
Tillage	C sequestration with NT: (+ is the difference from CT) • +0.1-0.3 Mg C/ha/y for 0-20 cm soil (Six et al., 1999, 2004) • 9% more SOC in 0-30 cm soil compared to CT (Puget and Lal 2005) • 21% more SOC in 0-15 cm soil compared to CT (Senthilkumar et al., 2009) • 36% more SOC in 0-5 cm & 16% more in the 0-100 cm soil compared to CT (Van Eerd et al., 2014) • +23 Mg C/ha more SOC in 0-30 cm compared to CT after about 28 years (Vyn et al., 2006) GHG fluxes: • ZT growing season emission 36-54% less than CT and NT was +5 more to 41% less than CT (Drury et al. 2012) • Full year emission in Ontario rotation (average over years) 1.1 & 1.3 kg N/ha/y for NT and CT, respectively (Congreves et al. 2016) • –19 to –49% reduction in N_2O with ZT compared to CT and +10 to –19 difference N_2O with NT compared to CT (this report meta-analysis) • Yield is affected under NT (Ziadi et al., 2014; Drury et al., 2012; this report meta-analysis)	C storage under NT could depend on the depth of soil layer assessed. Intensity and frequency of tillage seems to have an effect on C storage and GHG emissions from reduced tillage systems Full-year N_2O vs growing season suggest similar emissions from till and no-till Yield is better under tilled system i.e. yield-scaled emissions are smaller
Variable fertilization rate	Total GWP reduction of 10% (modeling study; Li et al. 2016) Ontario study (Ma et al. 2014): potential of reducing N rate by half by using handheld sensor	Research is needed

Manure spring vs fall application	No consistent reports that support the preference of spring to fall applied manure for N_2O mitigation. NO_3^- losses still to be studied more.	Full assessment of all N losses still lacking
Manure application method	Average N_2O from injection 2.5 kg N/ha vs 2.0 kg N/ha for broadcast+incorporation (Abalos et al. 2016a) 30-60% of broadcast manure-N is lost as NH_3 (Meisinger and Jokela, 2000)	Broadcasting+incorporation produces less N_2O and NH_3 Injection only on dry soils can produce less NH_3 without increasing N_2O Injection on clay soils are more prone to NH_3 loss
Anaerobically digested manure & composted manure	Only when anaerobically digested manure was injected it produced less N_2O (2.5 kg N/ha) than broadcast (6.4 kg N/ha) or broadcast and incorporate (5.4 kg N/ha) (Cambareri et al. 2017b) Composted manure 39-45% less N_2O compared to raw manure on alfalfa (Guest et al., 2017; modelling assessment) Composted manure produced ca. 30% less N_2O compared to liquid manure in *non-growing season* (Kariyapperuma et al., 2012)	More research is needed for full-year assessment and full LCA. Interaction with application method and C:N ratio of anaerobically digested manure. Options for additives (e.g. biochar) to manure composting process to be further assessed.
Liming	Not enough data available to report values	More research is needed
Biochar	Not enough field data to report values Meta-analysis: ~7% increase in yield with biochar Inconsistent effect on N_2O & CO_2	More research is needed
Afforestation, SRC, and intercropping	Meta-analysis on afforestation:1) 10-65% C gain in long-term after conversion of cropland to forest/plantation (Guo and Gifford 2002), 2) potential for C sequestration –5 to +20% and broadleaf trees showed highest potential for SOC increase (Laganiere et al., 2010), 3) 0.2-1.9 Mg C/ha/y in SOC with agroforestry (Eagle et al., 2012), 4) 0.4-0.6 Mg C/ha/y on marginal land (Niu and Duiker, 2006), 5) 0.35 Mg C/ha/y for deciduous and 0.26 Mg C/ha/y for conifers in Michigan (Morris et al., 2007), 6) SOC accumulation potential in Saskatchewan shelterbelt 0.7-1.5 Mg C/ha/y (Amadi et al., 2016) SRC: 1) Soil C accumulation 0.4-4.5 Mg C/ha/y for willow & 1.8-4.7 Mg C/ha/y for HP plantation	Research is mostly from U.S. studies and shelterbelt studies from western Canada Questions remain about reversal after termination, soil C saturation and project timeframes.

	(Lafleur et al. 2015 & Winans et al. 2015), 2) av. 0.7 Mg C/ha/y (Eagle et al., 2012) Intercropping: soil C accumulation 1) 0.7-1.38 Mg C/ha/y for HP -hay (Winans et al. 2015), 2) 1.0 Mg C/ha/13y for spruce-barley and 3) 13.5 Mg C/ha/13y for HP-barley compared to sole barley (Peichl et al., 2006), 3) 12 Mg C/ha/25y for cedar-hay, 16 Mg C/ha/25y for HP-hay, 13 Mg C/ha/25y for oak-hay, 7 Mg C/ha/25y for spruce-hay, 6 Mg C/ha/25y for walnut-hay compared to sole soybean (Wotherspoon, 2014)	