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The environmental and economic efficacy of on-farm beneficial management practices for mitigating soil-related greenhouse gas emissions in Ontario, Canada

Review Article

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Abstract

Agriculture is a large source of greenhouse gas (GHG) emissions but changing management practices to those more beneficial to the environment could help mitigate climate change as long as they are economically and environmentally viable. This study examines the environmental (public) and economic (private) effects of adopting ten different beneficial management practices on a representative corn farm in Ontario, Canada. The study integrates changes in GHG emissions in carbon equivalents (CO_{2e}) and changes in profit from changes in costs and revenues in two dimensions to reveal the scope and scale of different kinds of practices. 4R nitrogen management practices are smaller in scale compared to cropping practices and, therefore, have smaller potential costs and benefits. Land use changes, from practices including biomass, afforestation, crop rotation and cover cropping, have larger impacts on soil sequestration and carbon-equivalent GHG reduction, but with significantly greater costs. Seven practices were found to, at least partially, be economically and environmentally beneficial. The adoption of no-till and N-rate reduction is firmly positive, whereas the production of biomass has the largest potential economic and environmental gains. Crop rotation and diversification and cover cropping can be mutually beneficial, as can changing N-application practices. The use of inhibitors may be economically beneficial if yield gains outweigh purchase costs.

Introduction

The Intergovernmental Panel on Climate Change (IPCC) issued a scathing report on global warming progress and highlighted the importance of limiting the increase in temperature to 1.5°C from 2.0°C (IPCC, 2018). This requires large scale changes in energy use and consumption patterns that could seriously affect the role of agriculture in greenhouse gas (GHG) mitigation and emissions. In Canada, the agricultural sector produces 8% of the total GHG (N_2O and CH_4) emissions, making it the second largest emitting sector after energy (Environment and Climate Change Canada, 2019*a*). However, agriculture also has the potential to mitigate GHG emissions by altering management practices and acting as a carbon sink.

Combined, the crop land and Land Use, Land-use Change and Forestry (LULUCF) sectors are currently net sinks of carbon (Environment and Climate Change Canada, 2019*a*). Whether a particular section of agricultural land is a net sink or a source of GHGs depends on management practices and land use decisions. However, the impacts of beneficial or best management practices (BMPs) are 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 practices and GHG mitigation along with the economic impacts associated with the implementation of those practices.

GHG emissions from soils are primarily mitigated through: (1) a reduction of N_2O; (2) a reduction of CO_2; (3) a reduction of CH_4 from soil; and (4) an increase in C sequestration in soil and in long-term biomass. Within crop agriculture, the main processes that serve to mitigate GHG levels are the reduction in N_2O emissions and the increase in C sequestration. Many BMPs exist that can potentially reduce GHG emissions, which can be categorized into three broad fields. Nitrogen fertilizer management practices, which include the 4Rs—right source, at the right rate, at the right time and the right place—directly affect N_2O emissions. Crop management practices, including the use of cover crops (CCs) and alternate crop systems, can reduce emissions and increase C sequestration. Finally, soil management practices, such as reduced tillage and soil amendments, can reduce overall emissions.

The economic effects of practices to reduce GHG emissions are the most contentious aspect. Although the general costs and benefits of many practices have been observed, few studies have investigated the cost and benefit details in ways that producers can relate to their farm operations. Many agricultural producers base their production decisions on profit maximizing behavior, meaning that GHG mitigation practices must be economically beneficial to observe widespread adoption in the absence of policy and legislative intervention. Practices that are both environmentally and economically beneficial usually require knowledge transfer campaigns to increase adoption. However, costly practices will often require incentives from government or other sources. This means that cost-effective practice adoption must consider both environmental and economic factors.

The efficacy of a variety of BMPs have been reviewed (Zebarth *et al*., 2009; Vanhie *et al*., 2015; Vyn *et al*., 2016; Bergtold *et al*., 2017; Hou *et al*., 2017; Qambrani *et al*., 2017; Wang *et al*., 2017; Banger *et al*., 2020), including meta-analysis (Guo and Gifford, 2002; Akiyama *et al*., 2010; Laganiere *et al*., 2010; Crane-Droesch *et al*., 2013; Kim *et al*., 2013; Abalos *et al*., 2014; Basche *et al*., 2014; Popelau and Don, 2015; Thapa *et al*., 2016; Yang *et al*., 2016; Daryanto *et al*., 2017; Eagle *et al*., 2017; He *et al*., 2017). However, these reviews focus on single or groups of practices and mostly lack an economic component. The economic and environmental trade-offs that underlie BMP adoption have also been studied; for example, for sediment abatement in Ontario (De Laporte *et al*., 2010), nitrogen reduction in Spain (Fernandez-Santos *et al*., 1993), phosphorous loading in New York state (Rao *et al*., 2012) and ecosystem services in Western Australia (Kragt and Robertson, 2014). However, these studies often examine a limited number of BMPs simultaneously, lacking a broad overview. Broader scenario analysis for GHG mitigation from agriculture has also been conducted globally (Smith *et al*., 2008) and nationally, in India (Sapkota *et al*., 2019), but only the latter study considers cost-effectiveness. Therefore, the main contribution of this paper is to integrate BMP environmental and economic effects at the farm level.

The purpose of this paper is to assess BMPs to reduce GHG emissions and net carbon equivalents (CO_{2e}) by synthesizing their environmental and economic effects with the broader aim of informing policy decision making. The geographic scope is focused on Ontario, Canada and similar climate regions (i.e., mainly eastern Canada, and the northeastern US regions).

The review begins with an overview of the methods, including the study area and the selected BMPs. This is followed by an analysis of the environmental and economic efficacy of selected BMPs, followed by an integrated summary and concluding remarks.

Method

Overview

To assess various farm BMPs in Ontario, the study used literature to establish their (public) environmental benefits in terms of CO_{2e}, on an area basis (t ha^{-1}), from changes in N_2O emissions (both direct and indirect) and C sequestration. Similarly, (private) economic benefits, from changes in costs and revenues (from changes in products and yields), were acquired from the literature or from Ontario farm budgets, to establish changes in farm profit, in area scaled Canadian dollar terms ($ ha^{-1}). This included changes in costs from changes in farm operations and input use. The environmental (public or social) and economic (private) spaces were then plotted together for each BMP to allow an integrated comparison of the selected nitrogen management practices (Pannell, 2008). This procedure was followed for each selected practice and the practices were then compared to each other in Section 'Integrated environmental and economic comparison of BMPs'.

The environmental analysis was primarily completed through literature review and subsequent conversion to area-based CO_{2e}. The literature review was conducted using Google Scholar, Web of Science and personal communications aiming to include only field studies. Greenhouse and incubation experiments were included only when data from field research were scant. There was a preference for the selection of literature from studies conducted first in Ontario, Eastern Canada and the North Eastern United states, then including studies from Europe, Western Canada and Japan when data from the first countries were scant. More details are available in Yanni *et al*. (2018).

Similarly, the economic analysis was completed using literature values and a farm model. The farm model buttressed Ontario crop budget changes with additional information from the literature and farm-trials. Therefore, the environmental and economic results were placed at the same scale—area-based changes from farm-level practices—using two-dimensional summary figures that attempt to capture the relative nature of the various examined practices to reduce CO_{2e} from agriculture. The representative farm model grows corn in Ontario, Canada, as the framework to examine the environmental and economic costs and benefits of the adoption of the carbon mitigating agricultural practices detailed in the next section. This model is further detailed in Section 'Farm model'.

Study area

The province of Ontario is large, covering multiple ecozones from the southern border with the United States to the shores of Hudson Bay. Northern Ontario is in the boreal zone. Southern Ontario is surrounded by Lake Huron, Lake Erie and Lake Ontario, and is a fertile agricultural region. The southernmost part of the province is latitudinally aligned with Northern California. Agricultural land quality generally decreases from the southwest to the northeastern sections of Southern Ontario.

The National Inventory Report (NIR) by Environment and Climate Change Canada (ECCC) (2019*a*) reports that agriculture accounts for 6.2% 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 2017, 58% (14.8 kiloton N_2O) of Ontario's N_2O emissions (26 kiloton N_2O) was generated by agricultural activities (ECCC, 2019*a*). The main agricultural sources of N_2O are agricultural soils, manure management and burning of residue (ECCC, 2019*a*). The province of Ontario, under the Climate Change Action Plan, is in the process of creating a Land Use Carbon Inventory (LUCI) for the estimation of GHG emissions from agriculture, forestry and other land uses (Climate Change Action Plan, 2017).

Selected BMPs

Many BMPs have been proposed to reduce GHG emissions and enhance environmental benefits. BMP selection was conducted based on information in Yanni *et al.* (2018) a review and

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meta-analysis that considered many studies from Ontario, Eastern Canada and other locations that have similar climatic conditions to Ontario. This paper derives directly from Yanni *et al.* (2018) and is an evolution that better addresses the integrated environmental and economic considerations of a variety of BMPs. Generally, BMPs were selected if they have consistent results showing reduction in GHGs or gain in C stock or a combination of experimental results showing benefits to GHG mitigation in addition to a soil ecosystem process that is scientifically known to lead to mitigation, with minimal economic costs. 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 (ECCC, 2019*a*).

There are many types of BMPs that can be broken down into several categories. Broad categories, of fertilizer management, or crop management or tillage management, can be broken down into sub-categories, like no-till (NT) or strip-till. Even within these sub-categories, there are many possible ways to achieve the general goal. For example, a sub-category of fertilizer management, N-rate optimization, can take the form of N-rate reduction, by 20 or 40 kg ha^{-1}. This means that there are many examples of specific BMPs to examine. Therefore, we examined a single, well defined (representative) practice, in Ontario, for ten types of BMPs, in three major categories, as listed below:

(1) 4R Nitrogen fertilizer management
  (a) N-rate optimization (170 to 150 kg N ha^{-1})
  (b) N-placement (injection to broadcast)
  (c) N-timing (at-planting to split application sidedress)
  (d) N-fertilizer type (anhydrous ammonia to urea)
  (e) Nitrification and urease inhibitors [urea to urea with urease and nitrification inhibitors (NI)]
(2) Crop management
  (a) Cover crops (corn to corn and red clover)
  (b) Crop rotation and diversification (corn to alfalfa hay)
  (c) Long-term perennial and biomass crops (corn to switchgrass)
  (d) Afforestation [corn to hybrid poplar in short-rotation coppice (SRC)]
(3) Soil management
  (a) Tillage (conventional to NT)

Farm model

The most straightforward way to integrate changes in BMPs is to consider the enterprise budgets of a representative farm to illustrate the economic effects of such changes. In this model, we assume that a crop farmer in Ontario calculates annual business profit per hectare, π, as:

$$\pi = P_i Y_i - C_i^{\mathrm{V}} - C_i^{\mathrm{F}} \tag{1}$$

where P_i is the sale price of the output from crop and management choice, *i*, Y_i is the resultant output yield, $C^{\mathrm{V}}{}_i$ is the variable cost of production and $C^{\mathrm{F}}{}_i$ is the fixed cost of production. This farmer receives no monetary benefit from carbon reduction, in that there is no payment for carbon or a carbon market. The soil health and other benefits to carbon captured by the farmer would only be captured by increases in crop yields (likely over a longer time frame). Therefore, this model treats environmental changes, including in N_2O emissions and carbon capture, as external to the economic decision in Equation (1). This model implies that the farmer will not make decisions to provide environmental benefits, which mostly accrue to society, unless there is a clear individual benefit—an increase in farm profit. An Ontario, Canada farm producing corn, in 2019, has the expected return variables as listed in Table 1 (OMAFRA, 2019). Prices are in Canadian dollars, converted from US dollars when necessary at 1.35 CAD/USD.

This farm has estimated annual total revenue of \$1900.81 ha^{-1} and expenses of \$1519.95 ha^{-1} for an estimated annual net revenue of \$380.86 ha^{-1}. These parameters allow the economic impacts of each of the practices to be estimated and compared.

Specifically, financial savings from N rate reductions come from decreased nitrogen use costs (Farm Progress, 2019) times the price of nitrogen. Nitrogen application in Ontario, whether broadcast or injection, is often custom work; however, possession of one type of equipment or the other could reduce costs, or incur new costs, roughly equivalent to the cost of custom work. Regarding N timing, split application sidedress requires an additional application pass at the custom work rate, while incurring a potential yield loss. Regarding N fertilizer type, anhydrous ammonia is \$0.238 kg^{-1} cheaper than urea, while generally being viewed as more prone to gaseous N losses (The University of Tennessee Agricultural Extension Service, 1995). The price of inhibitors is somewhat obscure from publicly available data. The cost of Agrotain Plus, which contains both a urease inhibitor (UI) and NI is between 40 and 80 CAD ha^{-1} (TheCombineForum, 2012; AgTalk, 2017).

Red clover is a CC that provides a nitrogen credit but requires additional seed and farming procedures, including planting and kill, to implement. The additional cost of seed, planting and kill were estimated at \$136.77 to \$150.12 ha^{-1} (Hoorman, 2015).

Switching from continuous corn to alfalfa hay serves as an example of diversification but comes at a potential loss relative to continuous corn. Alfalfa hay has a shorter effective lifespan before replanting (2–4 yr) than a more significant land use change, such as to a stand of switchgrass (10–15 yr), with lower yields. Furthermore, particularly livestock farmers, move in and out of alfalfa and corn, sometimes employing corn–alfalfa–alfalfa–alfalfa rotations. According to the OMAFRA (2017) alfalfa hay calculator, assuming a yield of 9.88 t ha^{-1}, the net return of alfalfa hay production ranged from −\$161 to \$439 ha^{-1}, depending on a price from \$70 to \$220 t^{-1}, with an expected value of \$180 t^{-1} based on the provided optimistic, expected and pessimistic price scenarios. The total variable annual cost of production, including fertilizers and application, crop insurance, baling, fuel, maintenance and labor, was \$1025.49 ha^{-1}. The total annual fixed cost, including depreciation, term interest and miscellaneous, was \$64.23 ha^{-1}. These values assumed a stand life of 4 yr and are averages across those years. The full enterprise budget for alfalfa hay is present in OMAFRA (2017).

The economic return to a complete land use change from corn to biomass is complicated by incomplete markets; however, the break-even price of switchgrass including the opportunity cost of land in Ontario is \$73.29 t^{-1} with an average yield of 15.7 t ha^{-1} (De Laporte *et al.*, 2014). In De Laporte *et al.* (2014) the stand life of switchgrass was assumed to be 10 yr, with no yield in the first year, lower yield in the second and max yield from year 3 to year 10. The establishment costs of switchgrass were \$868.08 ha^{-1}, with sustained annual nitrogen,

Table 1. Summary of economic parameters from growing corn in Ontario in 2019 and the potential examined practices causing changes in each parameter value

Source	Value[a]	Units	Practices causing change
Revenue			
Yield (Price: $189 t^{-1})	10.06	t ha^{-1}	Timing; inhibitors
Variable (operating) costs			
Seed	247.85	$ ha^{-1}	Cover
Seed treatment	3.95	$ ha^{-1}	Cover
Nitrogen (161 kg ha^{-1})	1.26	$ kg^{-1}	Rate; type; cover
Phosphorous (74 kg ha^{-1})	1.20	$ kg^{-1}	
Potassium (50 kg ha^{-1})	0.81	$ kg^{-1}	
Inhibitors	0.00	$ kg^{-1}	Inhibitors
Herbicide	64.49	$ ha^{-1}	Cover
Chemical application	27.55	$ ha^{-1}	Placement; timing
Fertilizer application	27.55	$ ha^{-1}	
Drying	18.48	$ t^{-1}	
Storage	9.40	$ t^{-1}	
Trucking	9.75	$ t^{-1}	
Marketing	0.44	$ t^{-1}	
Fuel	61.90	$ ha^{-1}	NT; cover
Maintenance	51.52	$ ha^{-1}	NT; cover
Labor	31.51	$ ha^{-1}	
Crop insurance	77.34	$ ha^{-1}	
Operating loan interest	32.37	$ ha^{-1}	
Fixed (overhead) costs			
Machinery depreciation	109.96	$ ha^{-1}	NT
Term interest	48.31	$ ha^{-1}	NT
General	16.80	$ ha^{-1}	

[a]Revenue and fixed and variable costs change with enterprise change to alfalfa (diversification) (OMAFRA, 2017), switchgrass (biomass) (De Laporte *et al.*, 2014) and hybrid poplar SRC (afforestation) (Allen *et al.*, 2013).

fertilizer application, land and harvest costs of $425.55 ha^{-1}. Annual per ton (switchgrass yield) costs included bailing, on-farm transport and replacement phosphorous and potassium, totaling $26.46 t^{-1} ($415.39 ha^{-1} at 15.7 t ha^{-1}). Straw prices, a potential substitute, are approximately $66.08 to $110.13 t^{-1} (AgTalk, 2011). The full enterprise budget for switchgrass in Ontario is present in De Laporte *et al.* (2014).

Land use change to SRC has potential in Northern Ontario but is relatively less profitable compared to corn (Allen *et al.*, 2013). Specifically, the 22-yr net present value of hybrid poplar SRC ranges from −$2512 to $3490 ha^{-1}. However, the 22-yr net present value of corn in the representative model, using the same terms, is $5503.85 ha^{-1}. The cost of SRC in Allen *et al.* (2013) was based on four scenarios with different total establishment ($2199–$5305 ha^{-1}) and maintenance ($564–$3055 ha^{-1}) costs, giving rise to a range in total production costs (over 22 yr with seven harvests) from $3176 to $8360 ha^{-1}. The harvesting cost was $25/ODT and the biomass price received was $85/ODT. The full enterprise budget for SRC is present in Allen *et al.* (2013).

Changing tillage practice to NT has been shown to be more profitable than conventional tillage (CT), particularly when there is no loss (or even gain) in yield. In this case, the expenses from NT corn production on the model farm drop to $1440.62 ha^{-1}, raising the net revenue of NT corn to $460.18 ha^{-1} (OMAFRA, 2019).

Environmental and economic effectiveness of BMPs

The environmental and economic effects of various GHG mitigating BMPs are presented in this section. Each sub-section begins with a discussion of the environmental and economic effects of each BMP in general. This is followed by a discussion of the environmental (N_2O emission reductions and carbon sequestration) and yield effects of the specific example BMPs highlighted in Section ‘Farm model’. These environmental and yield effects of the representative practices are summarized in Table 2, adapted from Yanni *et al.* (2018). Changes in farm costs and revenues are examined jointly with the environmental and yield effects established here, in Section ‘Integrated environmental and economic comparison of BMPs’.

Nitrogen fertilizer management

N-Rate optimization

Reducing the rate of N applied to match crop requirements results in less mineral N in the soil 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. Overuse of N also increases farm production costs. However, since the economic costs appear to be small (Weersink *et al.*, 2018) and the environmental benefits of reduced nitrogen application appear large, it makes sense to match N rates to crop needs, rather than over apply nitrogen. Auto-Steer GPS, a precision agriculture technology that helps reduce nitrogen overapplication while ensuring complete coverage, has been widely adopted due to the low cost of the technology. Conversely, variable rate fertilizer application technologies have not been adopted to the same level (Mitchell *et al.*, 2018) since the costs of overapplication are small, while the technology is more expensive (Pannell, 2017). 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.

The specific example of N-rate reduction examined in this study involves reducing the rate of fertilizer N in corn from 170 to 150 kg N ha^{-1} (Table 2). In different studies, this reduction lowered N_2O-N emissions by 0.1–0.2 kg ha^{-1} (Ma *et al.*, 2010) and by 0.29 kg ha^{-1} (Anderson, 2016). In Southwestern Ontario, optimal nitrogen rates for corn were found to be in the range between 100 and 150 kg ha^{-1} (Rajsic and Weersink, 2008; Rajsic *et al.*, 2009).

N-Placement

The placement of fertilizer, location in the soil and application method, is important to ensure fertilizer delivery to the crop during the critical growth stages. Placement options (broadcasting, broadcasting and incorporation, injection and banding) can affect emissions by influencing the form of N loss. For example,

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Table 2. Summary of environmental (N_2O and C) and yield effects of BMPs in Ontario and nearby regions

Management	Practice	Literature effect on N_2O, C or yield	Source
N-Rate	Reduce N rate from 170 to 150 kg N ha^{-1}	0.1–0.2 kg N_2O-N ha^{-1}	Ma *et al.* (2010)
		0.29 kg N_2O-N ha^{-1} yr^{-1} (DNDC model)	Anderson (2016)
N-Placement	Injection to broadcast	25–33% N_2O reduction	Eagle *et al.* (2017)
		0.5–2.1% of broadcast (urea-N) lost as N_2O	Drury *et al.* (2017)
		0.7–1.7% of injected (UAN-N) lost as N_2O	Drury *et al.* (2017)
N-Timing	At-planting to sidedress	+8 to −38% ΔN_2O	Abalos *et al.* (2016*a*)
		−25% N_2O	Drury *et al.* (2012)
		−10% yield in 1 of 3 years	Drury *et al.* (2012)
		−26% N_2O	Abalos *et al.* (2016*b*)
		−5% yield	Abalos *et al.* (2016*b*)
N-Type	Anhydrous ammonium to urea	−45% N_2O	Eagle *et al.* (2017)
Inhibitors	Urea to urea + NI + UI	−12 to −61% N_2O	Vyn *et al.* (2016)
		−13 to −39% N_2O	Eagle *et al.* (2017)
		−0.2 kg N_2O-N ha^{-1}	Drury *et al.* (2017)
		3 9% yield	Drury *et al.* (2017)
CCs	Add CC	1.2 ± 0.3 Mg CO_{2e} ha^{-1} yr^{-1}	Popelau and Don (2015)
		1.34 (−0.07 to +3.22) Mg CO_{2e} ha^{-1} yr^{-1}	Eagle *et al.* (2017)
Crop rotation	Corn to Alfalfa	8 ± 4 Mg C ha^{-1} in 25 years	VandenBygaart *et al.* (2010)
Biomass crops	Corn to switchgrass	−1.77 Mg C ha^{-1} yr^{-1}	Liu *et al.* (2017)
		2.1 Mg C ha^{-1} yr^{-1} (0–60 cm soil)	Valdez *et al.* (2017)
		0.37–1.58 Mg C ha^{-1} yr^{-1}	Nocentini and Monti (2017)
		13.5 Mg C ha^{-1} per 30 yr	LeDuc *et al.* (2017)
		0.13–0.29 Mg C ha^{-1} yr^{-1}	Emery *et al.* (2016)
		−3.4 ± 0.4 Mg C ha^{-1} yr^{-1} (CO_2 flux)	Eichelmann *et al.* (2016*a*)
		−4.1 ± 0.3 Mg C ha^{-1} yr^{-1} (CO_2 flux)	Eichelmann *et al.* (2016*b*)
		−0.3 to −2.5 Mg C ha^{-1} yr^{-1} (net flux)	Skinner and Adler (2010)
		−2.1 to −1.7 Mg C ha^{-1} yr^{-1} (net flux)	Adler *et al.* (2007)
		−66 kg N_2O-N ha^{-1} per 30 yr	Hudiburg *et al.* (2015)
		−3.4 kg N_2O-N ha^{-1} yr^{-1}	Smith *et al.* (2013)
Afforestation	Corn to hybrid poplar SRC	1.8–4.7 Mg soil C ha^{-1} yr^{-1}	Lafleur *et al.* (2015)
		1.8–4.7 Mg soil C ha^{-1} yr^{-1}	Winans *et al.* (2015)
		0.7 Mg C ha^{-1} yr^{-1}	Eagle *et al.* (2012)
Tillage	Conventional to NT	0.1–0.3 Mg C ha^{-1} yr^{-1} (0–20 cm soil)	Six *et al.* (1999, 2004)
		23 Mg C ha^{-1} per 28 yr (0–30 cm soil)	Vyn *et al.* (2006)

broadcast urea-based N fertilizer, which is left on the soil surface uncovered, is more likely to increase NH_3 volatilization leading to indirect N_2O emissions, whereas placement in concentrated areas deeper in the soil tends to increase direct N_2O losses. Though advocated as a BMP, broadcast followed by incorporation of the fertilizer seems to have an uncertain effect on N_2O emissions, and injection can create a hotspot for denitrification with a possibility of increasing emissions. Therefore, increased capital costs to change methods are not justified, preserving the status quo. More specifically, if a farm operation owns a specific applicator, it does not make sense to buy a different system based on this information.

For example, switching from injection to broadcast (Table 2) reduced emissions by 25–33% in corn-based studies in the United States and Eastern Canada based on hierarchical multi-level regression models (Eagle *et al.*, 2017), whereas Drury *et al.* (2017) showed that 40–68% of broadcast urea is lost as NH_3 gas. Drury *et al.* (2017) found that 0.5–2.1% of broadcast urea-N was lost as N_2O compared to 0.7–1.7% of injected UAN-N. This information makes the effect of fertilizer placement

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site-specific involving trade-offs between direct and indirect N_2O losses.

N-Timing

Synchronization of N application with crop demand is an important consideration, especially when the rate of N is reduced from the conventional to minimize N losses. The relationship between N_2O emissions and the timing of N application is linked to weather conditions where emissions increase when the soil is warmer and wetter; for this reason, sidedress and split applications have inconsistent emission reductions' effects. Sidedress application, split or not, could address some of these issues. Although some economic benefits of sidedress have been observed, including the ability to gain increased information before deciding on application rates, the increased machinery costs of sidedress and split application, in particular, do not seem to justify these uncertain environmental benefits, especially with an increase in the risk of not being able to apply in wet conditions.

The example practice related to timing is split application sidedress, where a small amount of nitrogen is applied at planting, followed later by a larger sidedress application (Table 2). Abalos *et al.* (2016*a*) showed that sidedress N applied mostly at the corn V6 stage reduced N_2O emissions by 18.5% on average (range −8 to 38%). In a 3-yr study in Ontario, Drury *et al.* (2012) showed that N_2O emissions decreased by 25% with sidedress under CT, but yields decreased by roughly 10% in one of the years. Interaction of this application practice with the type of tillage practice should be considered because N_2O emissions increase specifically in NT systems in wet years. For mineral N fertilizers, Abalos *et al.* (2016*b*) used two field experiments in Ontario to calibrate/validate DeNitrification DeComposition (DNDC) model scenario analysis showing that sidedress reduced N_2O emissions by 26% compared to N application at planting, with a resulting 5% yield loss.

N-fertilizer type

Due to their differing chemical formulations, changing the type of N fertilizer can have different impacts on emissions. This has to do with the relative bioavailability of the different types of nitrogen and their propensity to remain in the soil. The use of different types of mineral fertilizers does not seem to have a statistically significant impact on nitrogen emission reductions. Therefore, the cheapest mineral nitrogen product applicable to existing application technology makes sense economically. As for the use of manure, studies generally show that emissions are increased with raw manure application; composted and anaerobically digested manure have been shown to reduce N_2O (Kariyapperuma *et al.*, 2012; Cambareri *et al.*, 2017; Guest *et al.*, 2017); however, more research is still needed to determine the complex interactions between manure type, soil and crop types, application time, and application method. Due to availability, manure use has an ambiguous potential to reduce costs, despite a lower price; therefore, mineral fertilizers are more commonly used.

Many comparisons of N fertilizer types relate to the replacement of anhydrous ammonia with urea in the United States, as is the example practice here (Table 2). A shift in the use of anhydrous ammonia to urea is expected to reduce yield-scaled emissions by 45% on corn systems (Eagle *et al.*, 2017). This shift away from anhydrous ammonia is the practice that shows the most consistent effect on N_2O emission reduction in this BMP category.

Nitrification and urease inhibitors

NIs work with ammoniacal fertilizers to reduce the rate of conversion from NH_4^+ to NO_3^- by inhibiting the activity of the bacteria responsible for the first step of nitrification. UIs work with urea-type fertilizers through the inhibition of the microbial enzyme urease, which converts urea to NH_4^+. From an environmental standpoint NI and NI + UI have been shown to be beneficial. They have potential positive yield effects as well. A more efficient and cost-effective fertilizer reduces the amount that farmers use, resulting in cost savings (Ag Innovation Ontario, 2015). Cost savings could also come from the reduction in compliance costs for environmental regulations. Although they are more expensive, properly selected inhibitors may have overall economic benefits.

The example practice in inhibitors is urea + NI + UI (Table 2). Vyn *et al.* (2016) estimated 12–61% reduction in N_2O emission when urea + NI + UI was used compared to urea in rainfed systems. 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, whereas Drury *et al.* (2017) found that 2-yr averages of N_2O emissions were not significantly different: 1.7 kg N ha^{-1} for urea and 1.5 kg N ha^{-1} for urea + NI + UI. Yield increases of 3–9% were also observed for NI and UI treatments.

Crop management

Cover crops

CCs provide many benefits to soil health and to farmers. 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 fertilizer application rates through the provision of organic N for the following crop. There are numerous potential economic benefits of CCs including: increased financial sustainability through 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). 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. O'Reilly *et al.* (2012) assessed the impacts of CCs in Bothwell and Ridgetown, Ontario, showing that, at both sites, with or without N fertilizer, all CCs had profit margins that were as high as or higher than no CC. Legume CCs can provide a range between 45 and 224 kg ha^{-1} of available N for the following cash crop production, depending on the availability of nutrients in the soil, but there is production risk involved due to unknown yield impacts on the main crop (Bergtold *et al.*, 2017), especially if the actual amount of nitrogen credit is unknown. CCs show promise as a BMP to mitigate GHG emissions but considerations for the type of CC, legume or non-legume, tillage and application of additional N should be considered because there is an interaction between these factors that affects the overall emission of GHGs (Basche *et al.*, 2014). The economic benefits of carefully selected CCs show the potential for profit, or small cost increases, particularly when coupled with support programs.

The example CC considered in this study is red clover. Red clover is a common CC in Ontario that offers an N credit to future corn production (OMAFRA, 2019). Other winter CCs may be more appropriate financially, including grains that have potential sale value for human or animal consumption. Particularly, a meta-analysis by Poeplau and Don (2015) reported an SOC stock change rate of 0.32 ± 0.08 Mg C ha^{-1} yr^{-1} from 37 studies, 73% of which were from temperate regions. Similarly, Eagle *et al.* (2012) reported an SOC sequestration potential of −0.02 to +0.88 (av. 0.37) Mg C ha^{-1} yr^{-1} from winter CCs in corn crop systems in the United States. The number of years since inclusion of CCs as well as the depth of soil sampling increases the variability of reported results. Initial SOC is also an important variable factor between soils that can affect the results.

Crop rotation and diversification

Adding a crop to the usual cropping cycle or using a different crop for one or few seasons are practices that are used to help control pests and improve nutrient management and soil health. 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. Meyer-Aurich *et al.* (2006*a*, 2006*b*) outlined increased profit, reduced yield variability and lower GHG emissions for rotations with additional crops relative to a continuous corn scheme. Risk averse growers were found to favor more complex rotations and the profitability of more complex rotations was less susceptible to increased energy costs and decreased crop prices. Gaudin *et al.* (2015a, 2015*b*) similarly found increased corn and soybean yields and increased N use efficiency when other crops (e.g., wheat and alfalfa) were incorporated into the rotation. The inclusion of wheat and alfalfa in rotation also contributed to higher soil health scores in both Elora and Ridgetown (Congreves *et al.*, 2015). Although the GHG emission reduction from crop diversification is unclear, C sequestration does appear to increase with the adoption of more complex rotations. From an economic standpoint, the benefits of crop rotations have been established in history—they reduce production and financial risk and increase yields.

The example crop rotation change considered in this study is corn to alfalfa (Table 2). Although this may represent a significant diversification of crops, it does illustrate the extent of the potential environmental benefits, without moving to a more significant land use change, like perennial grasses or afforestation. Comparing corn to corn in rotation (corn–oat–alfalfa–alfalfa) in an Ontario long-term experiment Drury *et al.* (2014) found that growing season N_2O emissions were smaller from the corn in rotation than the continuous corn. The 3-yr average difference was about 12% for the corn phase only. Diversification was also shown to increase SOC in a 35-yr corn–oat–alfalfa–alfalfa rotation compared to continuous corn with a difference of 14–25 Mg C ha^{-1} in the 70 cm soil profile (Gregorich *et al.*, 2001). 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^{-1} in 25 yr (about 0.3 Mg C ha^{-1} yr^{-1}).

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 economic effects of biomass growth have been assessed in several locations, particularly Illinois (Khanna *et al.*, 2008; Jain *et al.*, 2010), North Dakota (De Laporte and Ripplinger, 2019) and Ontario (De Laporte *et al.*, 2014; De Laporte *et al.*, 2016). These studies show that biomass feedstock can be profitable to produce under specific conditions, without valuing carbon benefits. However, this is particularly tied to the price offered for biomass relative to the price of crops, or the opportunity costs of production. When crop prices are relatively lower, then large scale biomass growth could be attractive, given sufficient offtake opportunities for producers. However, due to the lower demand for biomass crops, large scale replacements of food stocks are unlikely. Therefore, local scale biomass developments, from energy to livestock feed, are the most likely economic path for these types of crops. Biomass crops generally increase SOC and can potentially result in GHG mitigation. There are also potential economic benefits under specific circumstances and market opportunities. Therefore, growing perennial and biomass crops show promise as a BMP for GHG mitigation.

The example biomass crop practice examined in this study is corn to switchgrass (Table 1). Nitrate leaching reduction in 30 yr of planting switchgrass in Illinois, United States was estimated to be in the range of 1.0–1.2 Mg CO_{2e} ha^{-1} compared to corn–soybean rotations (Hudiburg *et al.*, 2015). Similarly, Smith *et al.* (2013) found N reductions of 3.4 kg N_2O-N ha^{-1} yr^{-1} when corn is replaced with switchgrass or miscanthus. A life-cycle assessment in Pennsylvania (Adler *et al.*, 2007) estimated that switchgrass increases SOC by 0.4 Mg C ha^{-1} yr^{-1}. Similarly, a modeling study by LeDuc *et al.* (2017) showed that replacing NT corn with switchgrass increased SOC by 13.5 Mg C ha^{-1} in 30 yr. Rates of SOC increase were estimated at 2.1 Mg C ha^{-1} yr^{-1} for a 60-cm soil profile under switchgrass (Valdez *et al.*, 2017) and at 0.13–0.29 Mg C ha^{-1} yr^{-1} in a 100-cm soil profile (Emery *et al.*, 2016).

Afforestation

Planting of trees within practices such as afforestation, intercropping or SRC have been shown to increase the input of organic carbon in the soil and the storage of SOC in many cases and conditions. Mitigation of GHG could also be a potential benefit with these practices. 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. Afforestation could potentially increase profit for farmers, by planting trees on marginal land prone to erosion or drought (Yemshanov *et al.*, 2015), or if compensation is provided to farmers for planting trees as a way to offset their C emissions (Yemshanov *et al.*, 2005). However, despite consistent indications of C sequestration and potential GHG mitigation, it is difficult to find scenarios where the economic benefit of afforestation makes sense to landowners.

The example afforestation practice in this study is corn to hybrid poplar in SRC (Table 2). Eagle *et al.* (2012) reported that SOC sequestration for hybrid poplar was 0.7 Mg C ha^{-1} yr^{-1}. Hybrid poplar was also found to have SOC sequestration of 1.8–4.7 Mg C ha^{-1} yr^{-1} (Winans *et al.*, 2015).

Soil management

Tillage

Tillage management affects both N and C dynamics in soils and is related to the amount of Organic Matter (OM) input (removing

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Table 3. Summary of the environmental and economic effects of changing management practices on a representative Ontario corn farm

Management practice	Change	Reduction in CO_{2e} ($kg\ ha^{-1}\ yr^{-1}$)			Change in net return ($\$\ ha^{-1}\ yr^{-1}$)		
		Lower	Middle	Upper	Lower	Middle	Upper
N-Rate	From 170 to 150 $kg\ N\ ha^{-1}$	46.8	93.7	135.8	22.32	24.20	36.25
N-Application	Injection to broadcast	−96.0	98.3	191.9	−27.55	0.00	27.55
N-Timing	Traditional to sidedress	−60.3	188.5	286.5	−104.48	−66.01	−27.55
N-Fertilizer	Anhydrous to urea	305.3	339.3	373.2	−42.13	−38.30	−34.47
Inhibitors	Urea to urea + UI + NI	0.0	90.5	459.9	−80.00	−5.38	29.23
CC	Corn to corn + red clover	−70.0	1340.0	3220.0	−60.77	−46.58	8.27
Crop rotation	Corn to alfalfa	586.7	1173.3	1760.0	−541.86	−101.86	58.14
Biomass crops	Corn to switchgrass	−5390.0	8671.7	22,733.3	−113.21	232.61	578.42
Afforestation	Crop rotation to SRC	2566.7	6600.0	17,233.3	−364.36	−249.77	−91.54
Tillage	Conventional to NT	366.7	1100.0	3011.9	71.39	79.32	87.25

or keeping residues), the moisture regime, compaction and soil physical effects. Tillage practices can affect yield and quality of grain, equipment costs and labor and crop input costs. Beyaert *et al.* (2002) found that corn grain yields were not significantly different between spring plow, NT and ZT (zone tillage). Similar results were reported by Dam *et al.* (2005) over an 11-yr study. Long-term rotation trials showed greater corn yields for a CT system compared to a NT system in simpler rotations (e.g., continuous corn and corn–corn–soybean–soybean), but no yield difference was observed for more complex rotations (Munkholm *et al.*, 2013). Tillage choice seems to ambiguously affect crop yield, but greater yield losses due to NT depend on soil texture and increased moisture (Vanhie *et al.*, 2015). Forms of reduced tillage such as ZT provide reduced soil moisture and better aeration which suggests that it can be a better choice that combines increased SOC and reduced N_2O emissions compared to both CT and NT (e.g., Drury *et al.*, 2012). Weersink *et al.* (1992) demonstrated that more intensive tillage systems like moldboard and chisel plowing have an average total farm cost of \$870 ha^{-1} across farm sizes compared to \$792 ha^{-1} for less intensive tillage systems like NT and ridge-till. Reduced tillage shows promise as a potential BMP to mitigate GHGs because there is evidence showing increased C storage and no increase in N_2O emissions, along with generally favorable economic benefits.

The example tillage BMP chosen for analysis is CT to NT (Table 2). Carbon sequestration potential was estimated from 0.1 to 0.3 $Mg\ ha^{-1}\ yr^{-1}$ up to 20 cm of soil depth (Six *et al.*, 1999, 2004) and at 23 $Mg\ ha^{-1}$ up to 30 cm of soil depth over 28 yr (Vyn *et al.*, 2006).

Integrated environmental and economic comparison of BMPs

Each of the example BMPs analyzed in this review has potential environmental and economic benefits and costs. However, many of these are somewhat uncertain, especially in terms of magnitude. Although research on the potential CO_{2e} reductions has been more extensive, there is less economic certainty as to their effects. Over the years, many BMPs have been extensively adopted when they show economic potential, including crop rotation and diversification. However, many ideas remain in the nascent stages of adoption, despite their proven environmental benefits. Many practices can become more profitable over time, as the producer learns to implement the process quickly and efficiently. However, this process has costs, including added financial and production risk and uncertainty.

A summary of the environmental and economic changes brought about by changing management practices on a representative Ontario corn farm is summarized in Table 3. This considers the specific example practices from each of the considered categories in Section 'Selected BMPs' (Table 2).

Lower, middle and upper values were established considering the ranges and common values from Section 'Environmental and economic effectiveness of BMPs' and Table 2. Middle values are typically averages and medians, and occasionally modes. The literature dictates the lack of a single type of middle value, due to inconsistent results and insufficient data to sometimes form meaningful averages, whichever measure was the most readily available, or specifically outlined, was selected. The environmental parameters come from Table 2, which is a summary of the environmental information in Section 'Environmental and economic effectiveness of BMPs' (Yanni *et al.*, 2018). All N_2O emission reductions were converted into carbon reduction equivalents (Environment and Climate Change Canada, 2019*b*). When percentage changes are involved, the study also uses the IPCC default of 1% N_2O emissions per kg of applied N (Jarecki *et al.*, 2015) to derive carbon-equivalent values. As stated in Section 'Farm model', the representative farm applied 161 $kg\ N\ ha^{-1}$. All the environmental benefits were also annualized in per hectare terms. This allows the carbon gains of long-term land use change, including CCs, diversification, NT, biomass and afforestation, to be incorporated into the model on equal ground. The economic parameters come from estimates using the model farm parameters and additional information presented in Section 'Farm model' and Table 1.

The economic ranges detailed in Table 3 depend upon different dimensions of change, including changes in revenue (due to corn yield changes, or diversification into other crops) and/or changes in fixed or variable costs from changes in practice. In the case of N-rate optimization, the N cost changed from \$1.12 to \$1.81 kg^{-1} depending on market prices for different N sources (Farm Progress, 2019), with a middle value of \$1.21 kg^{-1} based on

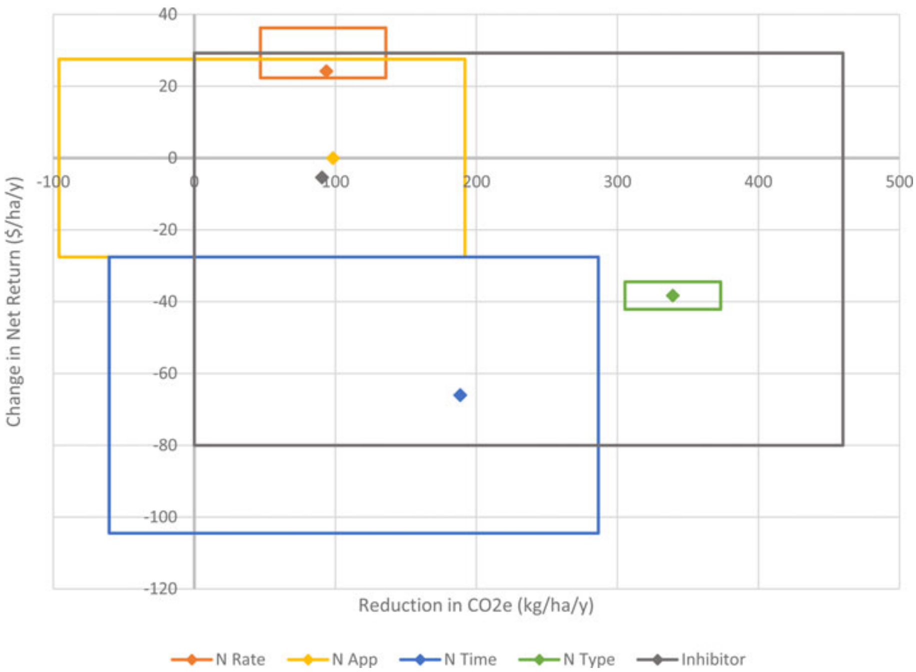

Fig. 1. Range of environmental (public) and economic (private) changes from adoption of 4R practices, including a reduction in N application from 170 to 150 kg ha^{-1} (N Rate), a switch from broadcast to injection (NApp), the use of split-application sidedress (N Time), switching anhydrous ammonium to urea (N Type) and the use of NI and UI (Inhibitor) on a representative Ontario corn farm, per hectare per year.

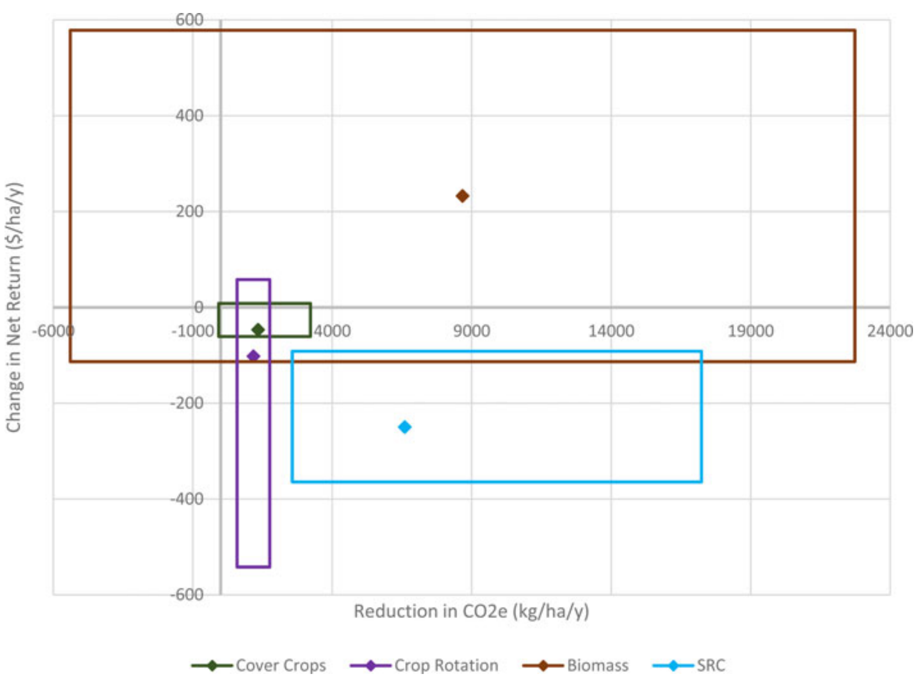

Fig. 2. Range of environmental (public) and economic (private) changes from adoption of land use change practices, including red clover CC (Cover Crops), corn to alfalfa (Crop Rotation), corn to switchgrass (Biomass) and corn to hybrid poplar in SRC (SRC), on a representative Ontario corn farm, per hectare per year.

OMAFRA (2019) crop budgets. Nitrogen placement only varied by the custom work rate. Nitrogen timing required additional custom work and a yield loss ranging from 0 to 10%, with an average of 5%. Nitrogen type return changes were determined by the increased cost of urea compared to anhydrous ammonia, plus or minus 10% of this value (−\$38.30 ha^{-1}). Regarding inhibitors, the cost of inhibitors was from \$40 to \$80 ha^{-1}, with an average of \$60 ha^{-1}, and the yield bonus was from 0 to 9%, with a middle value of 7.1% (Drury *et al*., 2017).

Economic ranges for CCs incorporated the N-rate differences in N-rate optimization multiplied by the 80 kg N ha^{-1} credit, along with the increased costs for CC seed, planting and kill ranging from \$136.77 to \$150.12 ha^{-1}, with an average of \$143.44 ha^{-1} (Hoorman, 2015). The range in corn to alfalfa net returns was determined by the differences in prices, from \$70 to \$220 t^{-1} with an expected value of \$180 t^{-1} (OMAFRA, 2017). The range in biomass net returns was determined by the ranges in straw prices, minus the break-even cost, multiplied by the average annual yield (Section 'Farm model'). Hybrid poplar ranges were determined by the range in expected values compared to the value for conventional corn (Section 'Farm model'). For tillage, the difference in net return from conventional corn (\$79.32

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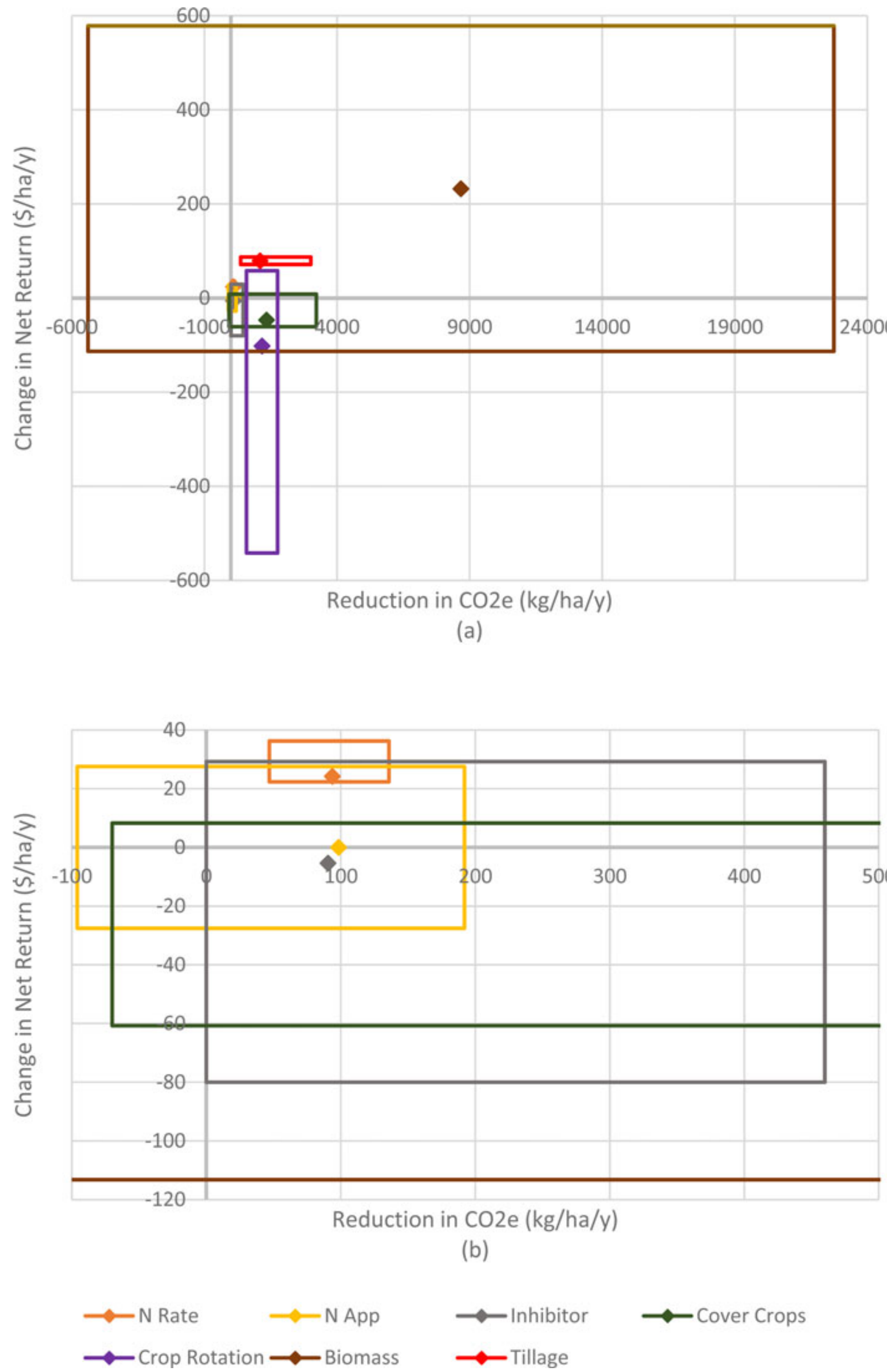

Fig. 3. (a) Range of changes from adoption of potentially both environmentally (public) and economically (private) beneficial practices, including a reduction in N application from 170 to 150 kg ha^{-1} (N Rate), a switch from broadcast to injection (N App), the use of NI and UI (Inhibitor), red clover CC (Cover Crops), corn to alfalfa (Crop Rotation), corn to switchgrass (Biomass) and the adoption of NT (Tillage) on a representative Ontario corn farm, per hectare per year. (b) Zoom of 4R practice area of panel (a).

ha^{-1}) was varied by plus and minus 10% to reflect relative cost certainty, but still establish a range since the yield effects of NT are close to zero.

To consider the relative scale of the various potential management practices for carbon mitigation, Figures 1, 2 and 3 compare changes in net return to changes in CO_{2e} reduction (positive is environmentally beneficial) per hectare per year, for the practices shown in Table 3. This means that annual averages over the lifetime of the project are considered, so that the carbon benefits of long-term projects, including CCs, diversification, biomass, afforestation and tillage are incorporated into the environmental and economic accounting on the same scale as the other BMPs.

Each of the figures presented here are composed of an upper bound, lower bound and middle value. Each of these lines was derived from the academic and extension materials presented previously (Tables 1 and 2). The upper and lower bounds were drawn to create a rectangular representation of the described ranges in economic and environmental values. The graphics are meant to illustrate the potential range of environmental and economic changes from each of the practices and to compare and contrast them.

From a policy standpoint, the quadrants represent different types of practices (Pannell, 2008). In the upper right quadrant, the practice brings both environmental and economic benefit.

In the lower left quadrant, the practice has both environmental and economic costs. In the upper left, the practice has economic benefits, but environmental costs. In the lower right quadrant, the practice has economic costs and environmental benefits. Practices in the upper right are generally encouraged and may need to be more widely disseminated through information and learning campaigns, while those in the lower left are discouraged and should not be adopted. Those in the upper left may be adopted despite their environmental cost and may need to be discouraged through policy. Those in the lower right may need to be encouraged through policy mechanisms if they provide a sufficient quantity of desired environmental benefits.

The highlight of Figure 1 is that 4R nitrogen management practices have different costs but are mostly associated with positive environmental benefits. However, there is some potential for environmental harm in the case of the timing of N application and the method of N application. In fact, these may present joint problems as timing and methods may interact with weather variables, resulting in increased N_2O emissions. Optimizing N rate is typically only beneficial as there is both savings from decreased N used, and lowered N_2O emissions, albeit with potential slight yield losses. Changing N type is costly in this example, with consistent environmental benefit. From a policy standpoint, it makes sense to prioritize the joint environmental and economic benefits of N-rate reductions.

Several of the management practices examined in this study are much more significant than those presented in the case of the 4Rs and represent land use and land cover change. The scale of these initiatives is typically much larger, with gains from carbon sequestration dwarfing the potential mitigation of N_2O emissions (Fig. 2). Although biomass, CCs, crop rotation and SRC all appear to have significant environmental benefits, SRC does not appear to be economically viable. Crop rotation and diversification has been shown to have long-term positive effects in the representative model shown here. Biomass returns are highly dependent upon stable demand but could be both economically and environmentally beneficial. Cover cropping of legumes appears to be effective in this example. Other forage or grain CCs could also be effective due to additional yields to offset seed and operations costs. From a policy standpoint, the benefits of CCs may need to be more widely disseminated to enhance their adoption. Many Ontario farmers already engage in crop rotation and diversification, which does not need assistance. The potential benefits of biomass may need to be supported, although likely also through policy encouragement. Afforestation does not make as much sense to forward through policy from an economic standpoint.

The findings of this study specifically apply to the corn-growing regions of the province of Ontario. As previously mentioned, the environmental effects are appropriate for similar climate zones. The economic effects could also be similar in nearby regions. However, the farming systems are most similar to the US Midwest, particularly Michigan, Ohio and Illinois. Therefore, although the economic effects are likely to be broadly similar in these areas, there are different policy, supply chain and pricing issues, including international exchange rate volatility, that make changing farm management practices on one side of the border, or the other, a different proposition.

There are different sources of economic variability in the study depending on the management practice. More specifically, there is variability in costs, yields and prices. For each practice, having variability in all three dimensions would be ideal, but this was often not available in the literature. Furthermore, there was a limit on the number of potential scenarios. Regarding the price of corn, it has been relatively steady over the last several years, making it a reasonable assumption to hold this constant. Conversely, the price of switchgrass, for example, is not well established, making variability surrounding potential prices important. The use of different dimensions of economic variability could mean that even larger ranges in net returns could be observed for many practices. However, this effect would be unlikely to change the results observed in the study, with many practices already having potential positive results and SRC, for example, being far away from economically viable.

There were also other sources of economic variability that were not included in the study. Particularly for CCs, crop diversification, SRC and biomass, the costs and benefits of these practices were not inflated to 2019 dollars. This is non-trivial as the basket of goods related to commonly available Consumer Price Index (CPI) measures are not particularly appropriate for agricultural operations over relatively short-time scales. For example, the prices of fuel, fertilizer and corn are highly variable over the last 10 years. This could mean that the costs of CCs, crop diversification, SRC and biomass are relatively underestimated, and the potential benefits may be overstated relative to the other BMPs.

Of the ten practices examined in Table 3, seven of them were found to, at least partially, exist in the upper right quadrant, representing potential economically and environmentally beneficial practices (Fig. 3). The adoption of NT and N rate reduction is firmly positive, while biomass has the largest potential economic and environmental gains. The use of inhibitors may be economically beneficial if yield gains outweigh chemical costs, but their use could shift N loss from direct (N_2O) to indirect (leaching), for example, making their environmental benefit less easy to estimate.

Conclusions

From the examination of the area-based environmental (public) and economic (private) effects of adopting ten different GHG mitigation beneficial management practices on a representative Ontario corn farm, we see that different types of practices have different scales and effectiveness. 4R nitrogen management practices are smaller in scale compared to cropping practices and, therefore, have smaller potential costs and benefits (Fig. 1). Land use and land cover change, from practices including biomass, afforestation, crop rotation and diversification, and cover cropping, had larger impacts on soil sequestration and carbon-equivalent GHG reduction, but with significantly greater costs, and production and financial risks (Fig. 2). Seven practices were found, at least partially, to represent economically and environmentally beneficial practices (Fig. 3). The adoption of NT and N rate reduction are firmly positive, while biomass has the largest potential economic and environmental gains. The use of inhibitors may be economically beneficial if yield gains outweigh chemical costs.

Although straightforward methods like two-dimensional analysis are not novel, they can be helpful when illustrating interdisciplinary connections, like those between economics and environmental studies, in integrated ways, especially at a broad overview scale. This paper spurs this line of inquiry and the approach here can easily be adopted to integrate otherwise disparate areas of study to encourage truly interdisciplinary research.

The broad scale ranges of results for multiple BMPs collected here have the potential to inform future research. Given the data collected and the ranges already assigned, future research could take those values, along with distributional data for currently static values, to conduct in depth Monte Carlo, or other forms of simulation analysis to establish more robust distributions and more closely examine the economic and environmental trade-offs. The research could be extended to multiple regions to create a truly global comparison of multiple BMPs. Future research could also consider integrating the carbon-equivalent environmental values into the economic space using, for example, carbon markets, to examine the societal benefit of practice adoption. Further consideration as to economic and mathematical maximization modeling could reveal the best suite of management practices to adopt, across regions, to most efficiently and effectively achieve environmental policy goals.

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