An economic analysis software for evaluating best management practices to mitigate greenhouse gas emissions and water pollution from cropland

by

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ABSTRACT. ***Many recent studies on soil and crop management practices have demonstrated their capability of mitigating greenhouse gas emissions (GHG) and reducing nutrient loss from cropland. The response of GHG emissions and water quality to management practices can be quantitative using biophysics-based agricultural system models. However, the economic feasibilities of adopting such management practices are yet to be evaluated, especially when producers must adopt profitable management plans. This thesis presents the development of a field-scale economic analysis software capable of estimating the net benefits under various management practices for greenhouse gas mitigation and water quality improvement. The calculated net benefits were based on the benefit-cost analysis (BCA), where GHG emissions were converted to the CO_2 equivalent and priced using the information drawn from the current carbon trade market. GHG emissions and crop yield are simulated using the Root Zone Water Quality Model (RZWQM2) which was coupled with BCA in this newly developed economic analysis software package. A case study for a cornfield at the Saint Emmanuel site near Montreal, Canada, from 2012 to 2015 under two water table management practices, i.e., free-drainage (FD) and controlled drainage (CD), showed that FD was more profitable than CD. Although less greenhouse gases were emitted under CD than under FD, the potential benefit under current carbon credit payment from GHG reduction under CD was far less than the additional cost of installing new instruments and excessive maintenance fees. The social benefit accuring from a reductionin N loss was 16 times greater than the social benefit from reduced GHG emissions. This study suggests that the government subsidy is needed to provide producers with further incentives to adopt best management practices targeting at mitigating greenhouse gas emission and improving surface water quality.***

Keywords. Carbon credit; Canada; Economic modeling software development.

Résumé

Plusieurs récentes études sur les pratiques de gestion du sol et des cultures ont démontré la capacité de celles-ci à mitiger les émissions de gaz à effet de serre (GES) et réduire les pertes en éléments nutritifs des terres cultivées. Avec un modèle informatique de système agricole basée sur la biophysique, il fut possible d'entreprendre une simulation quantitative de la réponse des émissions de GES et de la qualité des eaux aux pratiques de gestion. Cependant, la faisabilité économique de la mise en œuvre de tels pratiques n'a pas encore être évaluée, ce qui est particulièrement problématique quand les agriculteurs se doivent de choisir un plan de gestion rentable unique. Cet article présente un logiciel d'analyse économique à l'échelle du champ capable de chiffrer les bénéfices nets de divers pratiques de gestion visant à mitiger les émissions de gaz à effet de serre (GES) et réduire la dégradation de la qualité des eaux. Le calcul de ces bénéfices s'appuie sur un analyse coût-avantage (ACA) où les émissions de GES sont converties en équivalents de CO2, et leur prix est fixé selon le présent marché du commerce du carbone. Les émissions de GES et le rendement des cultures furent simulés avec le modèle de système agricole informatisé 'Root Zone Water Quality Model' (RZWQM2), jumelé au logiciel d'analyse économique permettant une ACA. Situés 60 km à l'ouest de Montréal (Québec, Canada), deux parcelles d'un champ de maïs situé à Saint Emmanuel, soumises entre 2012 et 2015 à deux modes de gestion de la nappe phréatique [drainage libre (DL) ou drainage contrôlé (DC)], servirent d'étude de cas. Le DL s'avéra plus rentable que le DC, et quoiqu'il y eût moins d'émissions de GES sous le DC que le DL, à présent le bénéfice potentiel en termes de paiements de

CRÉDITS DE CARBONE ADVENANT UNE RÉDUCTION DES GES SOUS LE DC (PAR RAPPORT AU DL), SERAIT BIEN INFÉRIEUR AUX COÛTS EXCESSIFS D'ENTRETIEN ET D'INSTALLATION DE NOUVEAUX INSTRUMENTS. LES BÉNÉFICES SOCIAUX DE LA RÉDUCTION DES PERTES EN N SONT 16 FOIS CELLES ADVENANT UNE RÉDUCTION DES ÉMISSIONS DE GES. CETTE ÉTUDE SUGGÈRE DONC QU'UNE SUBVENTION GOUVERNEMENTALE SERAIT NÉCESSAIRE AFIN D'OFFRIR DE NOUVELLES INCITATIONS AUX AGRICULTEURS À SUIVRE LES PRATIQUES EXEMPLAIRES DE GESTION QUI LEUR PERMETTRAIT DE MITIGER LES ÉMISSIONS DE GES ET AMÉLIORER LA QUALITÉ DES EAUX DE SURFACE.

MOTS CLÉS. CRÉDITS DE CARBONE; CANADA; DÉVELOPPEMENT DE LOGICIEL DE MODÉLISATION ÉCONOMIQUE.

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Chapter 1.

Introduction

The intensification of agriculture in recent decades has led to significant detrimental impacts on the global environment (FAO, 2015; Ritchie & Roser, 2020; Tilman, 1996). The most considerable impact takes place upon the freshwater and marine ecosystems (Tilman, 1996). In 2020, 78% of the global ocean and freshwater eutrophication is caused by agriculture. One of the primary drivers of eutrophication from agriculture is chemical fertilizers for crops (Ritchie & Roser, 2020; Tilman, 1996). From 1960 to 1990, synthetic nitrogen fertilizer input has increased 6.87-fold, while the phosphorus fertilizer has increased 3.5-fold. Ma et al. (2007) stated the level of input of the chemical fertilizers are often excess compare to the crop needs. As a result, the excess nutrient is lost due to either volatilization, surface runoff, or leaching towards the groundwater. Eventually, the excess nutrient would enter water systems and supply the cyanobacteria in the water bodies with sufficient nutrients to cause algal bloom and eutrophication.

On the other hand, agriculture is also a significant source of greenhouse gas emissions (GHG), contributing approximately 26% of global GHG emissions. Crop-related GHG emissions contain three main types of gases: CO_2, CH_4, and N_2O, and their emission mechanisms are complex (Liebig et al., 2005). The release of CO_2 is mostly a consequence of crops' respiration and decomposition of soil organic matter. The release of methane (CH_4) occurs when the soil is under anaerobic conditions, where the microbes produce CH_4 instead of CO_2 when decaying carbon-containing matters. Nitrous Oxide (N_2O) is produced during both nitrification and denitrification processes. Nitrification is the process that oxidizes ammonium (NH_4^+) into nitrate (NO_3^-) in aerobic conditions (Signor and Cerri, 2013), while denitrification occurs in anaerobic soils which generate nitric oxide (NO), nitrous oxide (N_2O), and molecular nitrogen (N_2) from NO_3^- (Senbayram et al., 2012).

In 2006, Canada initiated major GHG mitigation efforts through the establishment of the Climate Change Adaptation Secretariat. For Canada's GHG inventory, Environment and Climate Change Canada appointed five sectors, including the agriculture industry, energy, industrial processes, product use, waste, and land use (Kulshreshtha et al., 2015). To develop cost-benefit effective measures, researchers have proposed and evaluated GHG mitigation strategies, including on-farm and off-farm measures, to find the best management practices (BMPs) capable of reducing GHG emissions. For instance, Almaraz et al. (2009) discovered that no-till emits significantly less carbon dioxide than conventional tillage in a soybean field in southwestern Quebec. Drury et al. (2008) also demonstrated that crop rotation reduced N_2O emissions compared to the monoculture cropping system from a field study in Woodslee, Ontario.

Although many promising mitigation strategies have been proposed to reduce GHG emissions and nutrient pollution, fewer efforts have been focused on adopting such management practices in farms from the economic perspective (De Pinto et al., 2010; McCarthy et al., 2011). In other words, if a practice is not economical, its adoption would be low since farmers would not be incentivized to adopt the practice (Kulshreshtha et al., 2015; Wichel, 2007). Nevertheless, several researchers have studied various obstacles, including the potential risk of yield loss, learning cost, investment costs, variable cost, maintenance cost, and transaction costs (McCarthy et al., 2011). Thus, a decision support system will help analyze the “what-if” scenarios and promote farmers' adoption of GHG or nutrient pollution mitigating practices. Such an assessment can shorten the farmers' decision-making process regarding the adoption and provide a probability of seeking the BMPs among the available practices for a specific farm. However, most of the current economic studies related to GHG or nutrient pollution mitigating practices focus on regional level scenarios instead of field-scale scenarios. For instance, Kulshreshtha et al. (2015) evaluated the economics of various mitigations measures in Canada by adopting a "with or without" analytical framework. They suggested that soil nutrient management and grazing had the potential to achieve a "win-win" situation in Canada among practices. Similar research has been conducted in China by adopting the marginal abatement costs curve method (MACC) (Wang et al., 2014). Although these studies can deliver a general understanding of the benefit of a BMP's net economic return, they are incapable of providing a quantitative estimate at a field-scale scenario.

When attempting to forecast the potential economic outcome of a farm under a BMP, it is necessary to take GHG emission reduction as part of the private benefit. Consequently, GHG emission has already been monetarized. A precedent would be implementing a carbon tax in many countries, which also occur in provinces in Canada, such as British Columbia, as fuel gas tax. GHG emission has also even been commercialized under the establishment of the carbon market. Quebec and Alberta have implemented such a cap and trade system for GHG emissions (GOA, 2012; Government of Quebec, 2013). Compare to the carbon tax, the valuation of GHG emissions under such a system is more complicated as the reduction of GHG emissions may reflect an increase in revenue or decrease in cost, depending on whether the emitter's GHG emissions have exceeded the government's allowance. Nonetheless, carbon pricing is often distinct among countries or even provinces since separate carbon pricing and market policies are implemented in each region.

However, it is rare for researchers or policymakers to consider water quality improvement when conducting economic analysis for BMPs. The primary reason for not integrating water quality as a component is evaluating and valuating water quality improvement. First, the release of excess nutrients to the water bodies is non-point source pollution, which implies the difficulty of conducting a monitor, report, and verification (MRV) process for the released nutrient from a private owner. Secondly, neither a consistent value for nutrients, such as the social cost for carbon, nor a voluntary trading market such as the carbon market, exists.

In recent years, multiple pieces of literature have affirmed the potential economic value of water quality improvement (Crabbé et al., 2012; Dodds et al., 2009; EPA, 2015; Sena et al., 2020). The economic evaluation of water quality is often practiced on the scale of a watershed and based on three types of costs, including the willingness to pay (WTP) to remove excess nutrients from the local residents, the mitigation cost, and the economic consequences (Sena et al., 2020). Crabbé et al. (2012) simulate the monetary value of water quality improvement for the south nation river basin in Ontario to be 440,000 CAD$ per year if farmers adopt controlled drainage. Verburg (2019) reports an average value of 40.43 CAD$/kg for a Wisconsin waterway's phosphorus cleanup. Smith et al. (2019) estimate under uncontrolled condition, the future cost of algal bloom over 30 years will be 5324 million in CAD$ in Lake Erie basin, whereas controlling the blooms may only cost 2474 million in CAD$. Such studies reveal the potential to monetize and integrate water quality improvement when assessing the economic performance of a BMP. Although it may be challenging to include water quality improvement into the private benefit evaluation for farmers, the government should consider water quality as part of the social benefit for promoting farmers to adopt BMPs.

Considering that high temporal and spatial variability exists among farms, a BMP's performance is often varied upon different farms. It is common to receive contradicting results for other farms under the same BMP. To fully reveal the economic performance of a BMP, in-situ experimentations need to be conducted as premises for acquiring the results of crop yield, water quality, and GHG emission under the BMP. However, the financial burden accompanies in-situ experimentations operations since in-situ experiments require laboratory data under controlled conditions in the site and intensive labor work to collect the data. Thus, such researches are often time-consuming, costly, and difficult to establish under different temporal and spatial conditions (Fang et al., 2015). Therefore, the application of physical model simulations is indispensable, as modeling simulations require much less time and financial cost while delivering a reliable forecast of a BMP's influence.

In summary, an evaluation tool to evaluate the economic performance of various GHG and nutrient pollution mitigating BMPs, including the external costs or benefits of adopting the BMPs, is much needed. The integration of a physical model with such an evaluation tool is necessary to deliver a reliable projection.

1.1 Objectives

The objectives of this research were two-fold:

(1) to develop an economic analysis software package by combining RZWQM2 and Benefit-Cost Analysis (BCA) to access the cost and revenue of a crop farm when adopting best management practices in mitigating GHG emission and improving water quality.

(2) to demonstrate an application of the model through a case study for a cornfield near Montreal, Quebec (Jiang et al., 2019) under two water table management practices, free drainage (FD), and controlled drainage with subsurface-irrigation(CDSI).

1.2 Structure of the thesis

The thesis is structured based on chapters and is organized as follows:

Chapter 1: The background and objectives of this thesis.

Chapter 2: A literature review of the current agricultural best management practices and the field-scale models is presented. A review of current approaches to monetizing GHG emissions and water quality improvement is also included.

Chapter 3: The details of the methodology of building the economic analysis software package, including economic analysis' algorithm development and software development. The application of the economic analysis package is demonstrated through a case study at St-Emmanuel, southern Quebec.

Chapters 4 & 5: The case study results are demonstrated and discussed, including the current economic model's pros and cons and potential future upgrades.

Chapter 6: A conclusion based on the analysis and findings is presented.

Chapter 2.

Literature Review

The agriculture sector has grown to be one of the leading sources of greenhouse gas emissions and nutrient loss. Scientists seek to find sustainable management practices that can improve environmental quality and preserve farmers' current economic output. The projection of a best management practice's net benefit involves accounting for the total potential costs of adopting the practice and predicting crop yield, water quality, and GHG emissions under such practice. Considering that high temporal and spatial variability exists among farms, model simulation is indispensable. As a result, an interdisciplinary economic model covering both physical model simulation and economic appraisal simulation is essential for evaluating management practice's practicability.

2.1 Greenhouse gas emission and nutrient pollution from agriculture in Canada

The agriculture sector is one of the major contributors to greenhouse gas (GHG) emissions in Canada and is estimated to emit a total of 72 Mt CO_2 eq in 2017, representing approximately 10% of Canada's total emission of 716 Mt CO_2 eq of GHG (ECCC, 2019; Surendra et al., 2015). The expansion in agricultural production has led to a significant increase in GHG emissions since the mid-90s and recently reached around 70 Mt each year with no sign of slowing down (ECCC, 2020). Besides, agriculture is the leading contributor to methane (CH_4) and nitrous oxide (N_2O) emissions, which have 30-300 times more global warming potential than CO_2. In 2009, the agriculture sector accounted for 25% of CH_4 emissions and 72% of N_2O emissions in Canada (Kulshreshtha et al., 2015).

In terms of the eutrophication of surface water bodies in Canada, algal blooms occur in lakes such as Lake Winnipeg and Lake Simcoe and re-occurring in Lake Ontario and Lake Erie as well as other impoundments (ECCC, 2018). In 2011, Environment Canada had conducted a national-level assessment of the nutrient level in the Canadian watershed based on Environmental Canada water quality monitoring sites. Out of the 39 national locations, 30 demonstrate significant increasing nitrate-nitrite trends (ECCC, 2017). In many watersheds, agriculture is determined to be the predominant non-point nutrient source after urban point sources (Puckett, 1995).

2.2 Current Greenhouse gas emission, water quality mitigating crop management practices.

Tile drainage is a subsurface drainage method that is widely adopted by farmers in Canada. Such a strategy can prevent waterlogging by removing water from the vadose zone. Controlled drainage (CD) is a best management practice based on tile drainage but with a control structure, structure to regulate the tile outlet's height to maintain the optimum water table depth. In various in-situ experiments, CD demonstrates to effectively reduce nutrient leaching, increase crop yield, and mitigate greenhouse gas emissions (Crabbé et al., 2012; Gillete et al., 2018; Jiang et al., 2019; Sunohara et al., 2016; Tan et al., 2007).

Researchers have proposed winter cover crops as a best management practice of water and soil conservation (Hanrahand et al., 2018; Omafra, 2020). The planting of winter cover crops can utilize the residual soil nitrogen and thus improve carbon sequestration as well as N use efficiency (Basche et al., 2014). Nonetheless, some in-situ experimentations demonstrated no difference in N_2O emission when comparing winter cover crop with no winter cover crop. Mitchell et al. (2013), on the other hand, reported a contrasting outcome where the application of winter rye cover crop increased available N for denitrification, which increased N_2O emission.

No-till is a practice that plants crop seeds directly without plowing. It is suggested that such an approach can retain soil organic carbon and improve nutrient cycling (Creech et al., 2017). As a result, it is considered one of the potential mitigation strategies against global warming (Behnke et al., 2018). A meta-analysis conducted by Ogle et al. (2019) affirmed that conventional tillage contained less soil carbon than no-till.

Nonetheless, some scientists questioned the actual influence of no-till as studies had demonstrated that tillage promoted the carbon sequestration in a deeper soil profile. In contrast, no-till only promoted surface soil carbon sequestration (Angers et al., 2008; Luo et al., 2010). Data regarding carbon sequestration at deeper soil depth is limited, and more experiments are required to discover no-till's effect against soil carbon sequestration (Ogle et al., 2019).

Crop rotation, which may be a conventional management practice, but is effective in reducing N_2O emissions for corn planting. A study in Illinois conducted by Behnke et al. (2018) indicates that under a long-term soybean-corn rotation, 2kg of N2O emission per hectare is reduced compared to continuous corn. On the other hand, fertilizer application is instrumental in increasing yield, but high input of nutrients such as ammonium would increase the greenhouse gas emission and nutrient leakage in drainage. Split application of fertilizer, which is the strategy to reduce nutrient input but increase the nutrient uptake efficiency and thus achieve the goal of reducing greenhouse gas emissions and nutrient leakage. Burton et al. (2007) conducted 2-yr in-situ experimentation to discover the N2O emissions from potatoes under the split-N application. It is concluded that split-N fertilization reduced N2O emission compares to a one-time nitrogen application. A similar result is observed under a field experiment in an agricultural grassland conducted by McTaggart et al. (1997). However, some studies demonstrate no significant nitrogen reduction when applying split fertilization over a single fertilizer application (Yan et al., 2001).

A common characteristic of the management mentioned above practices is that the results from different research groups are contradictory even under a similar management practice and experimentation setup. Consequently, the performance of management practice is influenced by multiple factors, primarily spatial and temporal variability. Another generic characteristic is that those practices are double-edge blades, which can mitigate certain types of pollutants but favor the release of other kinds of contaminants. For instance, controlled drainage can decrease the level of N2O emission of a field and yields the risk of increasing the emission of CO2 (Jiang et al., 2019). Thus, further research regarding such practices is required, and model simulation may be an appropriate option considering in-situ experimentation's financial burden.

2.3 Applying Holos to perform an economic appraisal for farms

Holos, a whole farm-level model, was proposed by Janzen et al. (2006) to generate whole-farm GHG emission estimates. The model includes SLC (Soil Landscape of Canada Working Group, 2010) databases and uses simple algorithms (e.g., Emission factors) to estimate GHG emissions. Kröbel et al. (2015) enhanced its performance by integrating the Introductory Carbon Balance Model (ICBM) into the Holos model. In the latest update in 2017, Holos also included a basic economic cost-benefit analysis (AAFC, 2017). However, there are certain limitations in the estimation of a complete financial analysis of management practices. For example, the Holos model requires users to input the corresponding crop yield under a specific management practice since the model does not explicitly predict crop yield changes with management practices (Krobel et al., 2015).

Another limitation of the Holos model in economic analysis is excluding the amount of GHG emissions in the cost-benefit analysis. The potential economic benefit from GHG mitigation in agriculture can be substantial, given the current carbon market (De Pinto et al., 2010). For example, Canada has attached a monetary value to GHG by publishing the "Pan-Canadian Approach to Pricing Carbon Pollution" document as the federal benchmark for provinces to develop its carbon pricing systems and carbon market. The federal carbon tax (20 CAD$/t$CO_2$eq) is currently taking effect in Saskatchewan, Ontario, Manitoba, and New Brunswick since those provinces' carbon pricing systems did not meet the benchmark's requirement. On the other hand, provinces such as British Columbia, Quebec, Nova Scotia, Prince Edward Island, and Newfoundland and Labrador are implementing their own pricing systems (ECCC, 2020).

2.4 Current methods for evaluating GHG's monetary value.

The current carbon tax or carbon market applies only to large GHG emitters, such as large industries or electric power plants. However, the agricultural sector may take advantage of the incentive if farmers can adopt practices to decrease the amount of GHG emissions significantly. For example, the annual GHG emissions from farm operations ranged from 42 Mt to 54 Mt of CO_2 equivalent (CO_2 eq) in Western Canada (British Columbia, Alberta, Saskatchewan, and Manitoba) from 1991 to 2011. At the same time, those in Eastern Canada (Nova Scotia, Ontario, Quebec, New Brunswick, Prince Edward Island, Newfoundland) ranged from 22 to 24 Mt $CO_{2\ eq}$ annually (Dyer et al., 2018). As a matter of fact, Alberta has permitted the farmers to enter Alberta's carbon market by adopting an agricultural practice improvement since 2012 (GOA, 2020). In which a complete guideline for agricultural GHG emissions' monitoring, reporting, verification (MRV) has been established. Nonetheless, it is hard for farmers and governmental agencies to agree upon GHG emission reduction due to spatial and temporal variability due to management practices. Thus, not only should the GHG reduction be considered when conducting quantitative economic analysis, standardized practices and databases are needed in processing carbon credit payments.

Despite the potential private benefit of the reduced GHG for farmers, the reduced GHG emission's social benefits should also be considered for the government to evaluate any subsidization policies regarding GHG-mitigating management practices. The social benefits represent any co-benefits from reducing GHG emissions. Abounding researchers have noted the potential co-benefits of reducing GHG emissions from various perspectives. For instance, reducing GHG emissions prevents the negative impacts on ecosystems, including biodiversity loss, soil degradation, and ecosystem services loss (Deng et al., 2018; Harris et al., 2018; Phelps et al., 2012). From the perspective of human health, GHG emission reduction can reduce the co-emitted air pollutants and slow climate change, and an estimated 2.2 million premature deaths can be avoided in 2100 (West et al., 2013). However, such co-benefits don't contain a direct financial translation and thus needed to be estimated (UNECE, 2016). There is no current study that takes GHG reduction as a financial incentive, neither on private nor social benefits when conducting an economic appraisal of farms' management practice adoption.

2.5 The current valuation methods of water quality improvement

The potential monetary value from water quality improvement is significant (EPA, 2015; Sena et al., 2020; Smith et al., 2019). Nonetheless, it is rare to integrate water quality into evaluating the economic performance of management practice. Crabbé et al. (2012) attempted to simulate the potential financial gain from implementing controlled drainage for all cropland where controlled drainage (CD) is suitable. As the adoption of CD can significantly reduce the N and P in the runoff, Crabbé et al. (2012) estimated the social benefits of water quality improvements for the south nation river basin in Ontario to be 440,000 CAD per year. The monetary value projection is based on the economic value of the progress of the water quality index(WQI).

Sena et al. (2020) strive to monetarize the value of nutrient and nutrient pollution. Three categories of cost of nutrient pollution in water bodies are summarized. The first category is the economic consequences, which is the potential influence on various economic sectors. For instance, Dodds et al. (2009) examined the annual costs of eutrophication of U.S. freshwater systems from the perspective of the ecological goods and services (EGS), such as recreation and angling, drinking water costs for bottled water, and loss of biodiversity of a water body. As a result of eutrophication in U.S. freshwater systems, the cost is simulated to be 2.93 billion CAD annually. Smith et al. (2019) simulated the possible external cost of algal bloom in Lake Erie by examining the changes in the flows of the lake's EGS from unchecked to take action. A value of 2.8 million CAD$ reductions is simulated annually if algal bloom in Lake Erie is being controlled.

The second category of cost is based on the perspective of mitigation or restoration costs of nutrient pollution. Several U.S. studies have reported the cost of mitigating algal bloom in phosphorus excess waterbodies. Most of the studies focus on alum treatments, which are considered to be a standard phosphorus removal method (EPA, 2018). Burgdhoff & Williams (2012) applied alum treatment to Lake Ketchum as alum can permanently bind phosphorus in the water and sediment. Similar treatment has been considered by Chandra et al. (2013) in cleaning Twin Lake in Golden Valley, MN. Verburg et al. (2019), on the other hand, considers the method of sucking the muck, which is to physically remove the legacy phosphorus in the sediment of the water bodies. Based on that projection of cost and the corresponding amount of P being reduced, Sena et al. (2020) obtained the unit price of mitigating one kilogram of P in CAD, which are 6156 (Burgdhoff & Williams, 2012), 40.4 (Verburg, 2019) and 94.29 (Chandler et al., 2013), respectively. Such unit price can be utilized to estimate the monetary value from the potential water quality improvement from adopting a BMP.

Nonetheless, when considering the cost of mitigation, the mitigation practice's adverse effect should also be considered. For instance, alum treatment applications may increase dissolved aluminum, sulfate, and nitrous oxide concentration (Nogaro et al., 2013). The long-term effect of increasing such chemicals in water bodies on human health and biomass remains unclear.

Nonetheless, it has been reported that aluminum may be one of the causes of Alzheimer's disease.

The third category of cost is from the perspective of willingness to pay (WTP), which implies the amount of money a person is willing to pay for a good or service. In eutrophication, WTP can be interpreted as the maximum or minimum amount one is willing to pay for P reduction in the water bodies. Studies have been conducted in the mid-west U.S. to survey the local residents' WTP of reducing the P pollution of a water body near them (Sena et al., 2020). The WTP for P reduction ranged from 0.013 CAD $ per kg to CAD$ 6115 per kg (Sena et al., 2020). The value of WTP for P reduction falls in such an extreme range due to multiple factors, such as the geological location, the payment methods (i.e., through community taxes or simply donation), and the overall economic status. However, no such study has been established in Canada yet.

It can be observed that the projected value of water quality improvement varies significantly not only upon different regions but also in various valuating approaches. However, no current approach can provide a standard unit price that can be utilized to obtain general estimation even across different regions, such as social carbon cost.

Another issue is the missing of a totaled cost calculation of all the economic areas impacted by eutrophication. For instance, the eutrophication of a water body influences the EGS and requires measures to mitigate the eutrophication. Nonetheless, most of the current researches exclusively focus on one perspective, either the EGS, WTP, or mitigation cost (Sena et al., 2020). As a result, the potential cost should be at least a combination of both the EGS cost and mitigation cost.

Chapter 3.

Materials and Methods

To estimate the economic outcome of a farm under various BMPs, including the carbon credit/tax and water quality, and the economic algorithm is developed based on the BCA's net present value (NPV) method. Furthermore, an economic analysis modeling software is programmed based on JavaScript's electron framework to amalgamate the RZWQM2 with the economic algorithm to deliver integrated economic simulations. The projection of the social benefit of reducing GHG emissions from adopting BMPs is also implemented in the software to discover the potential of subsidization policy. The economic modeling software is applied to a case study for a cornfield in southern Quebec (Jiang et al., 2019) to explore the economic responses of adopting FD and CDSI.

3.1. Methods

3.1.1. Benefit-Cost Analysis (BCA)

The core algorithm in simulating the net economic output of a farm after the adoption of management practice is based on the Benefit-Cost Analysis (BCA) approach, as it is the only analytical framework to include all consequences, whether it's the yield or environmental quality, are considered when evaluating the adoption of new management practices (Pearce et al., 2006). The BCA analysis is the typical mainstream approach to economic appraisal, especially in an environmental project. (OECD, 2018) It is the primary analytical tool that economists employ to assess the economic efficiency of a particular policy or proposal (Kotchen, 2010; FAO, 1989). Boardman et al. (2011) defined Benefit-Cost Analysis as "a policy assessment method that quantifies in monetary terms the value of all consequences of a policy to all members of society."

Among four different types of Benefit-Cost Analysis, including the Benefit-Cost Ratio (BCR) method, the Incremental Cost-Benefit Ratio, the Net present Value (NPV) and the Payback Period, the Present Net Value (NPV) method (Eq. 1) is adopted (Zizlavsky, 2014; PAHO, 2014). The four methods share a similar concept when determining the economic efficiency of a project: to compare the sum of the benefits with the sum of the costs. However, only the NPV method can deliver a total gain or loss in monetary terms (PAHO, 2014). Considering the economic model not only targets researchers and government officials but also commercial farmers, a direct monetary output may be the most appropriate. Thus, the NPV method is adopted.

$$NPV = PV(B) - PV(C) \quad (1)$$

where

NPV = net present value

PV(B) = net present value of benefits

PV(C)= net present value of costs

The algorithm derived from the general NPV calculation (Eq. 1) may be rewritten as Eq. 2. All monetary parameters have a unit of Canadian dollar per hectare, including the final economic appraisal, NPV_t.

The economic algorithm is computed as follows:

$$NPV_t = \frac{\overbrace{P_c \times Y_t + C_{G,t}}^{Benefit} - \overbrace{(C_{a,t} + C_{p,t})}^{Cost}}{(1+i)^t} \quad (2)$$

where

NPV_t=net present value of the farm per hectare in the year *t*

P_c=sales price of the crop per hectare

Y_t =yield of the crop per hectare in the year *t*

$C_{G,t}$= carbon credit per hectare that earned from the reduction of GHG by applying the GHG-mitigating practice over the conventional practice in year *t*

$C_{a,t}$= adoption cost of a management practice per hectare in the year *t*

$C_{p,t}$= production cost per hectare of the crop in the year *t*

i= discount rate.

The private benefit includes only the revenue from selling the crops and carbon credit payment from GHG reduction in the year *t*, as demonstrated in equation 2. Crop prices from USDA and Worldbank have been included in the database (Section 2.3).

The carbon credit is computed using:

$$C_{G,t} = R_{G,t} * \Delta E_{G,t} \quad (3)$$

where

$C_{G,t}$= carbon credit of the GHG emissions reduction in year *t*

$R_{c,t}$= carbon credit rate in the year *t*

$\Delta E_{G,t}$= amount of GHG reduced by applying the GHG-mitigating practice over the conventional practice in year *t*

In the BCA analysis, the common metric is revenue (Vermeulen et al., 2016). Thus, to integrate GHG emissions into the economic analysis, the GHG emissions need to be monetarized. In our BCA analysis, Canada's carbon settlement rate in the carbon market is applied to GHG emissions, as carbon prices vary among countries or even among provinces within a country. For example, in Quebec, the provincial government implements a cap and trade system for GHG emissions (GOQ, 2013). The reduction of GHG emissions may increase revenue and decrease cost, depending on whether the emitter's GHG emissions have exceeded the allowance by the government. For simplicity, a carbon credit is issued when simulated GHG emissions under alternative management are lower than those under conventional practices. The carbon tax/credit for GHG emissions in terms of the megaton equivalent of CO_2 from several provinces in Canada and the USA is embedded in the database (Appendix, table. A1. A2).

To better compare the potential costs of adopting different GHG-mitigating practices, a farmer's annual cost is divided into two categories in the model (Figure 1). The first category is exclusively the costs that are associated with adopting management practices (adoption cost), which may vary across different practices. The second category is the typical production cost for a specific crop (production cost).

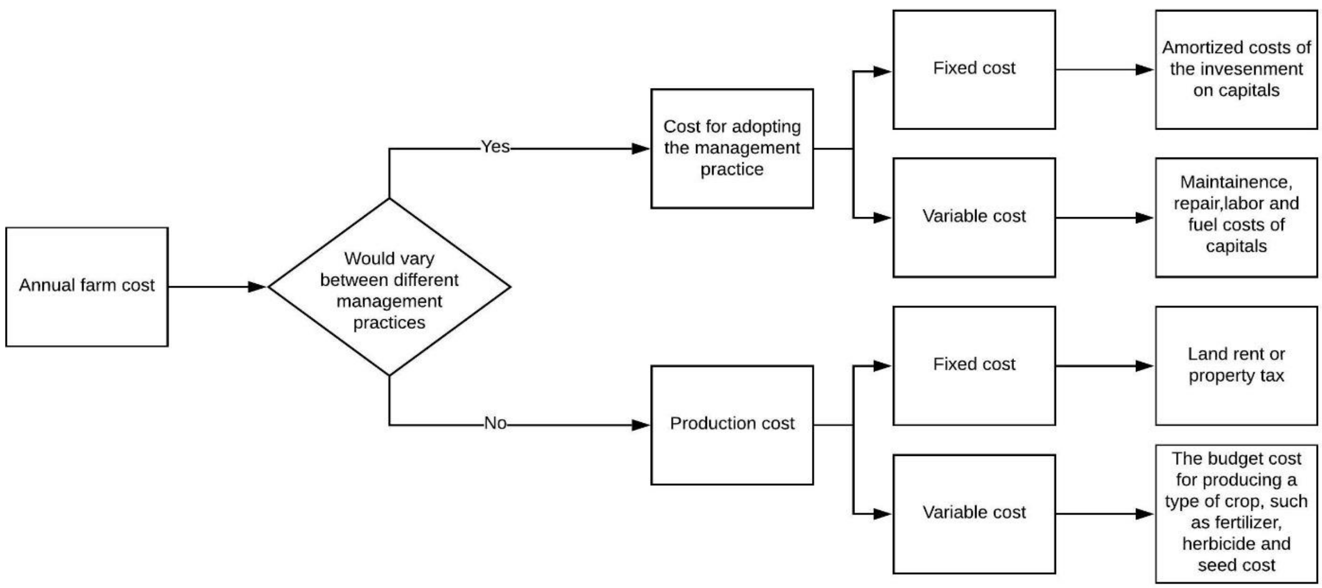


Figure. 1. Annual farm fixed cost classification's flow chart

The management practice adoption cost is computed as follows:

$$C_{a,t} = \sum_{1}^{n}(C_{c,t} + C_{m,t}) + C_{l,t} + C_{f,t} \quad (4)$$

where

$C_{a,t}$= management practice adoption cost in the year *t*.

$C_{c,t}$ = fixed cost, which is the amortized investment cost of the capital c in year *t*.

$C_{m,t}$= variable cost, which includes maintenance, repair, labor, and fuel (if applicable) costs of the capital in year t.

$C_{l,t}$ = labor cost of the capitals year t.

$C_{f,t}$ = fuel (if applicable) costs of the capitals in year t.

n= number of capitals required for adopting the practice.

The annual adoption cost is further divided into the fixed cost and the variable costs. The fixed cost is primarily the amortized investment costs of the capitals. Currently, the current database in the modeling software only contains the adoption costs for controlled drainage with subsurface irrigation (CDSI) and free drainage (FD).

The fixed cost is computed as follows:

$$C_{c,t} = \frac{C_i}{L_c} \quad (5)$$

where

$C_{c,t}$= amortized investment cost of the capital c in year t

C_i= initial investment cost

L_c= expected lifetime of the capital $_c$

The amortized investment cost C_c, for capital c, is calculated from the initial investment cost C_i divided by the capital's expected lifetime L_c. An example of such an initial investment cost on capital would be the control structure in the CDSI practice. Consequently, the presence of a control structure ensures subsurface irrigation in proper timing (Madramotoo et al., 2001).

The maintenance and repair cost are computed as follows:

$$C_{m,t} = C_{c,t} * A \quad (6)$$

where

$C_{m,t}$ is the maintenance and repair cost.

$C_{c,t}$ the amortized investment cost for capital $_c$.

A is the fixed percentage.

On the other hand, the variable costs consist of maintenance, repair, labor, and fuel (if applicable) costs, which depend on the level of usage of the capitals. In which the repair and maintenance costs are associated with each capital and are estimated as a fixed percentage of the initial investment of the capital (Evans, 1996).

The labor cost is computed as follows:

$$C_{l,t} = w_t * h_{l,t} \quad (7)$$

where

$C_{l,t}$ = labor cost in year t.

w_t = region's minimum wage per hour in year t.

$h_{l,t}$ = total labor hour that incurred by the adoption in year t.

The labor and fuel costs, if applicable, are estimated based on the anticipated usage of the management practice instead of every single capital. For example, CDSI, compare to FD, demands labor work for daily attention during the growing season and operation of the irrigation pump if needed (Evans, 1996). The labor cost is calculated by multiplying the region's minimum wage per hour per hectare with the total labor hour incurred by the adoption.

The fuel cost is computed as follows:

$$C_{f,t} = r_t * h_{f,t} \quad (8)$$

where

$C_{f,t}$= fuel cost in year t.

r_t= fuel rate per hour in year t.

$h_{f,t}$= system's demanded total operation hour in year t.

A similar method is applied to the estimation of fuel cost, which is to multiply the fuel per hectare rate with the demanded operation hours of the system. Table 1 lists the available parameters for the adoption cost in the model's database. However, users may enter their own values for other management practices.

Table 1. Parameters for available management practice adoption cost in the database. (C$/ha)

Adoption cost parameters	Component in FD	Component in CDSI	Reference
Capital initial investment cost (C_i)	Pipe	Pipe	CRAAQ, 2010, Essien, 2016
		Pump	
		Deep well	Stämpfli and Madramootoo 2006, Essien 2016
		Control structure	Tait, 1995, Essien, 2016, Crabbé 2012
		Drainage land preparation	Essien, 2016, CRAAQ, 2010
Capital Expected Lifetime (L_c)	Pipe	Pipe, Deep well, Pump, Control structure, Drainage land preparation	Evans, 1996
A fixed percentage of capital cost for	Pipe	Pipe, Deep well, Pump, Control structure, Drainage land	Evans, 1996

maintenance and repair cost (A)		preparation	
Labor hour, ($h_{l,t}$)	Doesn't require excess management	Daily attention for water level also depends on the type of control structure (i.e., flashboard type requires a change of board frequently)	Essien, 2016
Labor Rate/hr, (w_t)	N/A	Minimum wedge	Government of QC, 2020
Fuel hour, ($h_{f,t}$)	No fuel demand for FD	Pump water	Essien, 2016
Fuel rate/hr, (r_t)	N/A	kilowatt for operating the pump	Hydro Quebec, 2020

The production cost is computed as follows:

$$C_{p,t} = \sum_1^n C_v + R_t \quad (9)$$

where

$C_{p,t}$= total production cost in the year t.

C_v= cost for the variable; n is the number of variables in production cost.

R_t= rent or property tax in year t.

Similar to the cost of adopting management practice, which refers to the budget cost of producing a specific type of crop, includes fixed cost and variable cost. The fixed cost is the annual rent or property tax of the farm. No predefined values for rent or tax are stored in the database since such expenses are highly variable. On the other hand, the variable cost is expenses that are directly related to the level of production, such as seeds, fertilizer, and labor (Table 2). Among all the expenses, fertilizer is the dominant factor contributing to the fluctuation of production cost each year, as demonstrated by the fertilizer prices from the Quebec Reference Center for Agriculture and Agri-food (CRAAQ, 2019) in Table 3.

Table.2 Available variables with values in the production cost in the current database. ($/ha)

Variables	Description	Reference
Seed	Price for corn and soybean's seeds	
Fertilizer	Price for different fertilizers, see in Table 3.	
Limestone	Price for limestone and application	
Herbicides	Price for herbicides and application	CRAAQ, 2019
Transportation	Cost to transport the crop to crush plant	
Plow	Price for conventional tillage, assuming corn and soybean experience same tillage practice	
Seeder	Price for sowing the seeds	
Labor	Weeding, maintenance, and repair of machinery	Essien, 2016
Fuel and electricity	The energy cost for operating the farm	

Table 3. Fertilizer unit (kg N/ha or kg P/ha) cost from 2015-2019 from CRAAQ ($/t fertilizer,

CRAAQ 2019)

Fertilizer type	Nutrient content	2015	2016	2017	2018	2019
Calcium ammonium nitrate	27-0-0	701	670	606	608	678
Urea	46-0-0	780	702	654	665	735
Phosphate triple	0-46-0	1010	990	965	926	1013
Phosphate ammoniacal	18-46-0	911	910	822	840	930
Phosphate monomaniacal	11-52-0	905	945	833	793	N/A
Muricate de potassium	0-0-60	784	690	645	650	726

The cost-effectiveness (CEA, U_c) of reducing GHG emissions by applying GHG-mitigating BMP over conventional practice is expressed as:

$$U_c = \frac{\Delta C}{\Delta E} = \frac{C_c - C_f}{E_f - E_c} \quad (10)$$

where

E_f= GHG emission from conventional practice.

E_c= GHG emission from GHG-mitigating BMP.

C_c= The total cost of the farm after adopting GHG-mitigating BMP.

C_f = The total cost of the farm after adopting the conventional practice.

ΔE= amount of GHG reduced, from applying GHG-mitigating BMP over the conventional practice.

$\boldsymbol{\Delta C}$= increase in cost from applying GHG-mitigating BMP over the conventional practice.

After adopting a GHG-mitigating BMP, the NPV value of a farm can provide a reliable indicator of the farm's economic response. Unfortunately, it is hard to determine whether the adoption is cost-effective since the monetary benefit from carbon credit may often be little compared to the excess cost of adopting a new practice. On the other hand, cost-effectiveness analysis can evaluate the cost-effectiveness of mitigating GHG emissions under various management practices. The CEA can be compared across various management practices, and even the NPV value varies drastically from year to year. The CEA is calculated by dividing the change in cost over the change in GHG emissions. The cost-effectiveness can also be interpreted as the cost to reduce one kilogram of GHG emissions by adopting the BMP.

3.1.2. Social benefit of reducing GHG emission

The social benefits of the reduced GHG emission are estimated as follows:

$$C_{S,t} = R_{s,t} * \Delta E_{G,t} \tag{11}$$

where

$C_{S,t}$ = social benefit from GHG emissions reduction in year *t*.

$R_{s,t}$ = social price of GHG in the year *t*.

$\Delta E_{G,t}$= amount of GHG and nutrient loss reduced by applying the GHG-mitigating practice over the conventional practice in year *t*.

Aside from simulating farmer's net present value for adopting new management practices, the economic model also aims to be used as a decision-making support tool for the government to evaluate subsidization policies for GHG-mitigating management practices. Thus, the projection of social benefits from the reduced GHG emission is featured in the economic model. In our economic model, we propose the application of a social price to represent the social benefit of reducing per ton of GHG emissions to help quantify the social benefits. The social price should be aligned with the desired country's estimated social price of GHG for policy assessment to better match the local government's interest. For instance, Canada and the United States adopt the social carbon cost (SCC) approach. It is conceptually different from the marginal abatement cost (MAC) approach that reflects the cost of reducing emissions (Richard et al., 2007). SCC reflects the economic damage to the whole society that may be triggered by releasing an additional unit of carbon dioxide (Ricke et al., 2018). On the contrary, United Kingdom adopts the MAC approach for valuing GHG reduction (GOU, 2009).

The social benefit is calculated by multiplying the social price of GHG with a reduced amount of GHG emissions. In Equation 11, $E_{G,t}$ is provided from the RZWQM simulations. Currently, the social prices of GHG for policy assessment from Canada, from 2010 to 2020 are embedded in the database (ECCC, 2019. Table.4), along with researchers' estimated social GHG prices for other countries based on the SCC approach (Ricke et al., 2018; Tol, 2019).

Table 4. Canada's estimated social carbon cost

Year	Central SCC value in CAD$ per ton of CO2
2010	34.1
2015	39.6
2016	40.7
2020	45.1

3.1.3 Social benefit of water quality improvement

The social benefit from improving water quality under the BMP is computed as follows:

$$W_t = \sum_1^n (\Delta Z_{p,t} * P_p) \quad (12)$$

Where

$\Delta Z_{p,t}$ = Weight of the reduced nutrient loss of the nutrient p in kg/hectare by applying the BMP over the conventional management practice in year t

P_p = The unit price in CAD$/hectare for the reduced nutrient p in year t

W_t = The social benefit of water quality improvement in year t

n = The number of types of nutrients.

The economic model also integrates the social benefit of water quality improvement from the adoption of a BMP to abet the government to have a comprehensive understanding of a BMP's economic performance. The social benefit simulation is contingent on two parameters: the total reduced weight of the nutrient loss and the corresponding nutrient price. The reduced nutrient loss's total weight is retrieved from the RZWQM2’s simulation result.

On the other hand, the determination of the price for the reduced nutrient is complicated, as no standard unit price has been established by any governments nor officials. In theory, the reduced nutrient runoff's unit price should integrate all the possible costs that are induced by not implementing the BMP (Sena et al., 2020). The potential mitigation cost or MTP of the local residents and the EGS cost should be combined. However, current EGS costs are projected based on a watershed scale and are determined based on the WQI of the watershed. WQI is a much broader unit compare to the unit of reduced nutrient loss from a farm. Thus, it would be impossible to apply current simulated EGS costs as a unit price for the water quality improvement under a BMP unless the relationship between the reduced nutrient loss from the BMP with the change of the WQI of the watershed is well acknowledged.

WTP of the local residents to mitigate one type of excess nutrient may be a better choice than EGS cost. Consequently, when surveying the local residents, the unit can be manipulated to be the WTP of a local resident for removing a kilogram of a type of excess nutrient. However, WTP is subject to change by different regions, as other areas contain distinct characteristics such as wealthiness and education level (Mathew et al., 1999). For instance, Sena et al. (2020) report the range of WTP of mitigating per kg of phosphorus from 0.013 CAD$ to 6156 CAD$. Thus, to valuate the water quality improvement from a BMP for a farm through WTP, the WTP must be retrieved from the local residents living near the farm.

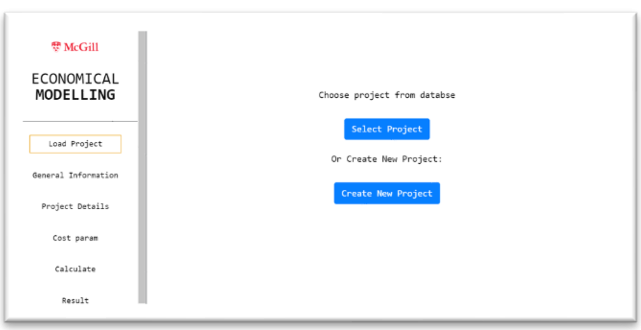
The mitigation cost for removing a specific nutrient is implemented in the model as the reduced nutrient loss price. Compare to the previous two values, and the mitigation cost can be transformed into the cost of mitigating a kilogram of the specific nutrient, it is also more generalizable compare to WTP as long as the mitigation method remains unchanged.

In the current economic model, only nitrogen is available when simulating the social benefit of water quality improvement of a BMP. Consequently, RZWQM2 only supports simulating nitrogen loss in surface runoff and tile drainage and lacks a P sub-routine (Sadhukhan et al., 2019). Although the submodule RZWQM2-P for simulating P losses is developed, it is not yet implemented in the current economic model due to time constraints. Due to the limited availability of published cost data for nitrogen removal for mitigating eutrophication, the reduced nitrogen loss price is adapted from the cost of removing nitrates through ion exchange water treatment from the Minnesota Department of Agriculture (MDA) in 2009 (Evans, 2012; MDA, 2020).

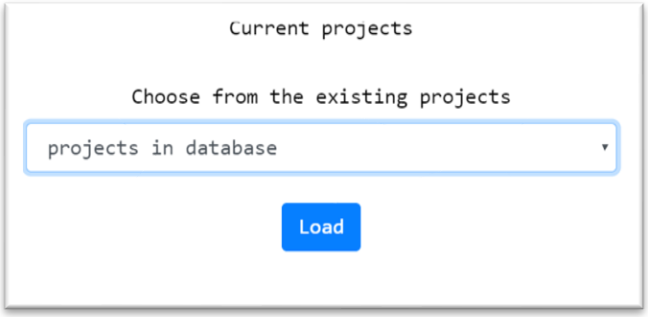
3.1.4. GHG emission simulation using RZWQM2

The RZWQM2 (Root Zone Water Quality Model 2), coupled with DSSAT 4.0 crop modules, is a comprehensive agricultural system model capable of simulating water movement, nutrient level, pesticide fate in agricultural soils along with the growth process of various types of crops under different management practices. The submodule OMNI simulates the mineralization, immobilization, nitrification, and denitrification processes of carbon and nitrogen in the soil. The model has recently been improved to simulate GHG emissions by Fang et al. (2015), who compared four different GHG emission algorithms from the DayCent, NOE, WNMM, and FASSET models in RZWQM2. Subsequently, the best algorithms for GHG emission simulation were incorporated into RZWQM2. The model performance of the improved RZWQM2 has been further validated by Gillette et al. (2017) to estimate N2O emissions under different tillage systems of irrigated corn in Colorado. Under a tile-drained corn-soybean system in Iowa, U.S. Jiang et al. (2019 and 2020) affirmed the model's performance in simulating GHG emissions from subsurface drained fields in southern Quebec and Ontario in Canada.

3.1.5. The economic analysis model and software development



(a)



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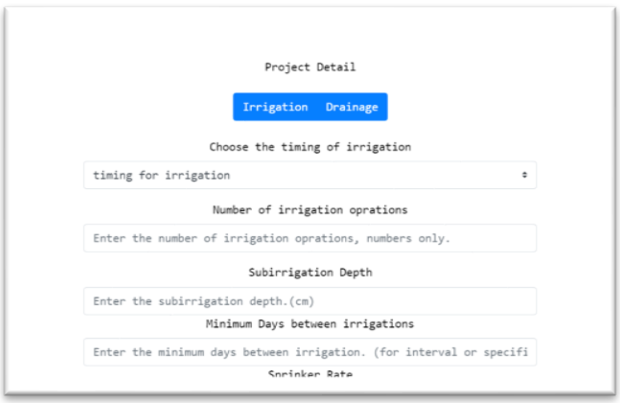

(c)

(d)

Figure. 2. The user interface of the economic modeling software (a) Choosing project from the database, (b) Checking existing projects, currently available in the database are free-

drainage(FD) and controlled drainage with subsurface irrigation (CDSI), (c) RZWQM2 parameter modification, currently can modify fertilizer plan, irrigation plan, and drainage plan, (d)Cost parameter modification, currently can modify management practice adoption cost and production cost.

The economic analysis software is developed based on the functional programming language JavaScript's cross-platform desktop apps framework: Electron (Figure 2). The software is comprised of three components, namely the frontend, backend, and the database. The frontend is also a graphical user interface (GUI) (Figure 2), which is developed using HTML and Bootstrap. The backend is responsible for handling the requests made by users from the frontend and perform corresponding actions. The database is based on the open-source relational lightweight database, SQLite3, that can be either deployed in a server or stored on a local computer. The database contains the predefined prices for different crops, capitals, and operations from the World Bank, USDA, CRAAQ, and the literature. The projects' information, users' modifications to the predefined values, and the simulation results will be recorded. A flowchart demonstrating the software's workflow, including the mechanism to interact with the RZWQM2 to acquire the projected GHG emission and yield, is shown in Fig. 3.

Yield, GHG emissions, and nutrient loss for each management practice were simulated with pre-calibrated RZWQM2 (Jiang et al., 2019). After calibrating RZWQM2 for the site, it was linked to the economic analysis software by accessing the RZWQM.DAT file that contains information on the location, land area, duration of the simulation, and management practices in the "general info" section. Upon successfully selecting the.DAT file, the software will draw all the related parameter's value, such as area, duration of the experiment, irrigation amount, and fertilizer from the.DAT file and save in the database under a specific management scenario. The software provides the interfaces for modifying most of the management practice's parameters of the RZWQM2 scenario, including irrigation scheduling, drainage details, and fertilizer application, as well as running the RZWQM2 simulation for each scenario (Figure 2.c).

After the setup of the RZWQM2 scenario in the software, the next step is to confirm/input the prices for each type of cost, including variables in adoption cost and production cost (detailed list of variables in Table 1 and Table 2) in the "cost param" section (Figure 2.d). Predefined prices will be loaded from the SQL database and displayed in the section. Upon the successful setup in the "cost param" section, users need to configure the "benefit param" prices. Similar to "cost param" section, the predefined prices for crops and carbon credit will be loaded from the database. However, considering the carbon credit varies considerably among different locations, the software requires users to choose the carbon credit from the predefined price list or manually input for each year in the experiment duration.

Similarly, the unit price of water quality improvement requires users to choose from prices in the database or manually input. Finally, users can click the "run economic analysis" button in the "calculation" screen to perform NPV projection. Results in the form of tables and graphs will be displayed on the software's final "result" screen.

A unique feature of the software is its ability to provide real-time prices for a wide range of variables. Consequently, a local database's predefined prices can only be updated when developers of the application publish a new update. Due to time and labor constraints, the economic modeling software's updating schedule maybe twice a year. Nevertheless, the prices of many variables could fluctuate within days. Therefore, despite providing the stored prices in the local database, four different web scrapers working for the crop price, climate, the carbon credit, and the currency exchange rate are implemented in the software. The web scrapers are established through

JavaScript's built-in requests module. The request module allows the software to send the request to designated APIs or websites, including the World Bank, QuandL, indexMundi, exchange rates API, Rapid-Api, and Business insider API upon each startup of the software (Table 4). The response of the request will contain the desired information, such as the most updated grain-corn's settlement price from the USDA market. The backend of the software will then handle the response and save the updated prices into the local database. As a result, users can establish an economic analysis simulation with the latest available prices.

Table 5. Model's scraping API or website links for crop price, currency exchange, climate, and carbon credit rate

Parameter	Reference	API link	Website
Crop price	Quandl	https://www.quandl.com/api/v3/datasets/	
	WorldBank		https://www.worldbank.org/en/research/commodity-markets
	IndexMundi-USDA		https://www.indexmundi.com/commodities/?commodity=
Currency Exchange rate	exchange rates API	https://api.exchangeratesapi.io/latest?base=	
Climate	Rapid-API	https://community-open-weather-	

		map.p.rapidapi.com/weather	
Carbon credit rate	Business Insider		https://markets.businessinsider.com/commodities/co2-european-emission-allowances
	Cap and trade-Cali		https://ww3.arb.ca.gov/cc/capandtrade/capandtrade.htm

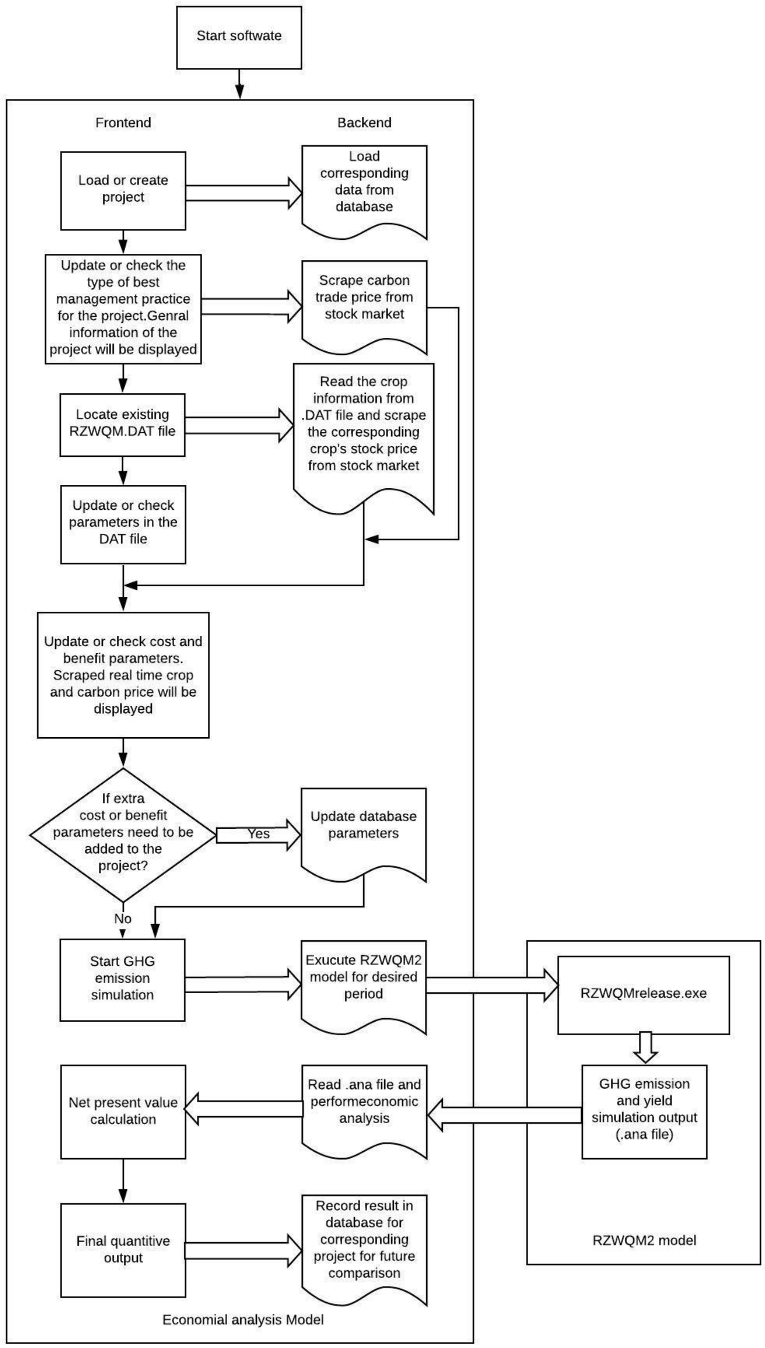

Figure. 3. Workflow of the economic analysis model based on the RZWQM2 simulated response of GHG emission to management practices. The slim arrows represent the direction of the simulating process of the frontend in the software. The large arrows represent the direction of the simulating process of the backend in the software.

3.2. Case study

This case study demonstrated an economic analysis for mitigating GHG emissions and reducing nutrient loss through water table management in a tile-drained cornfield near Montreal, Canada. Here, the RZWQM2 model was calibrated and validated by Jiang et al. (2019) against N_2O and CO_2 emissions data collected under free-drainage (FD) and controlled-drainage with subsurface irrigation (CDSI) for the field. Under the same scenario, the nitrogen loss is simulated as well. The software for economics was applied to the scenario to quantify the cost and revenue when water table management practice is adopted to mitigate GHG emissions. The net private economic return of the two management practices was then estimated based on cost-effectiveness from GHG reductions. The social benefit of GHG reduction and improved nitrogen loss by adopting CDSI over FD are also projected.

3.2.1. Field experiment site

This economic analysis is based on a field study conducted by Crézé et al. (2015) from 2012 to 2015 in St-Emmanuel, QC, Canada. The field is a 4.2-hectare subsurface-drained cornfield. The soil properties for each horizon are as follows: (0-0.25m): fine sandy loam with 5.0% organic matter, (0.25-0.55m): sand clay loam with 1.5% organic matter, (0.55-1.0m): clay layer with organic matter content. Soybean (Phaseolus vulgaris L.) was planted in 2012, and grain-corn was planted (Pioneer 9918 in 2013, Pioneer 9855 in 2014, and Pioneer 9917 in 2015) in other years. The field was divided into three equal blocks A, B, C of a width of 30 m (Figure 4). Each block was subdivided into eight identical plots (75m x 15m), and subsurface pipes were installed beneath the center of each plot at an average 1.0 m depth.

From 2012 to 2015, a conventional soybean-corn rotation was adopted. From 2012 to 2013, the gross field was free-drained. A split-plot design was established from 2014 to 2015 with two irrigation management practices, free-drainage (FD) and controlled-drainage with sub-irrigation (CDSI). Half of the plots from each block were dedicated to one of the treatments, either FD or CDSI, as demonstrated in Figure 4. Two instrument shelters were located between blocks to monitor drainage outflow from the three blocks and collect water samples. Split N application was applied based on farmers' practices. The detailed application procedure can be acquired in Crézé (2015).

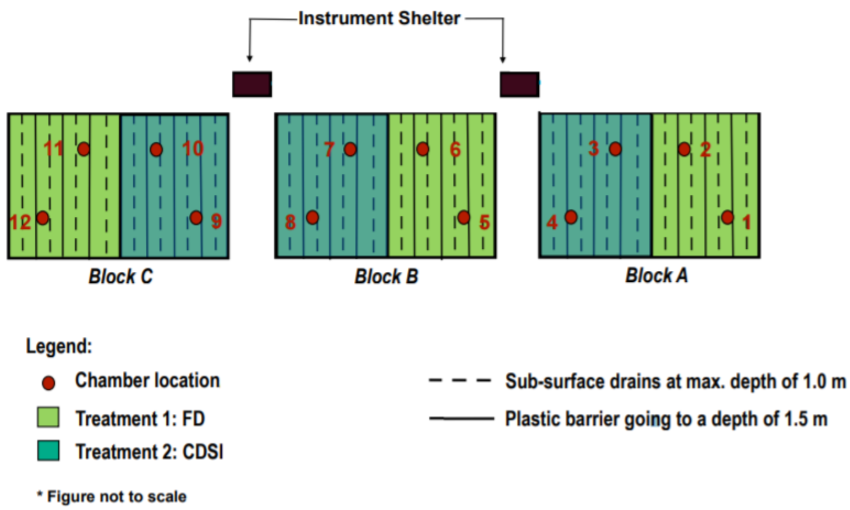

Figure. 4. Experimental plot layout of the field study in 2014-2015. In 2012-2013, all plots were under free drainage (adapted from Crézé, 2015). FD: free drainage, CDSI: controlled drainage with sub-irrigation.

GHG emissions were measured from the top chambers in each plot. A vented non-steady-state chamber method was used, which was adapted from Hutchinson and Livingston (Hutchinson et al., 2000; Hutchinson and Livingston, 2001; Livingston et al., 2006). From 2012 to 2013, all the plots were under FD treatment. From 2014-2015, half of the plots were under FD, while the other half were under CDSI.

3.2.2. RZWQM2 scenarios

The meteorology data for executing RZWQM2 were retrieved from a nearby weather station in

Côteau-du-Lac (Station ID – 7011947) from Environment Canada. The hydraulic parameters were calibrated against the observed soil moisture content from 2012 to 2013 under FD and validated using the data from 2014 to 2015 under FD and CDSI. Crop parameters were adjusted based on measured corn and soybean yields during the experiment. The nutrient parameters were calibrated according to the observed N_2O and CO_2 emissions data. The detailed calibration process and parameters can be found in Jiang et al. (2019). The RZWQM2 showed good accuracy in predicting daily N_2O emissions under FD with $|PBIAS| < 15\%$, $IoA \geq 0.68$, and $R^2 \geq 0.50$, while its predictions of daily N_2O emissions under CDSI were less satisfactory (PBIAS=13%, IoA=0.21, and $R^2=0.16$) because it failed to catch a peak of N_2O emission after a heavy rainfall event. For CO_2 emissions, the RZWQM2 accurately estimated the daily emissions under both FD and CDSI with $|PBIAS| < 10\%$, $IoA \geq 0.74$ and $R^2 \geq 0.62$. Generally, although the model tended to predict a few peak emissions earlier or later than the field measurements, the overall performance of RZWQM2 in predicting soil N_2O and CO_2 emissions should be regarded as satisfactory because it reliably estimated the cumulative emissions, which are the major concern of the current study.

Different from Jiang et al. (2019), where CDSI was not simulated in 2012-2013 in alignment with the field experiment. In this simulation study, we run RZWQM2 for both CDSI and FD for four years from 2012 to 2015 to capture the longer-term effects of drainage management effects and minimize weather effects on crop yield and GHG emissions. Under the same scenario, nitrogen loss is also simulated based on the four-year consecutive FD vs. CDSI management plan.

3.2.3. Cost and benefit prices

The adoption costs for FD and CDSI each year vary considerably due to the two management practices that demand different capitals (Table 6). The detailed calculation for pipe and its installation cost is listed in Table 7. For FD practice, only the installation of subsurface pipes land preparation is necessary. However, CDSI requires more capital, including a deep well, pump, and control structure (Table 6), because water needs to be pumped from a deep well to water control tanks to achieve a desired water table depth in the soil. As a result, it demands approximately $104 more to adopt CDSI than FD each year per hectare (Table 6).

The fixed cost in production cost, namely the rent or the property tax, is not included in the calculation due to its unavailability. However, it may not affect the comparison between FD and CDSI since the cost would be identical for both practices. The variable cost for different variables (Table 9) was retrieved from CRAAQ and The Financière Agricole du Québec (FRAQ). The fertilizer cost was calculated separately based on the fertilization rate in the RZWQM scenario from Jiang et al. (2019), as different fertilizer application rates and fertilizer formulation's price varies considerably. (Table 8) The unit price for urea in 2012, 2013, and 2015 was adopted from CRAAQ's archived documents and 2019 fertilizer and amendment report (CRAAQ, 2012, 2013, 2019). However, the urea price for 2014 was unavailable. Thus, the urea price from 2015 was used for 2014. Assuming urea was the only nitrogen fertilizer, the estimate of fertilizer costs each year was carried out by first determining the amount of urea needed per hectare and then was multiplied by its unit price to acquire the fertilizer cost (Table 9).

For the remaining variables, such as seed, spread lime, and transportation (Table 9), the prices in 2012 were derived from CRAAQ's archived soybean budget cost plan 2012 as the only soybean was planted in 2012. (CRAAQ, 2012) From 2013 to 2014, the variables' prices were adapted from CRAAQ's archived grain-corn budget cost plan in 2013 and 2014. (CRAAQ, 2012; Essien, 2016) However, the detailed expenses were not obtainable from CRAAQ for 2015. To better represent the reality, the seed price was adopted from FRAQ's 2015 grain corn budget plan while the remaining variable costs were adapted from 2014's variables costs. (Essien, 2016; FRAQ, 2015)

Table 6. Annual Management practice adoption cost for FD and CDSI

Capital	Description	Initial investment(t/ha)	Expected Lifetime (year)	Amortized cost ($/ha/year)	Maintenance and repair percentage	Maintenance and repair cost ($/ha/year)	Fuel and labor[b]	FD ($/ha/year)	CDSI ($/ha/year)
Pipe	Detail calculation can be found in Table 7	2292.6	50	45.582	2%	0.91704		46.769	46.769
Deep well	A 25-m depth deep well for the pump to pump the water to water control tanks	1337	30	44.56667	None assumed		No cost		44.56667
Control structure[a]	Simple valve control structure	205	20	47.865	2%	0.9573			48.8223
Pump	1-horsepower pump supplying water for 4.2-hectare field	957.3	20	10.25	1%	0.1025			10.3525
Land preparation	land grading for even surfaces	90.1	20	4.505	6.40%	0.28832		4.79332	4.79332
Management practice adoption cost ($/ha/year)								*51.562*	*155.3072*

[a] A control structure was implemented in 1992. The control structure includes control tanks with a weir, and float valves to control drainage and activate subsurface irrigation. The detailed mechanism of the system can be found in Tait et al. 1995 and Stämpfli and Madramootoo, 2006. The current control structure, along with price, can be found in Agri Drain, 2020.

[b] Labor: No significant daily attention is demanded in the system. Thus, labor has been included in the maintenance and repair costs.

Fuel: During experiment duration, precipitation is adequate each year. Therefore, no fuel cost since the subsurface irrigation systems were not turned on each year.

Table 7. Pipe cost calculation (including installation cost)

Pipe diameter (mm)	Price of pipe with filter ($/m)	Price of pipe installation ($/m)	Quantity (m/ha)	Pipe and installation cost($/ha)
100	1.65	1.25	601.6	1744.8
150	5.3	1.75	65.8	463.9
200	9.4	2.4	1.5	17.7
Joints				40.7
Outlet (250m)				25.5
Total pipe cost ($/ha)				2292.6

Table 8. Annual fertilizer cost from 2012 to 2015

Year	Urea price (CAD/kg), CRAAQ, 2012,2015	Kg of N ha^{-1}	Crop	Fertilizer cost ($/ha/year)
2012	0.745	70	soybean	54.6
2013	0.735	159	corn	124.02
2014	0.78	204	corn	159.12
2015	0.78	228	corn	177.84

Note: Assuming only dry urea is applied during fertilization. The unit price of urea in 2012, 2013, and 2015 from CRAAQ is adopted. As the urea price in 2014 is unavailable, its price in 2015 is applied to 2014.

Table 9. Annual variable cost in production cost for maize and soybean

Price in CAD$/ha/year		2012 (soybean)	2013 (grain-corn)	2014 (grain-corn)	2015 (grain-corn)
Supply	Seed price	206	255	261.94	269.51
	Spread Lime price	7.6	20	34	34
	fertilizer	52.15	116.865	159.12	177.84
	Herbicides price	16.63	27.56	29	29
Operation	Plow price	63.23	63.23	N/A	N/A
	Seeder price	19.92	19.5	20	20
	Transport to crushing plant	8.5	28.4	30.65	30.65
	Labor	13.2	20	18	18
	Fuel and electricity	21.6	21.6	23.6	23.6
Sum		408.83	572.155	576.31	602.6

Note: The fertilizer costs are calculated separately, as shown in table 8. The variable cost in production cost, from 2012 to 2014, is adapted from CRAAQ's soybean and grain-corn budget plans. The 2015's variable costs are unavailable, thus adapted from 2014's variable costs, except the seed price is adapted from FRAQ's 2015 grain-corn budget plan. In 2014 and 2015, the field isn't plowed; thus, the plow cost is inapplicable.

Each year, the prices of corn and soybean were derived from the US Department of Agriculture's commodity price database and converted to Canadian dollars. (USDA, 2019, Table 10). The GHG credit payment for each year in the scenario's duration was derived from the California air resources board's Cap-and-Trade Program's average carbon trade settlement price in each year (Table 10).

Table 10. Annual carbon credit rate, carbon credit for controlled drainage with subsurface irrigation (CDSI), and crop price

Year	Carbon credit rate ($/t/year)	GHG emitted FD (t/ha)	GHG emitted CDSI (t/ha)	GHG reduced (t/ha)	Carbon credit CDSI ($/ha)	Crop	Crop price ($/t)
2012	10	3.74811	4.2807	0.53	5.31	soybean	549.51
2013	11.1	2.83379	3.35115	0.52	5.72	corn	242.28
2014	11.34	2.16498	3.89743	1.73	19.62	corn	178.091
2015	12.1	2.33896	4.50899	2.17	26.22	corn	185.282

Note: The crop price is derived from USDA commodity price, 2019, for corresponding years and exchanged to Canadian dollars. The carbon credit rate is derived from the California air resources board's Cap-and-Trade Program's average carbon trade settlement price in each corresponding year.

The discount rate should be considered carefully since it can affect the final net outcome directly. Different farm owners' discount rates may vary due to the level of patience and attitude towards a management practice. Various economists have suggested different discounting rates, 6.1% to 8.2% by Burgess (1981), 8% by Treasury Board of Canada Secretariat (2007), and 3.5% by Boardman et al. (2011). Considering the model's objective to estimate the net present value, a discount rate of 5% suggested by Spiro (2010), was used for provincial government benefit-cost analysis in the current simulation.

Chapter 4.

Results

4.1. Yield and revenue from grain sale

Overall, RZWQM2 simulated similar yields between FD and CDSI, with FD's yield slightly higher than CDSI in the in all years. This result is consistent with the findings by Satchithanantham et al. (2012). Although contrary results have also been reported that CDSI contributes to higher yields than FD, water table management benefits may vary with regional climate and soils (Crabbé et al., 2012; Gottschall et al., 2016). In the first three years, the revenue from grain-sale from FD is higher than CDSI. In 2015, the gap between the revenue from grain-sale from FD and CDSI was only $46, with FD's revenue being $2896 (Table 11), while CDSI's revenue being $2848. (Table 12)

Table 11. Annual Net present value (NPV) for free drainage (FD)

Year	Yield	Revenue from crop ($/ha)	Management practice adoption cost($/ha)	Production cost ($/ha)	GHG emission (Kg/ha)	Carbon credit($/ha)	Discount rate	NPV($/ha)
2012	3452	1897	52	409	4281	0	5%	1437
2013	12440	3014	52	572	3351	0	5%	2276
2014	13850	2467	52	576	3897	0	5%	1668
2015	15628	2896	52	603	4509	0	5%	1937

Table 12. Annual Net present value (NPV) for controlled drainage with subsurface irrigation (CDSI)

Year	Yield	Revenue from crop ($/ha)	Management practice adoption cost($/ha)	Production cost ($/ha)	GHG emission (Kg/ha)	Carbon credit($/ha)	Discount rate	NPV($/ha)
2012	3183	1749	155	409	3750	5	5%	1134
2013	10857	2631	155	572	2836	6	5%	1818
2014	12026	2142	155	576	2168	20	5%	1299
2015	15373	2848	155	603	2342	26	5%	1832

4.2 GHG emissions and the carbon credit

The simulated GHG emissions in each year from RZWQM2 include CO_2 and N_2O. In 2012-2013, the GHG emission reduction was not as drastic as in 2014-2015, as the difference between the emissions was only 531kg in 2012 and 515kg in 2013 because of low precipitation. In 2014, FD emitted 3897 kg CO2 eq of GHG, while CDSI emitted 2168 kg CO_2 eq GHG. In 2015, CDSI emitted considerably less GHG than FD with the emission of 2342 kg CO_2 eq, while the latter emitted 4509 kg of GHG (Tables 11 and 12). Since CDSI does not necessarily activate under dry climate, the GHG emission reduction was not substantial in 2012 and 2013. The drastic decrease of GHG emissions in 2014 and 2015 under CDSI compare to FD was primarily due to a reduction in CO2 emissions because the CDSI resulted in higher soil water content and less soil O_2 availability for aerobic microbial respiration. The RZWQM2 simulated CO_2 emission was sensitive to the water table management, which suggested more significant CO_2 emissions under FD than CDSI (Jiang et al., 2019). Due to the positive relationship between carbon credit and GHG emission reduction, CDSI's reduction in GHG emissions also increased revenue from carbon credit payment.

4.3. NPV and the cost-effectiveness analysis

In all years, due to the additional fixed cost in adoption cost from CDSI for the new capitals and FD's slightly higher annual yield, the NPV of FD was higher than CDSI at an average of an extra $294/ha each year. Since the carbon credit in 2012 and 2013 for CDSI was only $5.3/ha and 5.7$/ha, the difference in NPV between the two practices was significantly higher than that in the latter two years. From 2013 to 2014 and 2015, CDSI emitted significantly less GHG than FD, resulting in a significant increase in carbon credit from $6/ha to $20/ha and $26/ha (Table 12). Such reduction also abated the narrowing of the gap of CDSI's NPV to FD's NPV.

In terms of the NPV, FD had overperformed CDSI (Figure 5). However, from the perspective of environmental stewardship, CDSI reduced more GHG emissions than FD. Accordingly, from the RZWQM2 simulation, both years applying CDSI demonstrated reductions in GHG emissions compared to FD. Considering the adoption cost was significantly higher than the potential carbon credit, a comparison of the cost-effective analysis (CEA) was also carried out to determine the cost-effectiveness of adopting CDSI over FD from 2012 to 2015 (Table 13).

Table 13. Cost-effectiveness analysis of GHG reduction by adopting Controlled drainage with subsurface irrigation (CDSI) over Free drainage (FD)

Year	E_c (kg/ha)	E_f (kg/ha)	C_c (CAD$/ha)	C_f (CAD$/ha)	ΔE (kg/ha)	ΔC (CAD$/ha)	U_c (CAD$/kg/ha)
2012	3750	4281	564	461	531	104	0.2
2013	2836	3351	727	624	515	104	0.2
2014	2168	3897	731	628	1730	104	0.06
2015	2342	4509	758	655	2167	104	0.05

Note: E_c is the GHG emission from adopting CDSI, E_f is the GHG emission from adopting FD, C_c is the total cost of adopting CDSI, C_f is the total cost of adopting FD, ***ΔE*** *is the decreased amount of GHG emission by applying CDSI over FD,* ***ΔC*** *is the increase in total cost by applying CDSI over FD, U_c is the cost-effectiveness in mitigating GHG emission by applying CDSI over FD.*

The CEA result can be interpreted as the cost of reducing one kg CO_2 eq of GHG emissions by adopting CDSI over FD, as shown in Table 12. The cost from 2014 to 2015, compared to the cost from 2012-2013, not only decreased drastically but also was comparatively cheaper than many other policies evaluated by Gillingham and Stock (2018). Admittedly, the adoption of the CDSI increased extra fixed costs and a reduction in NPV. Nevertheless, from the perspective of cost-effectiveness, CDSI demonstrated its value of adoption.

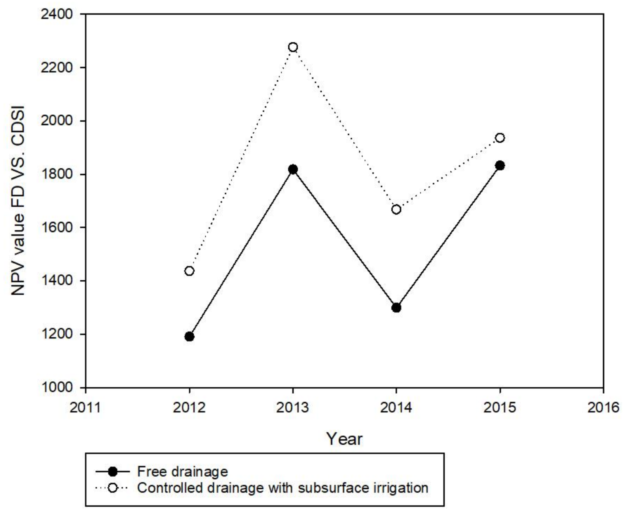

Figure. 5. Free drainage's annual net present value vs. controlled drainage with subsurface irrigation's annual net present value from 2012 to 2015

4.4. Social benefit of reduced GHG emission and government intervention

As stated in the previous sections, the government's intervention is a critical factor in adopting GHG-mitigating practices by farmers (De Pinto et al., 2010; Kulshreshtha et al., 2015; McCarthy et al., 2011). Although CDSI demonstrated its potential in reducing GHG emissions as well as provided a potential monetary benefit from carbon credit payment, the fact that it may lead to a decline in farmer's net use is undeniable. Even under the simple assumption that farmers are risk-neutral and would adopt one practice when higher net present value can be achieved (Antle, 2002; De Pinto et al., 2010; Gonzales-Estrada et al., 2008; Stavins, 1999), the adoption of an alternative practice still demands the practice to be profitable. Thus, additional payment made by the government is essential in persuading farmers' adoption.

For providing subsidies for such projects, the government often needs to evaluate the social benefits. The gap between the CDSI and FD's NPV can be considered as the least expected subsidy. The social benefit of the reduced GHG emissions by adopting CDSI over FD was projected by adopting the average of Canada's social carbon cost (SCC) price for CO_2 equivalent in 2010 and 2015 (Table 14). The average SCC was 37 CAD$/ton, which was 26 CAD$ higher than the carbon credit rate of 11 CAD$/ton, indicating the presence of a positive externality. (ECCC, 2019)

The additional positive externality of reducing GHG emissions was inadequate for covering the expected subsidy. In the first three years, the gap between the social benefits of reduced GHG emissions and the predicted subsidy was considerably large, as the social benefits can only cover 10% of the expected subsidy. In 2015, the social benefits can cover nearly 77% of the expected subsidy. Although the percentage of the social benefits over the expected subsidy in the four-year average is only 26.5%, another co-benefit exists from implementing the CDSI practice. The adoption of CDSI yields a positive effect on the water quality, as CDSI can significantly reduce the N and P runoff (Crabbé et al., 2012; Lalonde et al., 1996; Saddat et al., 2018). Crabbé et al. (2012) estimated the social benefits of water quality improvements for the south nation river basin in Ontario to be 440,000 CAD$ per year, assuming all cropland where controlled drainage is suitable is under controlled drainage. The potential economic value of water quality improvement should also be estimated and considered when the Quebec government evaluates the subsidy policy for CDSI.

The government of Canada has released "The low carbon economy fund" in the "Home and buildings" sector, which aimed to leverage practices that can (1) generate clean growth, (2) reduce greenhouse gas emissions, and (3) help meet or exceed Canada's Paris Agreement commitments (GC, 2020). The CDSI fulfills three prerequisites and has high cost-effectiveness in reducing GHG emissions, and may bear multiple co-benefits. It is rational to request an allocation of a similar fund in the agriculture sector. Currently, for farmers in the case study to achieve the same NPV as the old FD practice after adopting CDSI, the government needs to provide a stipend of

CAD$249/ha each year, which may be possible in the near future.

Table 14. Controlled drainage with subsurface irrigation's (CDSI) social benefit from reduced GHG emission in St-Emmanuel

Year	Social price for GHG/ton	GHG emission reduced t/ha	Social Benefit $CAD/ha	Expected subsidy for CDSI $CAD/ha	Percentage of social benefit over expected subsidy
2012	37	0.53	20	246	8 %
2013		0.52	19	458	4%
2014		2	64	369	17 %
2015		2	80	104	77%

Note: The expected subsidy is the difference in NPV between FD and CDSI.

4.5 Social benefit of water quality improvement and future actions

Table 15. Controlled drainage with subsurface irrigation's (CDSI) social benefit from reducing N loss in St-Emmanuel

Year	Reduced N from surface runoff $kg^{-1}\ ha^{-1}\ yr^{-1}$	Reduced N from tile-drainage $kg^{-1}\ ha^{-1}\ yr^{-1}$	Price of the reduced N loss CAD$ kg^{-1}	The social benefit of reduced N loss CAD$ $ha^{-1}\ yr^{-1}$
2012	0.25	26.6	49.5	1329.8
2013	0.011	26.6		1319.6
2014	0.89	13.1		693.2
2015	0.028	16.5		817.9
sum	1.2	82.9		4160.4

Note: The reduced N from surface runoff and tile-drainage is obtained through subtracting RZWQM2's simulated N loss in surface runoff and tile drainage under free drainage with RZWQM2's simulated N loss in surface runoff and tile drainage in controlled drainage with subsurface irrigation.

Aside from the social benefit of reduced GHG emission from adopting CDSI, the possible social benefit from water quality improvement is also simulated based on the same RZWQM2 scenario. From 2012 to 2015, 21 $kg^{-1}\ ha^{-1}\ yr^{-1}$ of total N loss reduction is simulated if CDSI is adopted over FD. As a result, a complete social benefit of 4150.4 CAD$ (Table 15) from 2012 to 2015 is projected for adopting CDSI based on the cost of removing nitrate in water through ion exchange water treatment. As section 4.4 has stated, the social benefit of reduced GHG emissions can hardly cover the average stipend of 249 CAD$ that needs to be issued to farmers to reach break-even states after adopting CDSI. Nevertheless, a contracting result is demonstrated if the reduced N loss's social benefit is integrated into the economic appraisal for CDSI. The four-year average total social benefit is 1085.9 CAD$ ha^{-1} yr^{-1} (Table 16). As a result, the social benefit from implementing CDSI can not only completely cover the stipend of 249 CAD$ ha^{-1} yr^{-1} for farmers but also yield an extra benefit of 836 CAD$ ha^{-1} yr^{-1}.

Table 16. Controlled drainage with subsurface irrigation's (CDSI) total social benefit

Year	Social benefit reduced N loss CAD$ ha^{-1} yr^{-1}	Social Benefit of reduced GHG CAD$ ha^{-1} yr^{-1}	Total social benefit CAD$ ha^{-1} yr^{-1}
2012	1329.8	20	1349.8
2013	1319.6	19	1338.6
2014	693.2	64	757.2
2015	817.9	80	897.9
Average	1040.1	45.8	1085.9

Compare to the social benefit gained from reduced GHG emission, and water quality improvement has demonstrated more significant economic potential. It is primarily due to the weight difference in the reduction of the two types of pollutants. Under the same kg^{-1} ha^{-1} yr^{-1}, the weight of reduced N loss is 20.7 in a four-year average, whereas the weight of reduced GHG emission is only 1.3 in a four-year average. Furthermore, the current price applied for the reduced N loss only represents the mitigation cost for N, and neither the EGS cost nor the mitigation costs for other pollutants such as phosphorus are included.

The social benefit from reduced N loss once again proved the economic value of reducing the nutrient input and improving water quality from adopting CDSI. However, as multiple literature has stated, the data for the financial assessment of nutrients based on a water quality improvement perspective is limited and unorganized (Sena et al., 2020; Smith et al., 2019). EGS cost assessment should be conducted for all major water bodies experiencing eutrophication in Canada, such as Lake Saint-Pierre, Lake Winnepeg, and Lake Simcoe. A single consistent value of a nutrient should be carried out by government officials to better promote and encourage mitigating nutrients pollution from agriculture and other aspects such as urban waste and industrial waste. (Sena et al., 2020)

Chapter 5. Discussion

5.1. Model implication

A substantial obstacle that has been described in the literature when attempting to deliver an appraisal of the economics of adopting GHG and water quality mitigating practices is the lack of information, such as lack of the original GHG emission amount, changes in input usages (De Pinto et al., 2016), or even local-applicable agroforestry options (Christianson et al., 2016; McCarthy et al., 2011). Management practice costs data in this area is also scarce, including the costs of enforcement, monitoring, management, organization, and negotiation of practices (De Pinto et al., 2010). The combination of RZWQM2, the economic analysis software, and its SQLite database may potentially solve the issue. As the software's users' number increases, simulations for different farmers' scenarios in various regions under various management practices can be carried out. The simulation outputs, both from an economic analysis model and RZWQM2, will be recorded in the database and may become available to other users under the original user's grant. As data exchange becomes widely available, farmers have easy access to identify the BMPs for their farms.

5.2. Model and price uncertainties

Model simulation errors and price uncertainty are inevitable during economic analysis. The influence of parameter fluctuation from the RZWQM2 model and the price uncertainty are treated dissimilarly. Considering the instability of parameters from a physical model such as RZWQM2 would have a non-linear effect on the GHG simulation. The impact of prices' uncertainty on the final economic projection is linear. Jiang et al. (2020) established a sensitivity analysis for GHG emissions by adopting RZWQM2 and found that field capacity at 1/3 bar was the most sensitive parameter for simulating both N2O and CO2. For instance, when the field capacity increased by

25%, the N2O emissions would increase by 302%, while CO2 would decrease by 12.2% (Jiang et al., 2020). Although the fluctuation of N2O appeared to be considerable, its actual influence on the final economic output was minimal because N2O only attributes a small percentage of the total GHG emissions in a field. Taking the St-Emmanuel case study as an example, the average N2O emissions per hectare were approximately 2.3kg ha-1, where the average CO2 emissions were approximately 3.9 Mg ha-1 (Jiang et al., 2019). As a result, simulated GHG emissions due to error in field capacity was around roughly 12.1%. Considering the carbon credit was minimal during the entire economic analysis, the 12.1% flocculation from simulated GHG emissions had no substantial effects on the final economic outcomes with only 0.2% error.

Among all the different price parameters, crop selling price is the single largest factor, whether under costs or benefits. As a result, the crop price fluctuation may have the most significant influence on economic outcomes. In the USA, both corn and soybean's price has decreased by approximately 50% from 2011 to 2020. In our case study scenario, a reduction of 50% in corn and soybean prices can result in approximately 48% of the final economic output change on a four-year average. However, the economic output fluctuation maybe even more considerable in reality since the change in prices is often a chain reaction. Other prices, such as labor and fertilizer costs, would also be affected. Overall, the price parameter uncertainty would have a more consequential effect on the economic analysis outcome compared to RZWQM2 model parameters.

The price uncertainty of water quality would also substantially influence a BMP's social benefit projection. In the case study, the simulated social benefit from reduced N loss is 16 times more than the social benefit of reduced GHG emission. The water quality rate only comprises the mitigation cost for nitrogen and missing the potential EGS costs. Nonetheless, it is rather difficult to quantify the influence of the price uncertainty of water quality to the economic appraisal, as no single consistent value for nutrient exists. Aside from the mitigation cost for nutrients, the EGS cost is much region-specific, similar to WTP. Consequently, various water systems would provide distinct ecosystem services, thus the economic impact of potential eutrophication among the water systems varies considerably.

5.3. Model limitations

Although a successful economic comparison has been established via the economic software application, calibrating RZWQM2 scenarios (i.e., relative root means square error less than 30%, PBIAS less than 15%, etc.) is a highly complicated process (Ge et al., 2017). Farmers can't use the model, but agro-consultants may learn how to use the RZWQM2 model with standardized model input databased for various agro-ecosystems. Such constraints can undoubtedly limit the applications of economic modeling software to many farms. Another major limitation comes from the errors of RZWQM2 predictions, which result unquestionably in uncertainties of economic analysis that affect the decision-making process. Consequently, since the GHG emissions and the subsequent carbon credit are based on the RZWQM2 simulations, only reliable RZWQM2 simulations can guarantee a legitimate economic return of the practice. Besides the model performance, the uncertainties from both social and economic systems (*e.g.,* prices of oils) need to be further addressed to expand the applicability of the economic modeling software.

Furthermore, parameters that should have been integrated, such as risk costs and monitoring, reporting, verification (MRV) costs, are absent due to either insufficient data, challenges of performing the quantification process, or time constraints. Current literature often assumes a simple condition in terms of adopting GHG-mitigating practices, which is that farmers are risk-neutral, and they would adopt an alternative practice if it can yield significant net revenue (Antle, 2002; Gonzáles-Estrada et al., 2008; Stavins, 1999). Nonetheless, several studies have concluded that farmers tend to be risk-averse instead of risk-neutral (Antle, 1987; McCarthy et al., 2011; Serra et al., 2006), especially in developing countries (Liu, 2013; Tankaya et al., 2013). Therefore, risk cost is a substantial parameter in a BCA analysis for the evaluation of adoption.

A mature MRV system for GHG emissions is essential to the successful establishment of an agricultural carbon market. Nonetheless, the implementation of an MRV system would induce a considerable amount of cost. Data-collecting cost during monitoring is a befitting instance. Consequently, acknowledging the amount of annual carbon sequestration or GHG emission flux is the premise of adopting any physical or economic model. However, the collection of such data is a costly and complicated process, and the investment cost on the installation varies among regions and various methods (De Pinto et al., 2010). Many other costs similar to data-collecting cost that would be induced by implementing the MRV system exist, as multiple pieces of research have emphasized. (De pinto et al., 2011; Tang et al., 2018; Wang, 2011) Thus, when evaluating the net economic returns of management practices assuming farmers would enter the carbon market, integrating MRV costs into the model is inevitable.

5.4. Feasibility of implementing a MRV system for the agriculture carbon market in Canada

In spite of the costs of implementing an MRV system, the feasibility of establishing an MRV system for the agriculture carbon market is often questioned, especially for the GHG emission estimation. (De pinto et al., 2010) Most of the problems that occur during the process of implementing MRV for estimating GHG emissions can be attributed to imperfect MRV guidelines, as well as the poor governance and enforcement of the MRV guidelines.

On the other hand, Canada has demonstrated the potential to be feasible for establishing an agriculture carbon marker. Consequently, provinces in Canada are capable of publishing and enforcing MRV frameworks for the carbon market. (ECCC, 2019) In the present, legal MRV frameworks for carbon markets have already been established in provinces including Quebec, Nova Scotia, British Colombia, as well as Alberta. It is common in the four provinces to submit a report annually, and accredited third-party organizations must verify the report. (CCNS, 2016; GOQ, 2016; GOB, 2019; GOA, 2020) Although specific protocols for agriculture industry programs lack in most provinces, agriculture-related measurement protocols, such as the Conservation cropping protocol, have already been published in Alberta. (GOA, 2012) Legal enforcement of the MRV framework has also been instituted to cinch the successful implementation of the frameworks. All four provinces' enforcement has demonstrated that noncompliance with the framework can result in serious financial penalties and even up to 18 months in jail in Quebec. (CCNS, 2016; GOQ, 2016; GOA, 2018; GOB, 2019)

Another Canada's crucial advantage in establishing agriculture compare to many other developing countries is the mature agriculture regulation framework. Various regulation has been set for different agriculture programs. For instance, in Quebec, detailed guidelines for fertilizer applications for crops, such as the fertilization rate for a specific crop, must be followed. Such a framework allows the local government to screen out the credible farmers who comply with the regulations over a long period of time to lower the noncompliance rate during the implementation for MRV for GHG estimation.

The accomplishment of the MRV for GHG estimation also depends on if appropriate subjects have been chosen from the GHG-mitigating perspective. Some of the farmers, naturally are not suitable for such GHG-mitigating program. Researchers have provided various models for evaluating the potential of reducing GHG emissions for a specific farm, such as RZWQM, Holos, and DNDC. The economic model that's presented in this paper also provides a solution for evaluating the possibility for a farm to adopt GHG-mitigating mitigations from an economic perspective. The combination of screening suitable farms and creditable farmers at the same time can abet the successful implementation of MRV for GHG estimation, thus allowing the agriculture carbon market to be feasible in Canada.

5.5. Model's future upgrades

Due to project time and workforce constraints, some features can only be implemented in future updates. The deployment of an online database may be the most desirable feature to increase its ability to collect data of various GHG-mitigating practices globally. Currently, a local SQLite database is included when downloading the software package. Implementing a common online database of the software provides an opportunity for users to share their databases and access others, such as the economic outputs of adopting a new BMP. The sharing of the data promotes new types of GHG-mitigating practices feasible for farmers. As a result, a meta-analysis can be performed to summarize the most cost-effective practices in a region.

Another crucial upgrade in the future would be the software's compatibility with other physical models capable of projecting GHG and crop yield, i.e., DNDC. Fundamentally, economic modeling software depends on outputs from the models to perform a separate NSPV computation. The ability to run multiple physical models will enable users to assess the uncertainty introduced due to model simulation errors. The other necessary updates are to include additional parameters in the NPV algorithm, such as risk costs and monitoring costs.

Chapter 6.

Conclusion

In this study, an economic analysis modeling software was developed and linked to a physical model (RZWQM2) for evaluating the economic feasibility of GHG-mitigating management practices. The results from the case study of projecting the economic appraisals for two different water table management practices in a soybean-corn rotating field in southern Quebec from 2012-2015 demonstrates the economic software's ability to provide quantitative monetary outputs. The RZWQM2 model simulates the CDSI emits 30% less GHG compare to free drainage (FD) from 2012 to 2015. The average annual carbon credit under CDSI from the GHG reduction is 14.2 CAD$/ha. The economic model's simulated average annual NPV for FD and CDSI would be 1829 CAD$ /ha and 1520 CAD$/ha. The NPV for FD is, on average, 24% higher than the CDSI annual net present value in the first three years, and only 6.5% percent higher in the last year. Although CDSI does emit less GHG emission than FD, the additional benefit through carbon credit is nominal compared to the other management practice adoption cost for adopting CDSI. Thus, CDSI's net present value is lower than FD throughout 2012-2015.

Successful cost-effectiveness analysis can be carried out and compared between different desired practices. The simulation result in the case study is much in agreement with the literature that GHG-mitigating practices are worthy of adoption from reducing GHG emission. Nonetheless, the excess cost from implementing GHG-mitigating BMP demands the governments to provide an incentive for such an ecosystem service. The simulated social benefit from reduced N loss by implementing CDSI over FD is 16 times greater than the social benefit of reduced GHG emission.

The developed economic analysis software is an initial attempt to provide economic assessment for farmers for adopting various management practices. Despite its limitations, the model offers both farmers and governments an opportunity to evaluate GHG-mitigating practices from an economic perspective. Hopefully, the current model's weaknesses and the present economic limitations in implementing GHG-mitigating practices will stimulate further discussion and advancement for simulating various management practices' economic outputs.

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Appendix:

Table.A1 Carbon Tax price for each province in 2019 (based on gasoline)

Provinces	Carbon Tax Price per tCO2e
Saskatchewan	20$
Ontario	20$
Manitoba	20$
New Brunswick	20$
British Columbia	40$
Quebec	22.3$
Nova Scotia	4.23$
Prince Edward Island	20$
Newfoundland and Labrador	20$
Alberta	30.3$
Yukon	20$
Nunavut	20$
Northwest Territories	21.3$

Table.A2 California Air Force Carbon settlement price:

year	price
Nov-19	16.8
Aug-19	16.85
May-19	17.4
Feb-19	15.62
Nov-18	15.33
Aug-18	14.9
May-18	14.53
Feb-18	14.53
Nov-17	14.76
Aug-17	14.55
May-17	13.57
Feb-17	13.57
Nov-16	12.73
Aug-16	12.73
May-16	12.73
Feb-16	12.73

Nov-15	12.65
Aug-15	12.3
May-15	12.1
Feb-15	12.1
Nov-14	11.86
Aug-14	11.34
May-14	11.34
Feb-14	11.38
Nov-13	11.1
Aug-13	11.1
May-13	10.71
Feb-13	10.71
Nov-12	10