Spatially explicit modeling of wetland conservation costs in Canadian agricultural landscapes

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Abstract

Agriculture is an important source of food, employment and tax revenue to society. However, agricultural expansion is an important driver of global natural ecosystem degradation, including wetlands. Economic theory shows that wetland loss is caused by a mismatch between the private wetland conservation costs borne by landowners and the public benefits generated. We develop a spatially explicit wetland management model to estimate the private economic benefit of wetland drainage in an agricultural landscape in Alberta, Canada. We estimate a full wetland supply curve and show that the private economic benefits of wetland drainage are highly heterogeneous within a watershed. We then combine these private costs of wetland conservation with non-monetary measures of public ecosystem benefits to assess four wetland conservation policy targeting scenarios. We find a positive correlation between the opportunity cost of wetland conservation on private landowners and the amount of environmental benefits wetlands offer, suggesting that conserving the wetlands that impose the lowest opportunity cost may not be optimal targets for wetland conservation policy. We contribute to wetland conservation economics by demonstrating that targeted wetland conservation policies can be more effective than a uniform conservation policy that assumes wetlands within agricultural landscapes have the same costs and benefits.

Keywords: Wetland Conservation, Farm-simulation model, Policy Targeting

1. Introduction

Agricultural landscapes are comprised of a mosaic of cultivated and developed land and undeveloped land with native vegetation cover. Increases in agricultural production, which is important for food, unemployment and tax revenue to many economies in the world, have come, in many cases, through the conversion of these natural ecosystems to managed agricultural systems. Wetlands are one example of a productive and diverse natural ecosystem (Keddy et al. 2009). They provide important ecosystem services that benefit society including the removal of excess nutrients, sediment and pesticides from agricultural runoff, sequestration of atmospheric carbon, water storage, release and recharge of aquifers, habitat for wildlife and biodiversity (Davies et al. 2008; Badiou et al. 2011; Gleason et al. 2011; Pattison et al. 2011; De Groot et al. 2012; Dias and Belcher, 2015; Vymazal, 2017). Wetlands make up about 6.1% of the worlds land surface, excluding Antartica (Junk et al. 2013; Reis et al. 2017). Unfortunately, the future health of wetlands is uncertain. After reviewing 189 reports on wetlands, Davidson (2014) has shown that the long-term natural wetland loss have averaged about 64-67% since 1900 AD. Moreover, actions to protect and conserve wetlands, which varies across the world regions, are relatively uncommon (Davidson, 2014).

Land conversion is one of the main threats to wetland ecosystems in agricultural landscapes (Davidson 2014). For example, 40 – 71% of wetland areas in the Prairie Pothole Region (PPR) of North America have been lost since European settlement (Tiner, 1984; Environment Canada, 1986; Kraus, 2019) with agricultural development a major driver (Badiou et al. 2011; Watmough and Schmoll, 2007). Also, according to Watmough and Schmoll (2007) about 202,342.8 ha of wetlands have been lost in Canada's PPR, of which about 125,452.5 ha

(62%) have been converted to agricultural land. Economic theory shows that wetland drainage on agricultural landscapes can be explained as a mismatch between private costs and public benefits of wetland retention and conservation. Wetlands within areas of annually cultivated land impose significant private costs in the form of foregone crop production, or opportunity costs, and nuisance costs, motivating agricultural producers to convert these areas to annual crop production (Cortus et al. 2011; De Laporte 2014). At the same time, the ecosystem services wetlands provide are enjoyed by society at large rather than just the agricultural producers. These relationships are exacerbated with increases in food demand and the associated increase in agricultural commodity prices and intensified production systems resulting in the incentive to convert wetlands on agricultural lands to increase, especially on those lands most productive for annual crop commodities (Lawley, 2019).

The PPR is a physically and economically heterogeneous landscape where the majority of wetlands are on private lands and farmer participation in wetland conservation programs is usually voluntary (Bruneau, 2017). As a result, wetland conservation objectives can only be reached if the policy measures are attractive to prospective landowners. In this context, it has been proposed that the targeting of conservation policy could be an effective approach to meet the conservation objectives with a lower budget or conserving wetlands with higher ecosystem values to society (which could be at a higher cost to society). This study will help to develop an approach to inform wetland conservation policy that targets those wetlands that are at higher risk of drainage due to their economic context and/or importance of their ecosystem services to society. Specifically, we design a spatially explicit wetland conservation cost model to estimate the private net financial returns from wetland drainage in the Canadian Prairies. We apply this model to a sub-basin in the Vermillion River Basin in Alberta, Canada as a case study. The main results of this study are a) private wetland conservation cost is heterogeneous within the study area, b) wetlands that have low conservation/retention costs (low opportunity cost) are more likely to offer less environmental benefits; in this case, the choice of policy targeting tool (or a mix of them) becomes important in achieving a conservation goals for a given conservation budget.

This paper contributes to the limited body of research that has examined the cost of wetland conservation or retention to agricultural producers. The majority of the previous research has focused on the field or farm scale where these costs are relatively homogeneous (Gelso, 2008; Cortus et al. 2011) or estimated for a uniform, average cost of wetlands across the landscape (see, for instance, Lawley and Towe, 2012 and Lawley, 2019). We extend the representative farm approach used by Cortus et al. (2011) and De Laporte (2014) to estimate a supply curve for wetlands in the study watershed. The estimates of wetland cost heterogeneity can help to inform wetland policy initiatives and policy targeting.

This paper is comprised of five sections. Section two provides background information on wetlands, including wetland conservation programs and wetland drainage on agricultural lands, in the Prairie Pothole region of North America. Next, the methods used to explicitly estimate the spatial heterogeneity in wetland conservation costs in the study area are discussed in section three. The results of the study are presented and discussed in the context of the literature on wetland conservation costs, in section four. Finally, the conclusion of the study, limitations of the study and suggestions for future research are provided in section five.

2. Background information on Wetlands in the PPR

The Prairie Pothole Region (PPR) of North America covers an area of 770,000 km^2 (Doherty et al. 2016) through three Canadian provinces (Alberta, Manitoba, and Saskatchewan) and four U.S states (North Dakota, South Dakota, Iowa, and Minnesota) (Brunet and Westbrook, 2011). Historically, wetlands, which are mostly characterized by depressional areas that receive water by local snowmelt or rain (Doherty et al. 2016), represented about 23% of the area in the PPR landscape (Euliss et al. 2006). According to Brunet and Westbrook (2011), wetlands in PPR are usually not greater than 15 ha, often less than 1.5m deep, and have a density of 5-90 per km^2. It should also be noted that wetlands are often not isolated basins with many being hydrologically connected through a process called fill and spill, in extremely wet periods (Spence, 2006).

Several studies have reported significant wetland loss across all three prairie provinces in Canada since European settlement (e.g. Kraus, 2019). For example, Euliss et al. (2006) estimated an annual wetland loss in Alberta of 0.5 percent. The policy context for this wetland loss includes, at the federal level, the government being responsible for wetland management only on federal lands, representing approximately 29% of wetlands in Canada (Rubec and Hanson, 2009). In Canada, the primary responsibility to manage wetlands lies with the provinces themselves (Rubec and Hanson, 2009). Wetland policies developed in each of the prairie provinces have distinctly different priorities with the Alberta Wetland Policy focusing on “no net loss” of wetland function and area by providing guidelines and directions on wetland management to sustain the benefits wetlands provide to the environment, society and the economy (Rubec and Hanson, 2009). In contrast, the Saskatchewan Water Management

Framework aims to manage wetland areas and their ecosystem functions by providing wetland drainage licenses to farmers who follow provisions of the Saskatchewan Watershed Authority Act (Rubec & Hanson, 2009). The Manitoba Water Right Regulation was amended in 2019 to require a license to drain lower risk wetlands that includes a requirement to mitigate and compensate for the loss of less permanent wetlands and prohibition of drainage of more permanent wetlands (Government of Manitoba, not dated).

Where wetlands are located on private lands, participation in wetland management programs is mostly voluntary which challenges the effectiveness of government policies to meet wetland conservation objectives (Farnese and Belcher, 2006). Policy instruments that provide monetary compensation to landowners are often viewed as a promising approach (Lawley and Towe, 2012). Claassen et al. (2001) argued that compensation programs could also allow for greater flexibility in wetland conservation than regulatory policies. For example, conservation easement programs provide compensation to landowners who agree to retain habitats (such as wetlands) and are present in several provinces in Canada through various conservation agencies, such as Ducks Unlimited Canada (Bruneau, 2017; Lawley, 2019). Other payment-based programs that have been implemented or proposed to conserve wetlands on private lands include land trusts, enhanced forage conversion and land purchases (Lawley, 2019).

2.1. Economics of Wetland Conservation Cost

Surface and subsurface drainage are the two main water management approaches by which the surface water contained in wetlands on agricultural landscapes are drained in the PPR of North America (Cortus et al, 2011). Wetland drainage is mainly aimed at moving surface water from potentially productive agricultural land. From the perspective of landowners who have wetlands on their private lands, to motivate them to conserve wetlands a necessary level of compensation would be approximately equal to the opportunity cost of the land in commodity production minus drainage cost. For instance, according to Heimlich (1994), enrollment in the Wetland Reserve Program (U.S wetland conservation program) required conservation easement payments at least equal to the capitalized net-returns from agricultural production in perpetuity, plus additional compensation for administrative cost and/or disruption to normal farming operations. Heimlich's (1994) analysis supports the assumption that the forgone net-returns from converting wetlands to croplands could serve as a proxy for private wetland conservation cost, at the margin.

Published estimates of wetland conservation costs have been based on implicit prices of permanent easements, since an easement restricts the conversion and development of the wetland basin, decreased land values capture the forgone agricultural production (Shultz and Taff, 2004). Shultz and Taff (2004), using a hedonic regression method, estimated the implicit price of wetlands under permanent easement to be \$133/ha and under non-eased permanent wetlands to be \$67/ha in southeastern North Dakota, US. Using a propensity matching method, Lawley and Towe (2014) showed that wetlands under permanent easement reduced farmland prices by \$37/ha in Manitoba, Canada. Using a discriminative reverse auction procedure, Hill et al. (2011) estimated the average cost of restoring a drained wetland in the Assiniboine River Watershed of east-central Saskatchewan to be \$294/ha/year for a 12-year contract.

A theoretical perceived cost of wetlands on agricultural lands was proposed by Gelso et al. (2008) as the difference between the certainty equivalence of cultivating cropland with no wetlands to cropping a field with wetlands. It is assumed agricultural producers perceive the presence of wetlands on agricultural lands to negatively impact profits, and this perceived cost increases with wetland area and having smaller wetlands dispersed on the cropland. Gelso et al.

(2008) estimated a perceived cost function using a contingent valuation method with data from farmers in Kansas, U.S, and showed that: 1) additional area of wetland in a quarter-section increases perceived cost by $0.24/ha, 2) additional costs imposed with wetlands dispersed in four separate areas instead of one is $1.75/ha, and it is $4.35/ha if wetlands are dispersed in 16 separate areas instead of one. In contrast, Cortus et al. (2011) used a dynamic stochastic cash flow simulation model to estimate the net private benefit of wetland drainage by a representative grain producer in Saskatchewan, Canada. They found that the annual net-private benefit of wetland drainage ranged from $38.35/ha on a 260 ha farm to $41.696/ha on a 1,295 ha farm.

3.0. Methodology

The study area is a sub-basin which covers an area of 156.91 km^2 or 242 quarter sections[1] in the Vermillion River Basin of Minburn County, Alberta, Canada (Figure 1). We chose this study area as we had access to good data coverage.

Figure 1. Distribution of Wetlands in the Area and their Distances to Watercourse.

The 2010 Alberta Merged Wetland Inventory (Alberta Environment and Parks, 2010) was used to spatially identify wetlands in the study area. The distribution of wetlands in the study area, including their distances to watercourses, is shown in Figure 1. We assumed that wetland basins that are located closer to a watercourse had lower drainage costs due to a requirement for less earth movement to establish drainage infrastructure. Flow outlet is the location where water from within the identified watershed exits to the adjacent landscape; it is generally the lowest point of the watershed.

[1] One-quarter section equals 160 acres or 65 hectares.

3.1. Modeling Wetland Conservation Costs

Our study adds spatially explicit cost information to the methodology of Cortus et al. (2011) to develop watershed-scale estimates of the net-private benefit of wetland drainage . For this study the net private benefit of drainage, or wetland conservation cost, is defined as the present value of net-returns from producing annual crops on the drained wetland basin (assuming a canola-spring wheat rotation), less the cost of wetland drainage, over 50 years. For our analysis, we assumed a canola-spring wheat crop rotation based on the fact the two crops are the dominant varieties cultivated in this region of Alberta, Canada (Alberta Agriculture and Forestry, 2016) and the canola-spring wheat crop rotation is a common crop management approach. In the canola and spring wheat rotation, odd year revenues will be based on canola yield and price and even years will be based on wheat yield and price. Information on crop yields, crop prices and production costs that will be used to estimate the the present value of net-returns from producing annual crops is obtained from Alberta Agriculture and Forestry (2021).

To estimate the present value of a 50-year crop production planning horizon, we used the capital market line theory approach (Sharpe et al. 2000) to estimate a risk-adjusted discount rate of 8.3%[2] which was used to discount the annual net returns of the canola and spring wheat rotation. Many studies have used the capital market approach to estimate discount rates in agricultural settings. For instance, Cortus et al. (2011), based on this approach, used a discount

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[2] We followed Sharpe et al. (2000) to estimate the risk-adjusted discount rate.The standard deviation of crop returns (2004 to 2018) which is $27/acre was estimated using data from ministry of Agriculture, Alberta (https://open.alberta.ca/publications/agriprofit-cost-and-return-benchmarks-for-crops-and-forages-irrigated-soil-zone). The expected market returns (2004 to 2020) from the Canadian stock exchange (TSX) and standard deviation are 5.6% and 16%, respectively, and were calculated using data froml Yahoo Finance (https://finance.yahoo.com/quote/%5EGSPTSE/history?period1=299462400&period2=1622678400&interval=1mo&filter=history&frequency=1mo&includeAdjustedClose=true). The expected market returns from yearly Canadian government issued treasury bills (2004 to 2020) which is 1.69% was estimated using data from the Bank of Canada (https://www.bankofcanada.ca/rates/interest-rates/t-bill-yields/).

rate of 10% for wetland drainage in Saskatchewan; Koeckhoeven (2008) estimated a discount rate of 7.5% for crop and cattle production in Alberta. To enable us to compare our wetland conservation cost estimates with the results in Cortus et al. (2011), a similar study, we chose a 50-year time horizon because our risk-free interest rate estimate was lower than that in Cortus et al. (2011) with a 20-year time horizon. The formula for estimating the wetland conservation cost \$/ha for a 50-year period (i.e. opportunity cost of the land) in the study area is provided below, Equation (1):

$$NPB_i = \Big(\sum_{o=1}^{49}(pr * a_i * (pc * w_i * yc - cc) * (1+r)^{-(o)} + \sum_{e=0}^{48}(pr * a_i * (pw * w_i * yw - cw) * (1+r)^{-(e)} - a_i * DC_i\Big)/a_i \qquad (1)$$

where,

$NPB_i =$ Net present benefit of the ith drained wetland (\$/Ha)

$pr =$ Probability of harvesting after drainage

$w_i =$ Relative Soil productivity weight for the ith wetland

$a_i =$ Area of the ith wetland drained (Ha)

$r =$ Private discount rate = 8.3%

$o =$ Odd years; year 1 to year 49

$e =$ Even years; year 0 to year 48

$pc, pw =$ Price of canola (\$673.5 per Tonne), price of spring wheat (\$294.2 per Tonne)

$yc =$ Average yield of canola (3.1 Tonnes per Ha)

$yw =$ Average yield of spring wheat (4.4 Tonnes per Ha)

$cc =$ Average cost of production of canola (\$822.2 per Ha)

$cw =$ Average production cost of spring wheat (\$659.3 per Ha)

$DC =$ Drainage cost per Ha of drained wetland

The expected spatial heterogeneity in wetland conservation cost, Equation (1), is driven mainly by the heterogeneity in crop yield and wetland drainage cost in the study area. Moreover, Equation (1) is estimated given certain parameters, in particular, crop prices, cost of production, probability of harvesting, surface drainage cost and crop yields. In a sensitivity analysis, using a Monte Carlo simulation method, we assess how changing the probability of harvesting, cost of production, and crop prices influences the NPV. The Monte Carlo simulations incorporate uncertainty into the analysis and can assess how changes in the parameters affect the NPV results. We assume triangular distributions for the parameters and Table 1 presents the minimum, average, and maximum values used for each parameter. We use 1,000 draws from these distributions and calculate Equation (1) for each of these draws. The main analysis incorporates uncertainty for surface drainage cost and crop yields. Next, the sources of heterogeneity in crop yield and drainage cost are discussed in sections 3.2 and 3.3, respectively.

Table 1. Triangular Distributions Assumptions

3.2. Spatial Heterogeneity in Net-returns from Cropping Drained Wetlands

Net returns from converting a wetland to annual crop production can be measured as the net crop revenue (crop price multiplied by the yield minus all production costs). In this study, we assume crop yield is heterogeneous, with site specific soil productivity driven primarily by the heterogeneity in specific soil characteristics, including soil nutrients, soil texture, and topography. Agricultural land assessment values in Alberta represent the potential productivity of soils for agricultural production (Alberta Municipal Affair, 2018) and we use these values as a measure of the heterogeneity in land productivity. The mean land assessment value within the study area was $278.43/ha, with a minimum (maximum) value of $17.30/ha ($895.12/ha). The interquartile range and standard deviation of land assessment values were $143.02/ha and $105.24/ha, respectively.

A relative soil productivity weight was used to account for the heterogeneity in crop yields across the landscape in the study area. To ensure that the variability in soil productivity followed that of crop yield variability in the study area, the relative soil productivity weight was derived from the land assessment values and the variability in canola yields in Alberta. The variability in canola yield was used as a proxy for yield variability within the study area. Specifically, the proportions of the maximum and minimum yields of canola, in Alberta, relative to the average yield of canola were used to constrain the relative soil productivity weight to lie between 0 and 1.2, and to reflect the variability in canola yield in the study area. Using this approach, soils with lower land assessment values will have lower relative soil productivity weights, which are closer to the ratio of minimum yield to the average yield of canola. The opposite will be true for soils with higher land assessment values. Table 2 shows the regional benchmarks on annual crop production in Alberta for black soil zone. In the absence of specific information, we make the assumption that the soil within a drained wetland basin would have similar productivity characteristics as the soil in the surrounding quarter section.

Table 2. Regional Benchmarks on Annual Crop Production in Alberta, Canada

The relative soil productivity weights for wetlands in the study area were estimated using the following equation, Equation (2).

$$RSW_i = ((LAV_i - MinLAV)/(MaxLAV - MinLAV)) * (MaxY - MinY) + MinY \qquad (2)$$

where: RSW_i is the relative soil productivity weight for the ith quarter section, LAV_i is the land assessment value for the ith quarter section, MinLAV (MaxLAV) is the minimum (maximum) land assessment value for the study area, MinY (MaxY) is the proportions of the minimum (maximum) yield of canola in Alberta relative to the average canola yield in Alberta.

Moreover, when wetland basins are converted to annual crop production, the basin, which represents a low area within a field, may be more prone to recurrent wetness than the adjacent upland which could delay seeding and the date when the crop is mature, or in very wet years, preclude seeding. To address this we assumed that the drained wetland basins would be too wet in two out of ten years while successfully producing annual crops eight out of ten years. Therefore, we included a probability of harvesting parameter of 0.8 to reflect this condition; however, the uncertainty of this assumption is addressed by testing the sensitivity of this assumption in a robustness check.

3.3. Spatial Heterogeneity in Drainage Cost

Surface drainage is the dominant form of wetland drainage in the PPR and we assume surface drainage as the basis for estimating drainage cost within the study area. Drainage cost is assumed to be a one-time investment that would enable drainage of the wetland basin for 50 years, with routine maintenances and rehabilitation of drainage infrastructure. To represent the spatial difference in drainage cost we further assume that cost is influenced by the relative distance of a wetland to a watercourse since wetlands closer to watercourses will require less earth movement with shorter and possibly shallower ditches that are relatively easier and cheaper to construct. We calculated the relative distance variable as the ratio of the distance of a wetland to a watercourse (meters) to the average distance (meters) of all the wetlands to watercourses in the study area. The range of values of the relative distance is from 0 to 3.99 (which are unitless). Wetlands with relative distances less than one represents wetlands located closer to the watercourse compared to the other wetlands in the study area. In contrast, a wetland with a relative distance value of 3.99 would represent a wetland that is farther from the watercourse compared to the other wetlands.

A drainage cost function was developed to estimate the surface drainage cost per hectare for the ith wetland, and it is the sum of the fixed cost and the variable cost of surface drainage, Equation (3). We assumed surface drainage fixed cost of \$494/ha for the 50-year lifetime of the drainage project. The fixed cost of drainage represents all other costs that could be associated with surface wetland drainage, including administrative costs and maintenance/rehabilitation costs. The variable cost component of the surface drainage cost is the product of the relative distance variable and the distance coefficient. A value of 277.2 was chosen for the distance coefficient such that the maximum (minimum) drainage cost across the study area will be \$1600/ha (\$494/ha), Equation (3); since the minimum and maximum relative distances of wetlands to watercourse are 0 and 3.99, respectively. Cortus (2005) used surface drainage costs of \$234/ha to \$1547/ha in 2021[3] for his wetland drainage study in Saskatchewan. The range of surface drainage cost in Cortus (2005) partly informed us to choose a maximum surface drainage cost of \$1600/ha in our study, and also to reflect the potential differences in costs between Saskatchewan and Alberta.

$$DC_i = FC_i + 277.2 * RD_i \tag{3}$$

where: DC_i is the drainage cost for the ith wetland; FC_i is the fixed cost of the ith wetland, and RD_i is the relative distance of the ith wetland to a watercourse.

3.4. Wetland Ecosystem Function Benefits

We use Alberta's Relative Wetland Value (RWV) Spatial Information (Creed and Aldred, 2015) as a measure of the ecosystem benefits of a wetland. The RWV categorizes all wetland basins located in a given quarter section using a four-level scale (A, B, C or D) whereby

[3] We converted the 2005 \$/ha to 2021 \$/ha by multiplying the ratio of the average of 2021 monthly (up to May) consumer price indezes to the average of 2005 monthly consumer price indexes (which is 1.3) to the surface drainage cost in 2005. The consumer price index data was obtained from https://www.bankofcanada.ca/?p=39863 , which was assessed on July 27, 2021.

A represents the highest ecosystem values and D represents the lowest ecosystem values. The ecosystem values are assessed on four characteristics: 1) hydrological health function, 2) water quality function, 3) ecological health function, and 4) human use function of the wetlands. As the names of the characteristic imply, the RWV measures ecosystem functions and not the relevant ecosystem services that can form the basis for monetary valuation. Lacking more detailed information on ecosystem services, we use the RWV metric as a proxy.

The RWV provides information on the areas of A, B, C, or D wetlands in each quarter section within the study watershed. From this information, we computed a variable (beneficial wetland ratio) as the ratio of the sum of the area of wetland A or B in a quarter section to the total area of all wetlands (A, B, C, and D) in the same quarter section; we assumed that a beneficial wetland ratio greater than 0.5 would mean that the wetlands in a given quarter section are likely to be of high ecosystem values. Therefore, we identified wetlands in the study area to be highly beneficial wetlands (that is either A or B wetlands) if they were located in a quarter-section with a beneficial wetland ratio of at least 0.5. Using this criterion, about 369 wetlands (with a total of 52.3 ha were identified as highly beneficial wetlands. Using this same approach we identified approximately 94 and 3,115 wetlands as C and D wetlands respectively. The number of C wetlands were relatively low, therefore these were grouped with the D wetlands to form beneficial wetlands (C and D wetlands) with a total of 413.3 ha.

Given the heterogeneity in wetland costs and benefits across the landscape, it appears that the targeting of wetland conservation efforts may be important to meet conservation objectives. We assess four different wetland conservation targeting scenarios:

a) high benefit targeting scenario– aimed at conserving wetlands with relatively high ecosystem functions (A and B wetlands),

b) least cost targeting scenario –aimed at conserving wetlands with relatively low conservation cost, and

c) low benefit targeting scenario– aimed at conserving wetlands with a relatively low ecosystem functions (C and D wetlands),

d) No targeting – aimed at randomly conserving about 50 percent of wetlands in a quarter section.

To characterize the range and pattern of wetland conservation costs in the study area we applied Equation (1) to the natural resource conservation targeting scenarios. We rank the costs of conserving each wetland from the lowest cost to most expensive to derive the wetland supply curve. The supply curve show the potential cumulative area of wetlands that could be conserved in the study area for a given "price" which covers the marginal cost of conserving each additional wetland.[4] The 95 percent confidence interval lines presented around the supply curves were generated using the "boot" package in R (Kirby and Gerlanc, 2013); the package estimated the sample mean of wetland conservation cost in 100,000 bootstrap replicates. Also, the 95 confidence interval was estimated assuming a normal distribution of the replicates.

Given a wetland conservation budget, a common policy objective might be to prioritize the conservation of highly beneficial wetlands (A or B) since they would provide relatively more valuable ecosystem services to society. The total wetland conservation cost for all the beneficial wetlands in this study (369 wetlands), would be the maximum estimated wetland conservation cost, Equation (1), for all the highly beneficial wetlands multiplied by the maximum of the cumulative sum of the wetland area for the 369 wetlands. The cumulative sum of wetland area would be computed by ranking the estimated wetland conservation cost for all the wetlands in

[4] The costs to conserve each wetland are average field costs, and these might be different from the marginal costs for each wetland.

the study area in descending order. This amount would be the potential wetland conservation budget for the beneficial wetlands in the study area. We adopt this wetland conservation budget as the total budget in the other targeting scenarios to ensure that the results are comparable at the wetland conservation budget level.

To identify wetlands in the cost targeting scenario, we ranked the estimated wetland conservation cost for all the wetlands in the study area in descending order and computed its cumulative sum. We selected wetlands for the cost targeting scenario such that the total wetland conservation cost for the nth wetland in the ranking (for all wetlands in the basin) would be equal to the wetland conservation budget.

Similarly, we selected C and D wetlands for the low benefit targeting scenario such that the wetland conservation cost for the nth wetland in the ranking (for all wetlands in the C and D wetland set) would be equal to the wetland conservation budget. The same procedure would be used to select the wetlands in the no targeting scenario, after about 50 percent of wetlands for each quarter section in the study were randomly selected. This random selection of wetlands could mimic situations where regulation requires producers to conserve 50 percent of wetlands in the sub watershed based on reasons that are not correlated with conservation costs. The sampling process was completed using the “Sample” function in the R statistical package.

We compute the correlation between the least cost wetland and low benefit wetlands using the “crosstable_statistics function” from the “sjstats v0.17 package” in R version 4.0. The “crosstable_statistics” function computes the Phi correlation coefficient which is appropriate for measuring the degree of correlation between dichotonomous variables (Akoglu, 2018) such as least cost wetlands, high benefit wetlands, and low benefit wetlands. A positive correlation stastistic, for instance, between low cost wetands and low benefit wetlands would mean that wetlands that have low net present values are also less likely to offer low environmental benefits.

4.0. Results and Discussion

4.1. Heterogeneity in Wetland Conservation Costs

About 38.3% of the wetlands have negative wetland conservation cost (net present value); this means that producers will not have an economic incentive to convert the wetland areas to cropland. Therefore, for this section we present the results on the wetlands with positive wetland conservation cost. The mean annual net present value (NPV) is \$2,404/ha. The standard deviation of the NPVs is \$2,796/ha, which indicates that the wetland conservation cost is heterogeneous within the study area. The maximum NPV is \$19,617/ha. We rank all the wetlands from lowest to the highest cost and we illustrate the resulting supply curve for wetland conservation in Figure 2; since wetlands with negative NPV are likely not to be drained we change their wetland areas to 0 ha and NPV to \$0/ha.

Figure 2. Supply Curve for Wetland Conservation in the Study Area.

The positive slope of the supply curve (Figure 2) shows that wetland conservation cost (or payment) has a positive effect on the total wetland areas that could be conserved in the study area, all things being equal. Moreover, the slope of the supply curve increases along the curve (from left to right on the x-axis) which shows that changes in wetland conservation cost changes as the total wetland area conserved decreases along the curve (or the elasticity of the supply curve reduces along the curve). The elasticity of the supply curve represents the percentage change in wetland area conserved given percentage change in wetland conservation cost. For instance, for a 100 percent increase in wetland conservation cost, for example from $2,500/ha to $5,000/ha, an additional 140 ha of wetlands could be conserved for $1,325,000, but when wetland conservation costs increases from $5,000 to $10,000 (100% increase) an additional 70 ha could be conserved for $2,650,000. Therefore, as we move along the supply curve there is an increase in the marginal cost of conservation, or, fewer wetlands could be conserved by society at a higher cost.

The average annualized wetland conservation cost is $187/ha[5]. Cortus (2005) estimated the average annual net benefit of wetland drainage to be in the range $50/ha - $60.2/ha in 2021 dollars (which increased with farm size). Any potential difference in our estimate and Cortus (2005) estimates could be attributed to differences in crop rotation ued to estimate the values, productivity of eastern Saskatchewan compared to eastern Alberta, and changes in production technology over time. Again, the unit of crop yields (tonne/ha) and crop prices ($/tonne) in our study are different from Cortus (2005) which were bushel/ha and $/ha, respectively, and this shows how economic changes could drive cost changes over time. Also, Lawley and Towe (2014) estimated that wetlands under permanent easement programs in Manitoba, Canada, reduced farmland prices by $48/ha in 2021 dollars. The conservation cost of wetland in other studies are $237/ha[6] (Shultz and Taff, 2004) and $341/ha[7] (Hill et al. 2011). Therefore, we feel that our estimate could be in the higher region of the range of wetland conservation costs in the Prairie Pothole region of North America. However, average or representative wetland conservation cost estimates only capture a small part of the information presented in the spatial

[5] Following Macháč et al. (2016) The formula for the estimating the annualized wetland conservation cost (AnWCC) is $AnWCC = NPB * (r * (1 + r)^{50})((1 + r)^{50} - 1)^{-1}$, where NPB is net present benefit of drainage for 50-year period; r is discount rate (8.3%).

[6] A conversion factor of 1.37 (ratio of 2020 CPI to 2004 CPI) was used to convert US$133/ha (the implicit price of wetlands under permanent easement in North Dakota) to US$182/ha, and then to C$237/ha using an exchange rate of 1.3 (C$/US$). US CPI data was obtained from https://fred.stlouisfed.org/series/USACPIALLAINMEI.

[7] The cost of restoring a drained wetland in the Assiniboine River watershed in eastern Saskatchewan.

heterogeneity of wetland conservation costs in a landscape (Figure 2). Therefore, these estimates should be used with caution to inform a wetland conservation policy.

The sensitivity analysis results show that, on average, canola and wheat commodity pricesm probability of harvest, and cost of production have significant effects (positive) on NPV estimates (Figure A2 in appendix). This means that future changes in crop commodity prices, in particular, would have positive influence on the areas of wetlands that are converted to croplands, which is consistent with the observation in De Laporte (2014).

4.2. Wetland Conservation Policy Targeting

Unlike a wetland conservation policy targeting approach, Figure 2 does not show the differences in wetland conservation costs for conserving different classes of wetlands, such as wetlands with high environmental attributes and/or low economic values (NPVs). The summary statistic results (Table 3) of the wetland conservation targeting in this study show that the average wetland conservation cost in the least-cost targeting scenario is $189/ha, which is lower than the average cost in the other targeting scenarios, including the high benefit targeting scenario. The average conservation cost in the high benefit targeting scenario is the largest ($2,322/ha). However, there is variation in the wetland conservation cost estimates for the scenarios.

Table 3. Summary Statistics of Wetland Conservation Targeting Scenarios and the Status Quo (Nontargeting)

The supply curves of the wetland conservation targeting policies (high benefit targeting, no targeting random scenario, low benefit targeting and least cost targeting) are shown in Figure 3. Similar to the supply curve for all the wetlands (status quo scenario) in Figure 2, the elasticities of the supply curves decrease along the curve from left to right. Therefore, the observation in Figure 2 that as we move along the supply curve (from left to right) fewer wetlands could be conserved by society and at a higher cost, is also true for the supply curves for the targeting scenarios. However, the supply curve for the highly beneficial targeting scenario is more elastic than the other two, which show that significantly less wetland area could be conserved at a significantly higher cost than in the other two scenarios.

Figure 3. Supply Curves for Wetlands in the High Benefit, Random, Low Benefit and Least Cost Targeting Scenarios.

When examining the correlation between wetland conservation cost and wetland benefit, about 9% of the wetlands identified as least cost were also high benefit wetlands (A and B wetlands) while approximately 91% of the wetlands identified as least cost were also low benefit wetlands (C and D). Further, a Phi coefficient of 0.07 between (significant at 1%, p-value < 0.001) between the two dichotonomous variables, least cost and low benefit wetlands, show that wetlands that are identified as least cost are more likely to offer low environmental benefits for the study area; the result shows that choosing the right policy targeting tool becomes important in achieving a conservation goal for a given conservation budget (Wu et al. 2001). For instance, if we assume society has a goal of conserving all the 52.3 hectares of wetlands (high ecosystem valued wetlands (A and B); at the margin, it would cost society about $6,466/ha at the margin, with a total wetland conservation budget of $338,172 over a 50 year period (Figure 3). Given this same wetland conservation goal of 52.3 ha, society would spend $52,300 (at $500/ha at the margin) for both the least cost targeting and low benefit targeting scenarios, respectively. If society was to change its conservation target of high ecosystem value wetlands by just 50%, from 52.3 ha to 26.1 ha, the total conservation cost would decrease to $78,300 (at $3,000/ha), leaving about $259,872 to conserve other wetlands. About $230,000 could conserve about 100 ha of least cost wetlands (at $1,200/ha at the margin) and about 100 ha of C and D wetlands (at $1,100/ha at the margin). Therefore, using policy targeting (in this case a blend), society could conserve more wetlands, including a significant area of highly beneficial wetlands. What we can not demonstrate is if the increase in conserved wetland area results in an overall decrease or increase in the provision of wetland ecosystem services when compared to the targeting of high ecosystem value wetlands. This is because from a social welfare perspective, it will not always follow that greater area of conserved wetland area is always preferred to smaller areas of quality wetlands.

In the long-run, significant reductions in the wetland conservation target of 52.3 ha (high ecosystem valued wetlands) is expected to free more resources to conserve significantly more wetlands in the other scenarios without reducing the potential quality of ecosystem services. Society might benefit from these additional conserved wetlands, especially if these conserved wetlands, including those with lower ecosystem values, improve the quality of their ecosystem functions over time.

5.0. Conclusion

We developed a spatially explicit wetland conservation cost model to estimate the economic value of private returns from wetland drainage scenarios, using an agriculture dominated sub-basin watersheds in the Vermilion River Basin, Alberta, as a case study. The study shows that wetland conservation costs are highly heterogeneous within the study area. Also, the study showed a positive correlation between the least cost wetlands and wetlands that offer low environmental benefits, suggesting that the choice of a wetland policy targeting strategies is important in achieving a wetland conservation goal in the study area under a given conservation budget.

The results of our study contribute to the literature on wetland conservation by showing that not all wetlands impose the same cost of conservation on an agricultural producer in the study area. Therefore, using a targeted wetland conservation policy could help society conserve more wetlands than using a non-targeted policy (which assumes that all wetlands are the same in terms of economic and environmental values). Other studies on wetland conservation costs produce localized estimates of conservation costs unlike the landscape-scale estimates from our study which were mostly used in non-targeted wetland conservation policies.

Our study is limited only to the private benefit of wetland drainage. While we did include a rough proxy for ecosystem service provision from wetlands using Alberta's Relative Wetland Value classification scheme, an important complement to our study would be spatially explicit monetary estimates of the social benefit of wetland conservation. Knowledge of the social monetary benefits of wetland conservation would enable an informed wetland conservation policy recommendation, from a spatially explicit perspective. Future studies are encouraged to conduct a spatially explicit model on the social benefit of wetland conservation, which has not been done in the wetland conservation literature. Again, future studies are encouraged to model off-site drainage costs, which may be an important component of wetland drainage costs to society.

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Table 1. Triangular Distributions Assumptions

Simulation Parameter	Triangular Distribution Parameters		
	Minimum	**Maximum**	**Average**
Cost of Production - Canola ($/Ha)	564.20	920.00	685.20
Cost of Production - Wheat ($/Ha)	650.52	933.33	803.70
Price of Canola ($/Tonne)	223.74	673.45	420.95
Price of Wheat ($/Tonne)	192.86	295.11	242.44
Probability of Harvesting	0.5	1	0.8

Table 2. Regional Benchmarks on Annual Crop Production in Alberta, Canada

	Low		High		Average	
	Canola	**Spring Wheat**	**Canola**	**Spring Wheat**	**Canola**	**Spring Wheat**
Yield (Tonne/Ha)	2.14	2.71	3.28	4.19	2.74	3.59

Adapted from the Government of Alberta Agriculture and Rural Development (2012a and 2012b).

Table 3. Summary Statistics of Wetland Conservation Targeting Scenarios and the Status Quo (Nontargeting)

Variable	Nontargeting	Least Cost Targeting	High Benefit Targeting	Low Benefit Targeting
Mean NPV/Ha	$2,404	$189	$2,322	$207
95% C. I	[$2,313-$2,496]	[$170-$208]	[$2,090-$2,555]	[$186-$229]
S.D NPV	$2,796	$408	$2,270	$435
Maximum NPV	$19,617	$1,526	$6,466	$1,623
Total Wetland (ha)	465.6	148.4	52.3	140.5
Total Ha of A & B	52.3	14.5	52.3	0
Total Ha of C $ D	413.3	134	0	140.5
Number of Wetlands	3,584	1,781	369	1,621

NPV denotes the net present value of drained wetland; C.I denotes confidence interval; S.D denotes standard deviation; Num denotes number; A represents wetlands with the highest ecosystem values and D represents wetlabds with the lowest ecosystem values.