The Economics of Wetland Drainage and Retention in Saskatchewan

Brett G. Cortus,[1] Scott R. Jeffrey,[2] James R. Unterschultz[3] and Peter C. Boxall[4]

[1]*Former Graduate Student.*
[2]*Professor, Department of Rural Economy, 531 General Services Building, University of Alberta, Edmonton, Alberta T6G 2H1, Canada (corresponding author: phone: 780-492-5470; fax: 780-492-0268; e-mail: scott.jeffrey@ales.ualberta.ca).*
[3]*Associate Professor, Department of Rural Economy, 531 General Services Building, University of Alberta, Edmonton, Alberta T6G 2H1, Canada (phone: 780-492-5439; fax: 780-492-0268; e-mail: jim.unterschultz@ales.ualberta.ca).*
[4]*Professor, Department of Rural Economy, 531 General Services Building, University of Alberta, Edmonton, Alberta T6G 2H1, Canada (phone: 780-492-5694; fax: 780-492-0268; e-mail: peter.boxall@ales.ualberta.ca).*

The public and private net benefits of retaining wetlands in agricultural cropland in east central Saskatchewan are evaluated in a policy case study. Wetland drainage on agricultural lands continues to occur despite evidence from existing studies concerning societal benefits derived from wetland retention. A simulation model was developed to estimate on-farm costs and benefits associated with wetland drainage in east central Saskatchewan. The private net benefits were compared to existing estimates of the public benefits to retaining wetlands in this region. The analysis suggests that payments from existing "beneficiary pay" policies, such as public incentive payments to farmers to retain wetlands, are too low to retain wetlands at risk of drainage.

Les avantages nets, publics et privés, de la conservation des milieux humides sur les terres agricoles cultivables du centre-est de la Saskatchewan sont évalués dans une étude de cas sur les politiques. Le drainage des terres humides en milieu agricole se poursuit malgré l'existence d'études montrant que la conservation des milieux humides procure des avantages à la collectivité. Nous avons élaboré un modèle de simulation afin d'estimer les coûts que doivent engager les agriculteurs pour drainer les terres humides dans le centre-est de la Saskatchewan et les avantages qu'ils en retirent. Nous avons comparé les avantages nets privés avec les estimations existantes des avantages publics liés à la conservation des terres humides dans cette région. Les résultats de notre étude autorisent à penser que les primes prévues dans les politiques existantes du «bénéficiaire payeur», telles que les primes d'encouragement versées aux agriculteurs pour conserver les milieux humides, sont insuffisantes pour protéger les milieux humides contre le risque de drainage.

INTRODUCTION

Wetland areas serve a number of important functions that result in ecological goods and services. These ecological functions include provision of plant and wildlife habitat, opportunities for recreation activities, flood control, erosion prevention, groundwater recharge and storage, carbon sequestration, and improved water quality (Adams 1988; Costanza et al 1997; Brander et al 2006). However, the benefits arising from the resulting goods and services accrue to society rather than to individuals owning the land on which

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the wetlands are located. Conversely, there are often economic incentives for agricultural producers to drain wetlands as this allows them to realize additional private economic returns. The result is divergence between social and private interests.

Evidence suggests that wetland drainage has been a significant issue in the Canadian prairie region. Various reports (e.g., Environment Canada 1986; Watmough and Schmoll 2007) state that 40% or more of prairie wetlands have been lost in the past century due to drainage activities. As well, wetlands on prairie agricultural land continue to be drained. For example, average losses between 1985 and 1999 in Saskatchewan are estimated at 4.65% (Saskatchewan Watershed Authority 2007, p. 90). Watmough and Schmoll (2007) provide similar estimates for overall wetland loss in the prairies from 1985 to 2001, and conclude there is little evidence that the rate of wetland loss in the prairies has slowed in the past 50 years.

Public policy may be used to mitigate wetland losses. For example, wetland conversion to agriculturally productive land might be avoided if farmers were compensated for the foregone incremental benefits of drainage (van Kooten and Schmitz 1992). However, informed public policy should be based on estimates of both private and public benefits associated with alternative land uses (Pannell 2008).[1] In the case of wetland policy, net benefits accruing to agricultural producers from draining wetlands should be weighed against the value of wetlands to society. While there is an extensive literature on wetland valuation (e.g., Brander et al 2006 provide a review of prior studies), little research has been done in a Canadian setting, particularly in terms of comparing private and public benefits. Van Vuuren and Roy (1993) presented an Ontario case study estimating private benefits of drainage and public benefits of wetland retention but their discussion did not provide a framework for developing a public policy response to this information. With continuing wetland loss to agriculture, there is a need to evaluate the economic trade-off between the private benefits and public costs of draining wetlands.

The purpose of this paper is to examine and quantify the economic trade-offs associated with wetland drainage in the Canadian prairie region. In particular, the objectives of this study are to: (1) model wetland drainage decisions for a representative grain producer and (2) use the results of this analysis to discuss policy options that may be considered in an attempt to incorporate societal values associated with wetlands. A case study approach is used here. Results from a stochastic simulation model designed to estimate the private benefits and costs of wetland drainage for an east central Saskatchewan grain farm are compared to available estimates of the public benefits of wetland retention. These comparisons provide the framework for a discussion on public policy responses.

THE ECONOMICS OF WETLAND RETENTION VERSUS DRAINAGE

Wetlands have a number of ecosystem functions that result in multiple ecosystem goods and services. Costanza et al (1997) examine ecosystem services and functions provided by different biomes, and group these services into 17 categories. Wetlands generate goods and services for 10 of these 17 categories; retention of greenhouse gases (GHGs), flood control, water flow regulation, provision of water for agricultural production, filtering of nutrients, waterfowl habitat, provision of game for food, forage for livestock and ungulates, esthetic qualities, and recreation (e.g., hunting).

While identification of the services provided by wetlands is relatively straightforward, valuation of these services is not. There is a growing literature that examines the value of ecosystem services provided by wetlands. Given the diversity associated with wetlands in terms of the types and levels of services provided, there will also be significant heterogeneity in resulting societal values. An obvious hurdle in quantifying wetland values is the lack (in most cases) of markets. Brander et al (2006) provide an overview of alternative methods used in valuing wetlands. These include contingent valuation, hedonic pricing, travel cost, replacement cost, and opportunity cost.

The wetland values resulting from these analyses vary greatly, depending on location, type of wetland, type of service, and methodology. However, there is also significant variability among estimates within each category. For example, the mean annual value (in 1995 US$) from valuation studies for wetland marshes reported by Brander et al (2006) was approximately $3,000–$4,000/hectare (ha). The median value, however, was just over $100/ha.

It is obvious, then, that the economic value of benefits accruing to society from wetlands is uncertain. Adding to the challenge of assigning value is the issue of transferability. There is limited information available for the value of wetlands in Western Canada. The ability to transfer values estimated for wetlands in other locations is limited due to factors, such as uniqueness of wetland characteristics and the uncertainty associated with the original estimates (Brander et al 2006).

While society benefits from retention of wetlands, there are private benefits associated with draining of wetlands that accrue to the landowner. The most obvious of these is the gain in agriculturally productive land; specifically, revenues can be generated on land that could not previously support crop production. The magnitude of this benefit depends on resulting crop yields. There is no comprehensive literature comparing yields on drained areas to upland yields. Anecdotal evidence provided by Lyseng (2002) and Wanchuk (1986) suggests yields could be greater on drained areas for reasons, such as better soil and moisture availability during dry periods. Whether or not any yield differences would continue indefinitely is also unknown and not addressed in the literature.

Draining wetlands also provides benefits through reduced nuisance costs. Wetlands are obstacles that increase the distance a farm operator must travel in a field, resulting in inefficient field operations. These inefficiencies are referred to as nuisance costs (Danielson and Leitch 1986). The magnitude of these costs depends on the size, shape, and number of wetland areas present in a field. Gelso et al (2008) conclude from a survey of U.S. farmers that wetlands areas impose substantial nuisance costs on farmers and these costs increase with the number of wetlands in a field.

Drainage reduces the number, size, and duration of wetlands and low spots in fields. Reductions in nuisance costs are the result of fewer turns with machinery, lower risk of equipment becoming mired in fields, reduced wear and tear on machinery, less waste of crop inputs, and a reduced need to return to fields and seed low spots (Eidman 1997). Fewer turns result in reduced time spent in the field by farm operators and cost savings in machinery operating expenses used to travel the extra distance around wetlands and low spots. Savings in crop inputs are also realized because fewer areas are overlapped.

Limited research has been done to quantify nuisance costs (e.g., Leitch 1983). Typically, minimal consideration is given to the influence of nuisance costs when investigating the economics of drainage activities. The farm level economic drainage model used in this study captures the key private benefits and costs of wetland drainage, including nuisance costs.

REPRESENTATIVE CROP FARM

The representative grain and oilseed farm modeled in this study is located in the rural municipality (RM) of Emerald. This RM is in the aspen parkland region of east central Saskatchewan, and lies within the prairie pothole region, an important region for migratory waterfowl habitat. Drainage in this RM is straightforward with farm operators building surface ditches and contouring their fields using (typically) their own equipment. Of the nearly 1,300 quarter sections of land in the RM, most have already experienced some drainage but, unlike other districts in Saskatchewan, much of the wetland area still remains intact.[2] As a result, this area is considered "at risk" for significant further wetland drainage.

Analysis of 2001 Agricultural Census data for this region of Saskatchewan suggests that farm size in the Emerald RM varies, with the majority of farm revenue generated from annual cropping activities. Therefore, alternative representative farm scenarios are modeled, allowing the farm size to vary from 4 (259 ha) to 20 quarter sections (1,295 ha).

A crop rotation for the base farm was developed using expert opinion to reflect local agronomic practices and crop rotations. The result was a five-year rotation of canola, barley, flax, spring wheat, and summerfallow. Agricultural census data suggested that the practice of summerfallow, although declining, still represented a significant portion (17%) of cropland area in the Emerald district. Therefore, this practice was included in the rotation. Zero till practices are used by the farm.

Operating costs for each crop were estimated based on enterprise budgets from Saskatchewan Agriculture, Food, and Rural Revitalization (SAFRR). These included input costs (e.g., fertilizer, pesticides, and seeds) and machinery operating expenses (i.e., fuel and maintenance). Machinery operating costs were based on the farm's machinery complement. SAFRR's (2004) *Farm Machinery Custom and Rental Rate Guide* was used to determine the machinery complement for the base farm. The machinery complement influenced the time required to perform field operations on each quarter section. This, in turn, influenced the amount of time available to undertake drainage activities. The size and makeup of the machinery complement was varied according to farm size.

CASH FLOW SIMULATION MODEL

A dynamic stochastic cash flow simulation model was developed in a whole farm setting to estimate the net private benefits for wetland drainage. The model incorporated economic and technical relationships so that an annual net cash flow was calculated for each year of a 20-year time horizon. Participation in public agricultural safety net programs was also included in the model. The resulting net cash flows were used to calculate a net present value (NPV) for alternative drainage scenarios.

The simulation model incorporated dynamic and stochastic commodity prices, weather variables, crop yields, land base, and constraints on time available to undertake drainage. The model was built to permit sensitivity analysis to be done with respect to key components, including farm size, nuisance costs, and expected crop yields on drained wetlands. What follows, is a summary discussion of important policy relevant features and the stochastic components of the model. Detailed documentation of the model as well as results of validation and verification tests is available from the authors upon request.[3]

Table 1. Crop yield model parameter estimates

	Coefficients			
Variable	Canola	Barley	Flax	Wheat
GS/GDD	8.740**	22.830***	8.543***	18.063***
	(3.762)	(5.431)	(2.798)	(3.494)
(GS/GDD)	−21.297**	−47.444***	−17.767**	−42.969***
	(−2.092)	(14.699)	(7.574)	(9.457)
Constant	0.357	−0.101	0.156	0.133
	(1.100)	(0.469)	(0.242)	(0.301)
Std. Error	0.255	0.369	0.190	0.237
R^2	0.157	0.549	0.390	0.504

Notes: Standard errors are in parentheses. GS = growing season precipitation; GDD = growing degree days.
** and *** represent significance at the 5% and 1% level, respectively.

Crop Yields

Crop yield models specific to the RM of Emerald were developed, based on the assumption that yields were a function of weather. Weather plays a significant role in the growth of plants and weather variables were therefore incorporated as explanatory variables. A water use to water demand ratio was developed using growing season precipitation (GS) as a proxy for water use and growing degree days (GDD) as a proxy for water demand. Crop yield equations were specified as follows:

$$y_t^i = \phi_0^i + \phi_1^i \frac{GS_t}{GDD_t} + \phi_2^i \left(\frac{GS_t}{GDD_t} \right)^2 + \varepsilon_t^i \tag{1}$$

where y_t^i is the yield for the ith crop in year t, ϕs are parameters to be estimated and ε is the error term. Yield equations were estimated for canola, barley, flax, and wheat.

GS and GDD variables were calculated using nearby weather station data. The growing season used was May 15 to August 13 (Saskatchewan Institute of Pedology 1994). Total precipitation over the growing season was used to represent GS. Daily GDD were calculated based on Corbally and Dang (2002) and then summed over the growing season to obtain annual GDD for the year. Historical crop yields for the RM of Emerald were used to estimate the coefficients. Table 1 provides the parameter estimates for the crop yield models.

To simulate weather, values were drawn from empirical GS and GDD distributions. Logistic distributions provided the best fit for both variables; a negative correlation of −0.30 between the two variables was included in the model. Upper and lower bounds for GS were established using existing precipitation distributions to avoid extreme values while still maintaining the mean GS from the data; bounds were beyond any observed values in the existing data series. No bounds were necessary for the GDD variable.

Draws from the GS and GDD distributions were used as "inputs" to calculate the water demand to water use ratio in the simulation analysis. Standard errors in the yield equations were adjusted upward so that the maximum yield would be similar to those provided by the Saskatchewan Crop Insurance Corporation for the years 1997–2002 (Hulston 2004) and to adjust for the greater variance in yields found at a farm level versus the RM level. The adjustment factors on standard errors were comparable to Marra and Schurle (1994) results for Kansas wheat yields at the county versus farm level. Cross-equation standard error correlations were incorporated into the model using the following structure (from Hull 1997), consistent with a Choleski decomposition approach:

$$\begin{aligned} &\varepsilon_m = \sum_{k=1}^{k=m} \alpha_{mik} x_k \\ &\textit{subject to}: \\ &\sum_k \alpha_{mk}^2 = 1 \\ &\sum_k \alpha_{mk}\alpha_{jk} = \rho_{m,j} \end{aligned} \tag{2}$$

where ε_m was the corrected error for crop m, x_k was the initial standard normal error draw scaled according to the respective standard deviation of the crop, and $\rho_{m,j}$ was the correlation between the errors for crops m and j. Hence, yields were affected by GS and GDD draws from the weather distributions, but individual crop yields were also influenced by unique risk modeled from the yield equation (i.e., independent of the weather variables).

Crop Prices

Crop prices were also assumed to be stochastic in the simulation analysis. Saskatchewan crop price data (1960–2002) (SAFRR 2004) were deflated using the latest annual consumer price index and then converted to natural logarithms. The price data were tested for stationarity, with test results being mixed. A nonstationary price model ($P_t^m = \alpha_0 + P_{t-1}^m + \varepsilon_t^m$) was initially estimated. However, nonstationary processes often result in wide confidence intervals for forecasts over time. In this case, simulated prices for crops reached levels as high as \$20,000/ton and as low as \$1.00/ton. The extremely wide range in prices that resulted over the 20-year simulation time horizon precluded its use in the analysis and instead a stationary commodity price model was estimated and used in the analysis.[4]

A lagged annual time-series crop price model was estimated using these adjusted annual price data. The lag length for each crop was determined using the Akaike Information Criterion statistic. This resulted in three lags for wheat, canola, and flax prices and one lag for barley price. The stationary price model was estimated as follows:

$$\begin{aligned} P_t^C &= \gamma_0 + \gamma_1 P_{t-1}^C + \gamma_2 P_{t-2}^C + \gamma_3 P_{t-3}^C + \varepsilon_t^C \\ P_t^B &= \gamma_0 + \gamma_1 P_{t-1}^B + \varepsilon_t^B \\ P_t^F &= \gamma_0 + \gamma_1 P_{t-1}^F + \gamma_2 P_{t-2}^F + \gamma_3 P_{t-3}^F + \varepsilon_t^F \\ P_t^W &= \gamma_0 + \gamma_1 P_{t-1}^W + \gamma_2 P_{t-2}^W + \gamma_3 P_{t-3}^W + \varepsilon_t^W \end{aligned} \tag{3}$$

Table 2. Crop price model estimates

	Coefficients			
Variable	Canola	Barley	Flax	Wheat
P_{t-1}	0.878***	0.544***	0.813***	0.674***
	(0.113)	(0.079)	(0.092)	(0.086)
P_{t-2}	−0.464***		−0.445***	−0.270***
	(0.158)		(0.115)	(0.093)
P_{t-3}	0.274**		0.257***	0.217***
	(0.114)		(0.083)	(0.067)
Constant	1.934***	2.395***	2.305***	2.097***
	(0.503)	(0.422)	(0.466)	(0.420)
Std. Error	0.195	0.253	0.268	0.241
R^2	0.737	0.604	0.715	0.684

Notes: Standard errors are in parentheses. P_{t-k} is the logged annual price, lagged k years. The dependent variable is logged price.
** and *** represent significance at the 5% and 1% level, respectively.

where P_t was the current price, P_{t-n} was the price lagged n times, and γ_is were estimated coefficients. Table 2 provides the resulting coefficient estimates for the crop price model.

Modeling Wetland Drainage

Drainage activities are modeled to be consistent with typical practices in Emerald RM. Contour (i.e., surface) drainage is assumed to be feasible for all wet areas on every quarter section of the representative farm. Drainage activities are carried out in the fall after harvest has been completed. Any drainage initiated for a particular quarter section is assumed to be continued until completed; that is, no wet areas would remain upon full completion of a drainage project. Drainage costs incorporated in the model include costs for ditch construction, length of surface ditch, rehabilitation, and maintenance. For example, rehabilitation costs of \$217/ha and maintenance costs of 1.5% of ditch construction cost are inflation adjusted from Alberta drainage studies (Wanchuk 1986; Myhre 1992). Base case nuisance costs are calculated using Desjardins (1983) and are shown in Table 3 as percentage increases in selected variable costs.[5]

Variability in wetland characteristics is incorporated by changing the farmland base for each iteration of the model. At the beginning of each iteration, wetland characteristics for the representative farm are determined by drawing from a distribution of 100 quarter sections. Wetland characteristics (i.e., spatial data) for 600 quarter sections of land from Emerald were provided by Ducks Unlimited Canada. From these, 100 quarter sections were selected for use in the model. Selection was done using a stratified sampling procedure that ensured a representative cross-section of numbers and hectares of wet areas, both wetlands and cropped basins, per quarter section of land.[6] Varying the initial wetland characteristics for each iteration in this fashion removes bias in the initial choice of land base for a farm and captures the variation that occurs across different farms in the RM.

Table 3. Percentage increase in crop input costs due to wetland nuisance factors by farm size and number of wetlands per quarter section[a]

	Farm size (number of quarter sections)				
Number of wetlands	4	8	12	16	20
1–3	8.0%	9.0%	11.0%	14.0%	18.0%
4–6	9.0%	10.0%	12.0%	15.0%	19.0%
7–9	11.0%	12.0%	14.0%	17.0%	21.0%
>9	14.0%	15.0%	17.0%	20.0%	24.0%

Note: [a]Values represent percentage increase in machinery costs versus a quarter section with no wetlands, and are based on Desjardins (1983).

All 100 quarter sections selected have at least one wetland. However, almost two-thirds have only one wetland and only seven have more than three wetlands. For the 100 quarter sections included in the model, the mean (minimum, maximum) counts of wetland and cropped basins per quarter section are 1.84 (1, 13) and 14.17 (1, 52), respectively. The mean (minimum, maximum) values for wetland and cropped basin areas are 7.97 ha (0.25, 24.96) and 1.71 ha (0.01, 11.37), respectively.

Decisions to drain are made endogenously within the model. Expected net benefits for each drainage project are calculated each year. The project with the greatest net benefit is initiated in a particular year if: (1) the expected net benefit is positive and (2) there is sufficient time available in the fall season to undertake the project. Once a particular project is initiated, it continues in subsequent years until completed. Depending on the size and characteristics of the wetland area to be drained and the time available for drainage activities in subsequent years, projects are completed in three or four years. If a project is not initiated in a particular year due to negative farm cash flows, a negative benefit estimate, or lack of time to conduct drainage activity in the fall of that year, expected net drainage benefits are recalculated the following year and the drainage decision is evaluated again.

The time available to conduct drainage was stochastic and was based on the date of crop maturity and the number of workdays available for drainage activities. Specifically, it was calculated as a residual after other field activities were completed, the first of which was harvesting. Six possible crop maturity dates were used within the simulation: August 7, 14, 21, or 28, September 4 or 11. The crop maturity date was determined by accumulated GDD (discussed earlier). Lower (higher) GDD values for the growing season resulted in later (earlier) crop maturity dates.

Harvest activities were then scheduled according to the workdays required for completion. After harvest, the remaining time before freeze-up was allocated to other activities, such as maintaining existing drainage ditches, fall herbicide spraying, and summerfallow operations. After subtracting times required for these activities, any remaining workdays could be allocated to construction of new drainage ditches. Workdays in the four-month period from August to November (i.e., from crop maturity to freeze-up) were modeled using workday probability schedules from Rutledge and McHardy (1968).

Table 4. Summary of wetland drainage simulation results by farm size (2008 $)

	Farm size (number of quarter sections)				
Variable[a]	4	8	12	16	20
Area drained (ha)					
Wetlands	11.85	23.91	39.43	51.84	62.10
Cropped basins	2.28	4.60	7.48	9.65	11.83
Total drained as % of owned wet area	46.75%	47.70%	53.13%	52.47%	50.66%
Net private benefits ($/ha drained)					
PV of benefits[b]	$278	$340	$357	$383	$407
Standard deviation	$485	$404	$332	$280	$246
90% CI[c]					
Upper bound	$1,148	$1,072	$910	$871	$836
Lower bound	−$408	−$235	−$162	−$49	$34
Average annual benefit[d]	$27.76	$34.04	$35.67	$38.27	$40.66
% of iterations with positive benefits	58.20%	76.80%	86.00%	92.20%	96.30%

Notes: [a]Values presented here are means calculated over all stochastic iterations of the simulation.
[b]The present value (PV) of benefits is the difference between the perpetuity NPVs for the drainage and no drainage scenarios.
[c]CI is the 90% confidence interval.
[d]The annual benefit is calculated by converting the PV of benefits to an annuity.

Simulation Analysis

The model simulates the performance of the representative farm over a 20-year time horizon using @Risk software (Palisade Corporation, Ithaca, NY). One thousand iterations provide a consistent distribution of outcomes and estimates for the benefits and costs of cumulative wetland drainage over a 20-year time period. A real risk-adjusted discount rate of 10% is used.[7] Within the model, a perpetuity NPV resulting from the scenario allowing drainage to occur is compared to the corresponding NPV for the base scenario in which no drainage takes place.[8] This comparison is used to calculate the benefits (if any) associated with wetland drainage for the representative farm. Two farm programs, the Canadian Agricultural Income Stabilization program and crop insurance, are included in the analysis as participation in these programs influence revenues and costs and so could influence drainage decisions (Cortus et al 2009).

Calibration and validation of the stochastic model components were performed to confirm that each element performed as expected and were consistent with other available data. For example, crop yield and price models were checked to determine whether they generated infeasible values and if means of the resulting distributions of parameters approximated those values from other data sources. Each component was calibrated and validated separately and then changes were incorporated into the complete simulation model structure.

MODEL FARM DRAINAGE RESULTS

Table 4 reports basic simulation results associated with wetland drainage for the different representative farm scenarios. The average area drained ranges from 46% to 51% of owned wetlands for the farm. The annual net private benefits increase with farm size and range from \$27.76/ha of wetland drained for the 4 quarter farm to \$40.66/ha for the 20 quarter farm.

As indicated in Table 4, there is significant variability associated with the net private benefits of wetland drainage. For example, the total net benefit (in present value terms, before annualization) associated with wetland drainage for the 4-quarter section farm is \$278 while the standard deviation associated with this value is \$485. As a consequence, the 90% confidence interval for all but one of the farm sizes includes negative net private benefits.[9]

Also as reported in Table 4, variability in net private benefits decreases with increased farm size. This result, combined with the direct relationship between farm size and average net benefits of drainage, suggests that larger farms may have greater incentives to drain wetlands.

The variability in the simulation results is due in large part to the stochastic nature of crop yields and prices in the model. However, another contributing factor is the fact that the initial set of wetland characteristics for the representative farms is varied from iteration to iteration, as discussed earlier. Farm wetland characteristics will obviously impact the economic results. Private returns to drainage could be significantly positive/negative if a particular farm comprised quarter sections where drainage costs are sufficiently low/high and/or the benefits from drainage are high/low.

One other potential source of uncertainty with respect to model results is the level and pattern of nuisance costs. As discussed earlier, limited information is available from published literature with respect to the significance of these costs, and the relationship of nuisance costs to the number and makeup of wetlands, as well as the impact of farm size. The default scenario in this study was based on Desjardins (1983); specifically, the "nuisance factor" in terms of percentage increase in costs is assumed to increase with greater numbers of wetlands per field.

In order to test the sensitivity of the results to this assumption, two alternative nuisance cost scenarios were modeled for two farm sizes (i.e., the 8- and 16-quarter section farms). The alternative scenarios were: (1) a constant percentage increase in cropping costs resulting from the presence of wetlands, regardless of the number of wetlands and (2) zero nuisance costs. The impact on drainage activities and net benefits from drainage varied depending on the scenario. The results of this analysis are provided in Table 5.

The difference between an increasing rate of increased cost versus a constant rate of increase was small: \$34 versus \$35/ha drained for the 8-quarter section farm and \$38 versus \$37/ha drained for the 16-quarter section farm. Not surprisingly, the net benefits from drainage were significantly reduced with an assumption of zero nuisance costs. Under this scenario, benefits per hectare drained decrease from \$34 to \$24 for the 8-quarter section farm and from \$38 to \$27 for the 16-quarter section farm.[10] However, even without nuisance cost considerations, the average annual net benefits associated with drainage are highly positive.

POLICY DISCUSSION

The farm level simulation results presented in the previous section suggest that there are positive economic incentives for producers in this area to drain wetlands. This is consistent

Table 5. Summary of wetland drainage simulation results for alternative nuisance cost scenarios by farm size (2008 $)

	8-quarter section farm Nuisance cost scenario[a]			16-quarter section farm Nuisance cost scenario[a]		
Variable	Increasing	Constant	None	Increasing	Constant	None
Area drained (ha)						
Wetlands	23.91	24.11	21.11	51.84	51.00	44.73
Cropped basins	4.60	4.65	3.82	9.65	9.43	7.78
Total drained as % of owned wet area	47.70%	48.15%	41.43%	52.47%	51.56%	44.65%
Initial nuisance costs as % of total variable costs[b]	2.51%	2.65%	0.00%	3.35%	3.02%	0.00%
Initial total farm nuisance cost	$2,305	$2,434	$0	$5,665	$5,068	$0
Annual net private benefits ($/ha drained)[c]	$34	$35	$24	$38	$37	$27

Notes: [a]Increasing represents a scenario whereby nuisance costs (%) increase at an increasing rate as the number of wetlands within a quarter section increase. Constant represents a scenario whereby the rate of nuisance costs increases at a constant rate as the number of wetlands within a quarter section increase. None represents a scenario where nuisance costs are assumed to be zero.
[b]Initial nuisance costs are costs associated with maneuvering around wetlands before any drainage is conducted.
[c]These are the present values of the net benefits associated with drainage, converted to an annuity value.

with empirical evidence of ongoing drainage activities through much of the pothole region in Saskatchewan. Therefore, if it is deemed socially optimal to retain wetlands because of the ecological goods and services provided, some form of policy intervention is required. At the time of this study the province of Saskatchewan's drainage policy allowed property owners to drain wetlands if the water did not affect surface flows to other properties (Saskatchewan Watershed Authority 2008). Permits were required if drainage activities would lead to water entering neighboring lands. Thus, policy did exist in the form of regulations, to restrict wetland drainage. However, direct observation and consultations with experts indicated that, in the study region, drainage onto other properties often occurred without permits. Effectively, then, the province was ceding wetland property rights to land owners, which included the right to make changes to these wetlands.

Also at the time of this study, Ducks Unlimited Canada (DUC) (2004) was involved in a wetland retention pilot project in the RM of Emerald. Farmers were offered incentives in the form of land tax credits to retain wetland hectares on owned land that were not already cropped. These payments compensated wetland owners for the tax burden associated with wetland ownership. In exchange for the tax credit, the farmers agreed not to drain the wetland areas enrolled in the program during the year in which the payment was made. Yearly payments ranged from \$1.40 to \$2.50/ha of wetland enrolled in the program (Ducks Unlimited Canada 2004). Participation figures for the tax credit program indicated approximately 27% and 39% of wetlands in the RM of Emerald were enrolled in the program in 2003 and 2004, respectively.

So what is the appropriate type of policy intervention to provide for effective and socially optimal levels of wetland retention? Pannell (2008) provides a policy framework for encouraging environmentally beneficial land use change. The premise of this framework is that policy should be based on the sign (positive versus negative) and relative magnitude of public versus private net benefits associated with specific land use changes.[11] For example, if a particular land use change generates both public and private benefits that are positive, only extension or education activities should be required in terms of policy. Conversely, if public and private benefits are "in conflict" (i.e., one positive and one negative), then Pannell's framework suggests that the appropriate policy response may be positive or negative economic incentives depending on relative magnitudes of benefits as well as other factors, such as property rights.

The farm level simulation results provide estimates of the private net benefits associated with wetland drainage, but what about public benefits of wetland retention? As noted earlier, the ecological goods and services provided by wetlands provide benefits to society. However, the value of these benefits is uncertain.

A number of published studies have estimated the value of wetlands. Given the variability in types of wetland, type of wetland services considered, valuation methodology used in these studies, it is not surprising that there is a large range in resulting wetland values. In their review of wetland valuation research, Brander et al (2006) classify studies according to a variety of criteria, including type of wetland. The average and median annual per hectare wetland values from studies of freshwater marsh wetlands (i.e., the closest wetland classification in their paper to what is found in the prairie pothole region) are \$3,000–\$4,000 and approximately \$100, respectively (1995 US\$).[12]

Unfortunately, few wetland valuation studies have been done in Canada, particularly in the region of interest in this study. Belcher et al (2001) estimate annual public benefits of retaining wetlands in agricultural lands for the Upper Assiniboine River Basin (UARB) in the aspen parkland ecoregion in eastern Saskatchewan, using various sources and methodologies.[13] The UARB is adjacent to the region surrounding the RM of Emerald, the area modeled by the farm case study. Table 6 provides their high, best, and low estimates of annual benefits, inflation adjusted to 2008 \$, excluding any net private farm benefits to wetland drainage. Their best estimate of \$81.55/ha of retained wetland is significantly greater than the range of average annual net private benefits associated with wetland drainage obtained from the simulation analysis, discussed earlier. The low estimate of \$39.66 is roughly equivalent in magnitude to the average benefits from wetland drainage.

As shown in Table 6, annual public benefits from wetland retention include a variety of ecological goods and services. However, reduction in GHG emissions and carbon sequestration together represent the greatest contribution to public benefits. At \$34.55/ha (i.e., \$11.18 + \$23.37), they represent over 42% of the total best estimate value. While there is uncertainty associated with all values in Table 6 (i.e., hence the high and low estimates),

Table 6. Estimates of annual net public benefits for wetland retention in the Saskatchewan UARB, excluding private farm benefits of drainage (2008 $)[a]

Public benefits (costs) $/ha/year	High	Best estimate	Low
Saved government payments	22.95	15.30	7.66
Saved crop insurance premiums	6.28	4.19	2.10
Improved water quality (reduced sediment)	11.14	5.51	1.60
Water-based recreation	1.63	1.09	0.55
Decreased wind erosion	4.78	3.18	1.60
Reduction in GHG emissions	16.78	11.18	5.59
Carbon sequestration	35.06	23.37	11.69
Increased wildlife hunting	22.79	12.77	6.39
Increased wildlife viewing	7.69	4.96	2.48
Gross public benefits excluding private farm	$129.10	$81.55	$39.66
Program administration costs paid to farmers to retain wetland	−1.24	−2.48	−3.72
Wildlife depredation compensation to farmers with retention of wetlands	−0.38	−0.76	−1.14
Net public benefits excluding private farm	127.48	78.31	34.80

Note: [a]Adapted from Belcher et al (2001, Table 17).

one of the more significant sources of uncertainty in wetland value undoubtedly arises from the assumption regarding carbon prices. In their original analysis, Belcher et al (2001) used a carbon price of $10 per ton (2001 $CDN). The public benefits in Table 6 reflect this price, with values updated to 2008 $.

It may be the case that this overstates the value of carbon. For example, the price of carbon on the Chicago Climate Exchange (CCX) in July 2008 was approximately US$4.00 (Chicago Climate Exchange 2009).[14] Applying this price to the GHG emission and carbon sequestration estimates used by Belcher et al (2001), the best estimate of public value for wetlands would be revised to $59.35/ha. Alternatively, if no value for carbon is included in the calculations, the best estimate would be reduced to approximately $47/ha.

A recent study by Hansen (2009) examined the value of restoring wetlands in the prairie pothole region, where the value focused specifically on sale of carbon offsets. Using the value estimates developed by Hansen (2009) for GHG emission reductions and carbon sequestration, and adjusting for inflation and exchange rate, combined with the other values from Table 6, results in a revised best estimate of $74.60/ha for retention of wetlands.

From the discussion provided here, there is a range of values associated with both public and private benefits of wetland drainage/retention. This is not surprising, given the degree of uncertainty associated with the economics of both wetland drainage and retention discussed earlier. However, two patterns emerge from the values in this table. First, both private benefits of wetland drainage and public benefits of wetland retention are not inconsequential. Second, there is support for the view that public benefits associated with wetlands are greater in magnitude than the private benefits for drainage.

Given the pattern of private and public benefit estimates, Pannell (2008) would suggest that government intervention in the form of "incentives" is an appropriate policy response. Incentives may be positive (e.g., subsidies) or negative (e.g., taxes, regulations). The appropriate choice will be influenced by the interpretation of what constitutes "land use change," wetland drainage or retention/restoration, and with whom property rights to wetlands reside.

If it is assumed that land owners hold the property rights for wetlands and that wetland retention or restoration represents land use change (i.e., wetlands either have been drained or under expected economic conditions will be drained), then there are significant positive public benefits from land use change and equally significant negative private benefits. In this case, positive incentives are an appropriate policy response. This would include options, such as direct subsidies, or market-based instruments, such as auctions. In effect, society would be compensating producers for providing valuable ecological goods and services through wetland retention/restoration.

Alternatively, if property rights for wetlands are assumed to belong to society and wetland drainage represents the land use change, there are significant positive private benefits and negative public benefits associated with the change. In this case, negative incentives would be appropriate in terms of a policy response. These may include taxes, fines, and/or regulations, restricting producers from undertaking wetland drainage activities. In this scenario, producers would be penalized for restricting or preventing, through drainage activities, production of ecological goods and services at a level consistent with what is demanded by society.

Government and nongovernment organization (NGO) approaches to wetland retention on private lands have tended to involve voluntary retention, education, and incentives. Historically the focus was on volunteerism and the cost of retention fell entirely on farmers. Given the magnitude of the estimates for public and private benefits highlighted in this study, it should come as no surprise that these policy approaches have not deterred drainage activities on privately owned land.

More recently, governments and NGOs have followed up on their education and volunteer approach by adding cost-shared incentive programs for retaining wetlands (e.g., National Farm Stewardship Program; DUC programs). This represents a move toward positive incentives as a policy approach for retaining wetlands. The DUC tax credit payment program, however, was established without the benefit of reliable estimates of the wetland drainage benefits for landowners. The comparable expected costs to farmers of retaining wetlands in the farm simulation study are over eight times the amount offered under the tax credit program. The participation figures noted earlier for the tax credit program suggest that the long-run effectiveness of a stand-alone program, such as the DUC program, remains an open question. As suggested by van Kooten and Schmitz (1992), farmers may enroll wetlands that have very high costs to drain or wetlands for which drainage is possible but not planned in the near future due to drainage time constraints. The discussion here suggests that under a beneficiary-pay-positive-incentives policy approach, payments to farmers must be well above the level provided by DUC tax credit payments to ensure permanent retention of wetlands that are at risk of drainage.

The net private benefit estimates from the simulation model highlight another relevant policy issue. In particular, the wide confidence intervals around the net private benefits suggest a need for more nuanced policy approaches, as there is likely a significant degree of diversity in terms of the magnitude of the trade-off between public and private benefits. For example, in many cases the private benefits of drainage may be greater than the expected net public costs of losing the wetland. In other words, for a number of wetlands the appropriate public response may be to take no action if the landowner initiates wetland drainage. For other wetlands, the appropriate response may be negative incentives (e.g., fines) or, given the current policy environment, public incentive payments to retain wetlands. As well, there are a number of private outcomes where drainage of selected wetlands may be minimally beneficial to farmers. Incentive payments to maintain these wetlands should not be required in these cases.

CONCLUSIONS

A grain and oilseed farm level simulation model for east central Saskatchewan was developed and used to estimate the private benefits and costs of wetland drainage in agricultural lands. The net benefits have wide confidence intervals but the annual average net benefit increased from \$28 to \$41/ha drained as farm size increased. Alternatively, these benefits can be viewed as the net costs to land owners to retain wetlands at risk of drainage. A comparison of the net private costs to the public benefits of wetland retention in the same region (Belcher et al 2001) indicates the annual average private costs are lower than the public benefits of wetland retention. When this analysis is considered in terms of policy options, current government and NGO payments/cost sharing with farmers are too low to permanently retain wetlands at risk of drainage under the current "beneficiary pays" policy framework. In part, this case study on wetland drainage and retention explains why drainage continues to occur in the Western Canadian Prairie Pothole Region.

Optimal policy is contingent on the magnitude of private versus public benefits of wetland drainage/retention and is also influenced by effective property rights. Based on the discussion in this paper, a case can be made for the use of either positive or negative economic incentives to encourage wetland retention or restoration. Given historical precedent, positive incentives are more likely to be politically palatable. An alternative policy approach, such as polluter pay, might be an appropriate policy response in selected wetland retention situations based on the case study analysis. For example, provincial regulations governing wetland drainage could be enforced. However, polluter pay approaches may be difficult to implement given issues, such as historical precedence, current land rights, and previous government support for drainage. This research provides more reliable estimates of the private benefits of drainage and places the policy discussion in a case study framework that suggests alternative policy directions.

NOTES

[1]For the purposes of the discussion in this paper, private costs and benefits associated with a particular land use accrue to the landowner while public costs and benefits accrue to everyone else. This distinction is consistent with the terminology used by Pannell (2008).

[2]Personal communication with Ducks Unlimited Canada. For example, in contrast to the study region of Emerald, one district near St. Gregor SK (i.e., in the RM of St. Peter) has lost 90% of the wetlands due to drainage between 1974 and 2002 (Saskatchewan NAWMP Technical Committee 2008, p. 18).
[3]A complete explanation of the simulation model is provided by Cortus (2005).
[4]Augmented Dickey-Fuller (ADF) and Kwiatkowski, Phillips, Schmidt, Shin (KPSS) tests were used to test for stationarity. Nonstationarity was uniformly not rejected using the ADF test statistic. Conversely, there were mixed results from the KPSS test with stationarity not being rejected for some price series, depending on the significance level. The decision to reject the nonstationary price model is supported by Dixit and Pindyck (1994) who argue that (1) commodity prices should, over the long run, revert back toward the marginal cost of production and (2) it is difficult to distinguish between a nonstationary and stationary process with (as is the case here) relatively few years of data. Verbeek (2004) also noted that the ADF test has low power. ADF and KPSS test results for the price data used in this study are available from the authors upon request.
[5]Desjardins (1983) is used as the basis for determining the impact of wetlands on efficiency of machinery operations in the field. This is represented as a percentage increase in crop costs. This percentage is then applied to the per acre crop costs for the representative farm to obtain a dollar value estimate of nuisance costs.
[6]Cropped basins are cultivated wetland areas that hold water during years with above average precipitation. The quarter sections were stratified into four quartiles, based on wet area counts and hectares of wet areas, and 25 quarters were randomly selected from each of the four quartiles.
[7]Capital market line theory and variability of returns from the simulation analysis were used to determine the discount rate, which incorporates a risk-free rate of return and a risk premium (Copeland and Antikarov 2003; Ross et al 2003). The 10% value used in this study is similar in magnitude to values used in previous drainage studies (e.g., Leitch 1983; Danielson and Leitch 1986).
[8]The NPV for each scenario was extended into "perpetuity" in order to capture the value of benefits beyond the end of the 20-year simulation time horizon. Present value calculations beyond the end of the 20-year simulation period were done using expected cash flows for the farm.
[9]Drainage decisions are based on *ex ante* model forecasts of benefits for drainage (i.e., expected benefits). However, negative wetland economic drainage benefits can occur in the model because actual (i.e., *ex post*) prices, weather and yields may lead to a negative economic outcome.
[10]Sensitivity analysis was also done for crop yields on drained wetlands; specifically, average yields were varied by 5% and 10% from those on adjacent "upland" areas. While not reported here, only minor changes in net private benefits resulted from these changes.
[11]Interested readers are directed to Pannell (2008) for a full explanation and discussion of the policy framework.
[12]Brander et al (2006) note the divergence between the average and median values, which suggests a skewed distribution of wetland values, with a relatively few number of very large values.
[13]Belcher et al (2001) extensively document their methodology and sources of estimates. A summary of the UARB estimates is more easily accessed in the report by Olewiler (2004, Table 8).
[14]As noted by Hansen (2009), however, the CCX price probably represents a very conservative estimate of the social value of carbon.

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