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FARM LEVEL ECONOMICS OF ECOSYSTEM SERVICE PRODUCTION

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Growing Forward

Farm Level Economics of Ecosystem Service Production

Abstract
This paper provides a review of four University of Alberta research studies that examine the farm-level economics of adopting agricultural Beneficial Management Practices (BMPs). BMPs considered in the four studies include land use changes and/or management practices that involve retaining natural areas, protecting natural areas, crop management and pasture management. Examples of BMPs include buffer strips, shelter belts, addition of legumes to crop rotations and crop residue management. Having accurate estimates of costs and benefits associated with these types of BMPs is important for informing policy development. The empirical results indicate significant heterogeneity in terms of economic impacts of adopting different BMPs, particularly with respect to the scale of the impact. However, in general adoption comes at a net cost to producers in the study regions. This implies that policy intervention would be necessary to achieve significant levels of adoption by producers. The paper concludes with a discussion of possible future research directions.

Keywords: representative farm analysis; Monte Carlo simulation; BMP; ecosystem services; net private benefits

JEL Classification: C15, Q12, Q15, Q24, Q25, Q28, Q57

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Abbreviations/Acronyms

AAFC	Agriculture and Agri-Food Canada
AAFRD	Alberta Agriculture, Food and Rural Development
AB	Alberta
BMP	Beneficial Management Practice
BRM	Business Risk Management
CAIS	Canadian Agricultural Income Stabilization Program
ES	Ecosystem Services
LLB	Lower Little Bow
MNCF	Modified Net Cash Flow
NPV	Net Present Value
OSW	Off-Stream Watering
REES	Department of Resource Economics and Environmental Sociology
SK	Saskatchewan
SWAT	Soil Water Assessment Tool
WEBs	Watershed Evaluation of Beneficial Management Practices

Table of Contents

Farm Level Economics of Ecosystem Service Production

I. Introduction

Policymakers and society in general are concerned with quality of the environment, and ways in which environmental quality may be maintained or (often) enhanced. One means to account for and potentially quantify environmental quality is through the concept of ecosystem or ecological goods and services. Various definitions or descriptions of these goods and services have been proposed and used (e.g., Cook and Spray 2012; Costanza et al 1997; Gómez-Baggethun et al 2010; Norgaard 2010; Sagoff 2011). In this paper, the definition provided by Boyd and Banzhaf (2007, p. 619) is used; ecosystem services (ES) are "components of nature, directly enjoyed, consumed, or used to yield human well-being". As suggested by the definition, ES are "things" of value to humans (i.e., benefits are derived from them) that are produced as a result of properties or processes (biological, chemical and/or physical in nature) associated with ecosystems.

Just as there are alternative definitions of ES, there are also alternative means of classifying them (e.g., Costanza et al 1997; Fisher et al 2009). Boyd and Banzhaf (2007) provide an inventory of ES, grouped according to type(s) of benefits. For example, crop populations, soil quality, water availability are examples of ES that provide harvest benefits to humans. Biodiversity, varied natural land cover and wilderness provide aesthetic, spiritual and/or emotional benefits. In some cases, individual services may provide multiple types of benefits; water availability, for example, provides benefits related to harvest, drinking water provision and recreation. Human activities obviously affect performance of ecosystem process with resulting impacts on production of ES. These effects may be positive or negative.

There has been a significant effort made by environmental economists to estimate values for ES. Examples of recent papers include Brander et al (2012), Gascoigne et al (2011), Johnson et al (2012), and Ma and Swinton (2011). The results from this type of research have significance in terms of contributing to the ability to quantify ES in a "green accounting" framework. Given the increasing importance of the environment in public policy, having estimates of the value of ES also provides guidance to policymakers.

Agriculture and Environmental Services

The agricultural sector and associated activities contribute to ES production. As with other human activities, the effects of agricultural production practices on ES production may be positive or negative. If used inappropriately, for example, conventional tillage practices will result in reduced soil quality through increased soil erosion. Conversely, taking land out of production and planting shelter belts would improve soil quality (i.e., "increased" ES production) through a reduction in soil erosion. In some cases, specific practices may have mixed effects. For example, wetland drainage results in additional land being available for crop populations (i.e., enhanced harvest benefits) while at the same time reducing "natural areas" (i.e., diminished aesthetic benefits).

There has been ongoing and increasing public interest in and attention to the role that agriculture plays in production of ES. This has resulted in research and extension activities to identify practices that contribute positively to environmental stewardship in agriculture. These practices are referred to as Beneficial Management Practices (BMPs). There are multiple definitions in the literature of what exactly constitutes a BMP (e.g., list provided by Brethour et al 2007). Commonalities in the various definitions are a reference to protection or improvement of "environmental quality" as well as maintenance of economic viability or performance. For the purposes of the discussion here, a BMP is defined as a management practice or land use that results in avoidance of or reduction in environmental risks.[1] Examples of practices considered to be agricultural BMPs include buffer strips around riparian/wetland areas, nutrient management programs, manure management, shelter belts, etc.

Through their effects on environmental risk, agricultural BMPs are linked to production of ES. Examples of BMPs and associated ES and resulting human benefits are provided in Figure 1. For example, establishing buffer strips (i.e., leaving a strip of land idle to serve as a buffer) around riparian and/or wetland areas may have a number of effects, including enhanced provision of wildlife habitat; this is ecosystem "function". The resulting ES is increased bird population which provides human benefits in terms of recreation (i.e., bird watching, hunting, etc.). Similar "flows" from BMP to ES provision can be identified for the other three examples in Figure 1.

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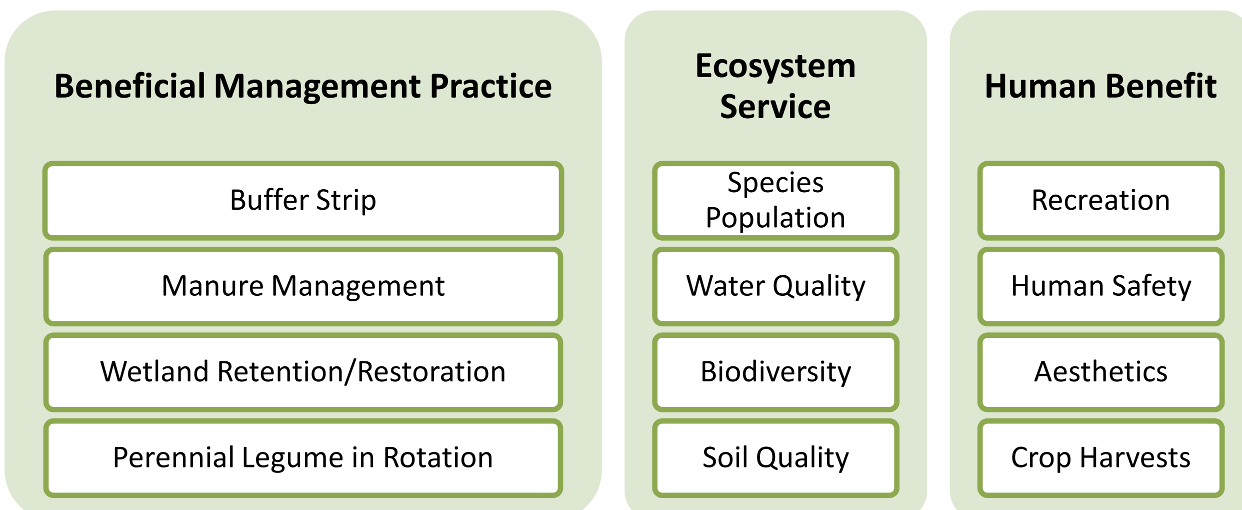
[1] This definition combines elements of others provided in the literature (e.g., AARD 2006; Boxall et al 2008; Brethour et al 2007). No reference is made to economic viability in this definition, however. The impact on farm performance of production practices identified as agricultural BMPs is still an "open question". As well, to date much of the work suggests that many agricultural BMPs have a net positive farm level cost associated with their adoption.

Economic Problem

As noted above, adoption of agricultural BMPs is linked to mitigated environmental risk which in turn leads to increased production of ES. This additional ES production has value to society (i.e., positive net public benefits) which, in many cases, is "non-market" in nature. However, there is some evidence (e.g., as is presented and discussed in this paper) that this increased ES production comes at a net cost to producers who adopt the associated BMP. Given the nature of a number of different agricultural BMPs, this is not surprising. For example, setting aside land to create a buffer strip around a wetland (and associated riparian area) promotes production of multiple ES (e.g., natural land cover, biodiversity, wildlife species populations, water quality) that are of value to society. However, a (potentially) significant opportunity cost accrues to the agricultural producer as a result of this land use change in the form of foregone returns from crop production on the area that is converted to the buffer strip. Similar arguments can be made for other types of agricultural BMPs.

Figure 1: Agricultural Beneficial Management Practices and Ecosystem Services



Given the discrepancy between the nature of public benefits (positive) and private benefits (negative) associated with ES production resulting from adoption of BMPs, it is likely the case that policy intervention is required to achieve socially optimal levels of ES from agriculture. A relevant question then is "What type/s of policy instrument/s is/are appropriate for governments to consider in order to encourage sufficient rates of adoption of BMPs by agricultural producers?" A number of options are available, including command and control, subsidies or taxes and market instruments (e.g., auctions).

One way of considering this problem is through the use of a policy analysis framework developed by Pannell in his 2008 *Land Economics* paper (Pannell 2008). Pannell's framework uses the relative magnitude of net public and private benefits resulting from a land use change as the basis for identifying an appropriate policy instrument.[2] In order to make empirical use of this type of framework to inform agricultural/environmental policymaking, estimates of both public and private benefits for alternative land use practices (i.e., BMPs) are required.

Given the nature of the value associated with many types of ES (e.g., biodiversity, natural land cover), it is a challenge to develop estimates of public benefits for many BMPs. As noted earlier, environmental economists have undertaken non-market valuation studies for some types of environmental services. However, limited work has been done to examine and quantify the net private benefits (or costs) associated with adoption of agricultural BMPs, particularly in a Canadian setting.[3] Accurate estimates of the farm level economic effects of BMP adoption are required in order to advise policy makers in terms of appropriate programs and instruments, and currently this information is incomplete. This is the economic problem addressed by the current paper.

Objectives

In response to the information gap concerning farm level effects of BMP adoption (i.e., net private benefits associated with agricultural production of ES), a number of empirical farm level studies have been undertaken within the Department of Resource Economics and Environmental Sociology (REES) at the University of Alberta. These studies examine the impact on farm financial performance of adopting a variety of land use changes or other agricultural management practices that have been shown or are hypothesized to have a positive effect on environmental quality. The overall objective of this paper is to review these studies, and to summarize their results in the context of provision of ES. A secondary objective is to further assess the implications of the empirical results from these studies with respect to policy development; that is, place them within the context of the Pannell policy framework.

[2] A more detailed summary of Pannell's framework is provided in the appendix of this paper. As per the terminology used by Pannell (2008), "private benefits" are the benefits that accrue to the land owner/decision maker as a result of making a land use change. "Public benefits" are the benefits that accrue to everyone else (i.e., social benefits)

[3] Brethour et al (2007) represents one of the few Canadian studies to examine the farm level economics of BMP adoption. The authors use survey data and provincial budgets to estimate the impact of BMP adoption on profitability for a set of representative cropping farm operations. In terms of current research, one objective of the Watershed Evaluation of Beneficial Management Practices (WEBs) program (part of the AAFC's Growing Forward policy framework) is to measure the economic impact of adopting BMPs that affect water quality.

The remainder of the paper is divided into several sections. The next section provides an overview of the empirical studies reviewed in the paper; specifically, a summary of methodologies used, farm characteristics (i.e., location, size, enterprises), and types of BMPs or land use practices modeled. This is followed by a discussion of the empirical results for the studies. The results are “grouped” not by project but instead by type of land use change or management practice, and associated ES production. The subsequent section places the empirical results, combined with available information from the literature concerning the public benefits associated with BMP adoption, within the Pannell policy framework to provide an initial assessment of policy implications. The paper concludes with a discussion of lessons learned and suggestions for further research.

II. Empirical Studies

This paper focuses on the research done in four separate studies, which were completed by REES graduate students between 2005 and 2011. All four studies examine the farm level economics of agricultural management practices and land use changes that affect ecosystem functions/processes and associated ES in the Canadian Prairie region. The studies are consistent in terms of the general empirical methodological approach employed in the research; that is, representative firm analysis and dynamic Monte Carlo simulation techniques. More detail concerning empirical methods is provided below.

While there are similarities between the studies, they also are each unique in terms of the specific issue or problem addressed in the research. The studies examine farms in different geographic locations within the Canadian Prairie region, producing different types of agricultural commodities (i.e., crops and/or livestock). The types of management practices and land use changes examined also vary between the studies. This provides an opportunity to gauge the degree of heterogeneity in private net benefits (i.e., using the term from Pannell’s policy framework) associated with agricultural production of ES from BMP adoption. Table 1 provides a summary of the characteristics for the four studies.

Koeckhoven (2008) – Southern Alberta

Koeckhoven (2008) examines the direct farm level costs and returns associated with BMP adoption for a mixed cropping and beef operation in the Lower Little Bow (LLB) watershed. The LLB watershed is part of the Oldman River Basin in southern Alberta. The representative farm

Table 1: University of Alberta Farm-Level Land Use/Management Change Studies - Summary of Study Characteristics

Study	Farm Location		Size (hectares)	Enterprises	Land Use/Management Practice Changes
	Province	County/Rural Municipality			
Koeckhoven (2008)	AB	Lethbridge (Lower Little Bow Watershed)	5115	Beef/Crops	Buffer Strips/Permanent Cover Off-Stream Watering Cattle Exclusion from Stream
Dollevoet (2010)	SK	Silverwood (Lower Souris Watershed)	777	Beef/Crops	Land Conversion (riparian/forested to cropland/pasture) Pasture Management
Trautman (2012)	AB	Taber (Brown) Forty Mile (Brown) Starland (Dark Brown) Camrose (Black) Smoky River (Dark Grey)	1036 1295 1295 1036 777	Crops	Crop Rotations Shelterbelts Buffer Strips/Permanent Cover Residue Management
Cortus (2005)	SK	Emerald	259-1295	Crops	Wetland Drainage

operation, located in Lethbridge County, is 5115 hectares (ha) in size. Much of this area (3691 ha) is devoted to pasture (both native and tame), with the remaining area being split between annual crops (906 ha) and perennial forage (518 ha). Annual crops include durum and spring wheat, barley and canola. Forage crops grown are alfalfa-grass hay and barley silage.

Koeckhoven's study examines the impact of adopting BMPs that are associated with changes in water quality.[4] There is a stream flowing through the representative farm, resulting in lotic wetland and adjacent riparian zones.[5] The BMPs considered in the economic analysis include two associated with annual crop production and two associated with pasture management. Cropping BMPs modeled are establishment of a buffer strip (i.e., a strip of land "retired" from production) or permanent cover (i.e., strip of land permanently converted to perennial forage, which is harvested) as a "cushion" between the area of crop production and the riparian and wetland areas. Pasture BMPs modeled are combinations of off-stream watering to reduce cattle use of riparian and wetland areas, and construction of fences to exclude cattle from accessing riparian/wetland areas.

Dollevoet (2010) – Southeastern Saskatchewan

Dollevoet (2010) investigates the farm level economics of land use changes related to wildlife habitat preservation or restoration. The economic analysis is done for a representative mixed cropping and beef operation in the Lower Souris watershed. This watershed is located in southeastern Saskatchewan.[6] The representative farm operation is 777 ha in size. Of this area, 350 ha are planted to annual crops, including spring wheat, barley, canola and flax. An additional 117 ha are planted to an alfalfa-grass hay mix, and 207 ha are in pasture (evenly split between native and tame pasture). Besides the land in crops and pasture, an additional 104 ha are assumed to be left as "habitat".[7]

[4] Koeckhoven's study was funded through AAFC's WEBs program, which was focused on water-related BMPs.

[5] There are varying definitions of the terms "wetland" and "riparian" areas in the literature. For the purposes of this paper, the term wetland refers to an undrained area (typically with hydric soil) that is saturated with water for at least part of the growing season, and that supports hydrophytic vegetation. Lotic wetlands are associated with running water (e.g., a stream); lentic wetlands are associated with standing water (e.g., a marsh or pothole). Riparian areas are transitional areas between wetlands and uplands.

[6] The information used to define representative farm characteristics in Dollevoet's study came from several sources, including Agricultural Census data and expert opinion. The location of the farm, based on the weather and yield data used in the simulation analysis, is the rural municipality of Silverwood which is at the north end of the Lower Souris watershed, within the Pipestone Creek sub-watershed.

[7] The size of the habitat area is based on expert opinion for that region. This area is assumed not to be "managed" by the producer, but is used for grazing by cattle if adjacent to pasture area and as part of aftermath grazing if adjacent to cropped fields. Depending on the specific land use scenarios modeled by Dollevoet, this area is

The majority of the land use changes examined in Dollevoet's study involve either conversion of land from riparian/forested habitat to active agricultural production or conversion of land to increase habitat. Land use change scenarios are modeled whereby riparian or forested habitat is converted to either crop production or tame pasture. These land use change scenarios would result in reduced ES production (i.e., negative net public benefits). Alternative scenarios are also modeled in which land is converted from annual crop production to either perennial forage (i.e., hay) production or tame pasture. Given the potential for improved wildlife habitat associated with those land uses, these scenarios would increase ES production, with associated net public benefits.

One other set of land use scenarios is modeled by Dollevoet; reduced stocking rates and rotational grazing for area devoted to tame and native pasture. These strategies may be considered to have a positive effect on wildlife habitat in that the intent is to increase the amount and quality of forage available in the pasture areas (i.e., available to both cattle and wildlife). However, these land use changes likely also contribute to other types of ecosystem functions and associated services (e.g., reduced grazing pressure leading to improved soil quality and natural land cover).

Trautman (2012) – Alberta

Trautman (2012) examines the direct farm level costs and returns associated with BMP adoption for a set of five representative cropping operations in Alberta. The primary characteristic used to "locate" the five farms is soil zone, with farms being defined for the brown, dark brown, black and dark grey soil zones. Agricultural census and provincial agricultural statistics, along with expert opinion, are used to define one farm per soil zone. The exception to this is for the brown soil zone, where two farms are defined; irrigated and non-irrigated (i.e., dryland). All other farms are assumed to use dryland production practices.[8] The representative farms vary in size, according to what is typical in each soil zone region, from 777 ha in the dark grey soil zone to 1295 ha for both the brown soil zone (dryland) and dark brown soil zone. The base rotations for all five farms consist of annual crops, with spring wheat, canola and barley being predominant in most cases. The rotation for both brown soil zone farms also include durum wheat, while the irrigated brown soil zone farm does not grow barley

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alternatively assumed to be "riparian habitat" or "forested habitat". Please note that the individual areas do not sum exactly to the overall farm size, due to rounding.

[8] County or municipal district (M.D.) crop yield data were used in the farm simulation modelling. On this basis, the five representative farms may be said to be located in Taber (brown soil, irrigated), Forty Mile (brown soil, dryland), Starland (dark brown soil), Camrose (black soil) and Smoky River (dark grey soil).

but does include dry beans in the base rotation. Summerfallow is included as a land use for the dryland brown and dark brown soil zone farms.

Trautman's study examines the impact of adopting a variety of BMPs that are associated with increased agricultural production of ES.[9] These are divided into two general groups; crop rotation BMPs and non-rotational BMPs. Crop rotation BMPs considered by Trautman are generally changes to the rotation to incorporate legumes; alfalfa hay, field peas or red clover. In all three cases, benefits are assumed to accrue in the form of reduced chemical fertilizer use in subsequent years due to nitrogen being fixed in the soil by the legume. This reduces the potential for runoff or leaching of chemicals (i.e., improved groundwater or surface water quality). In the case of red clover, the crop is grown as a green manure crop and rather than being harvested it is worked back into the soil to add organic matter, thus improving soil quality. An additional rotational BMP is modeled in which oats are added to the rotation. Oats require fewer chemical input requirements (both fertilizer and pesticides) than other cereals modeled in the analysis, again reducing the potential for chemical runoff or leaching.

A number of non-rotational BMPs are modeled by Trautman as well. These include establishment of shelterbelts which contribute to improved soil quantity and quality, or buffer strips/permanent cover which contribute to improved water quality (i.e., as per Koeckhoven 2008). The other non-rotational BMP examined by Trautman is residue management. For this BMP, crop residue is assumed to be left on the field in some years. This contributes to ES production in two ways. First, as the residue breaks down it adds to the organic content of the soil (i.e., soil quality). Also, the residue acts as a "cover" to enhance retention of soil moisture (i.e., water availability).

Cortus (2005) – East Central Saskatchewan

Cortus (2005) examines the farm level economics of wetland drainage for a representative cropping operation in Emerald, which is a rural municipality in east central Saskatchewan. This makes Cortus' study somewhat different from the other three cited here in that it does not involve an examination of practices to increase ES production. Instead, the study quantifies the

[9] Not all BMPs are modeled for all representative farms. Decisions regarding BMPs to model for each farm are made based on information from relevant literature and expert opinion concerning feasibility in each soil zone. For example, incorporating alfalfa into the rotation is not modeled for the dryland brown soil zone farm, based on expert opinion suggesting that it is not viable from an agronomic perspective (i.e., insufficient moisture). The specific combination of BMPs chosen to be modeled for each farm is evident from the discussion of the study results, later in this document. Further detail is provided by Trautman (2012).

economic incentives (i.e., private net benefits) associated with a reduction in production of ES associated with wetlands on agricultural land (e.g., water availability, associated species populations, biodiversity, etc.).[10]

The representative farm modeled by Cortus has a base rotation that includes spring wheat, barley, flax and canola, along with some summerfallow. In order to examine the impact of farm size on incentives to drain wetlands, Cortus models multiple versions of the representative farm, ranging from 259 ha to 1295 ha in size. Also unlike the other studies reviewed in this paper, Cortus models drainage activities such that decisions as to whether or not to initiate drainage operations are made endogenously in the model, based on expected returns for individual projects in any given year. In the case of the other three studies, the land use changes under consideration are "imposed" exogenously on the analysis.

III. Methodology

All four studies reviewed here use representative farm analysis to examine the economic impact of changes in land use and/or management practices. Specifically, farm characteristics are defined in such a way that the operations modeled in the analyses can be considered as "representative" or "typical" for the region in which they are located. This representativeness includes farm size, presence and concentration of enterprises, as well as yields, prices and costs.

The methodology used to model adoption of BMPs in the various studies can be characterized as dynamic Monte Carlo simulation. Values for stochastic parameters are drawn from analyst-defined distributions and used to generate model outcomes, in this case values for financial performance of the representative farm. The process is repeated in order to generate distributions of outcomes. The simulation in these studies is dynamic in that performance for the representative farms is also modeled over a specified time period. In these four studies, the main stochastic parameters modeled in the simulation analysis are production and commodity prices. In other words, technical and market risk are incorporated into the analysis.

[10] The wetlands considered by Cortus (2005) are lentic in nature (i.e., standing water). The study region is located in the prairie pothole region which extends from the American Midwest up through much of the Canadian Prairie region.

Stochastic Production

Stochastic yields are modeled for annual cereal/oilseed crops, forage crops and pasture. Historical yield data at the county, municipal district or rural municipality level are obtained from crop insurance and other sources. In all of the studies except for one, growing season weather data (i.e., growing season temperatures and precipitation) are also collected for nearby weather stations. Functional relationships are then estimated for yields, where the explanatory variables are linear and quadratic terms for the ratio of growing season precipitation (GS) to cumulative growing degree days (GDD)[11]:

$$y_t^j = \alpha_o^j + \alpha_1^j \left(\frac{GS_t}{GDD_t}\right) + \alpha_2^j \left(\frac{GS_t}{GDD_t}\right)^2 + \varepsilon_t^j \tag{1}$$

Where y_t^j is the yield for crop j in year t, GS and GDD are defined as above, the αs are parameters to be estimated and ε is the error term. The (GS/GDD) ratio represents a proxy for water availability relative to water demand. Within the simulation, draws are made from GDD and GS distributions and used to calculate stochastic yields. SUR estimation is used to estimate systems of yield equations and the error correlations are used in the simulation.[12]

As noted earlier, the yield data used in these studies are at the county (or equivalent) level. Using county yield data results in a lower measure of variability than would be the case at the farm level (Marra and Schurle 1994). An adjustment suggested by Marra and Schurle (1994) is used in all of the studies to adjust the variability in the yield distributions upward accordingly.

Two of the studies (Koeckhoven 2008; Dollevoet 2010) also include cow-calf enterprises for their representative farms. Beef production is not explicitly modeled as being stochastic in either study. However, production is linked to pasture productivity. A certain amount of forage is required to achieve the target selling weight for weaned calves. This is largely provided through pasture “production”. However, pasture capacity is stochastic in both studies. In cases where pasture is insufficient to support the required weight gain, it is assumed that the calves

[11] Growing degree day is a proxy for heat accumulation on a particular day, typically calculated as the difference between the average of the maximum and minimum temperatures and some minimum critical temperature required for plant growth (McMaster and Wilhelm 1997).

[12] The one exception to this approach is Trautman (2012). In Trautman’s study, yield values are drawn directly from distributions of yields. Correlations between yields for different crops are also incorporated into the process of generating the annual yield values. The reason for this approach is that it was not possible to obtain defensible yield-weather functional relationships for some crops on some of the representative farms.

are placed in drylots and fed until they reach the target weight. If pasture productivity is "higher than average", it is assumed that the calves are sold at a weight that is higher than the target. As a result, beef weight gain is indirectly stochastic, through the link to pasture productivity.

Stochastic Prices

Prices for both beef and crops are modeled as being stochastic or "risky" in these studies. In all cases, historical provincial level prices are collected from secondary sources, deflated to real values, and tested for stationarity. Based on the results of these tests, time series models are estimated for all commodity prices.[13] Tests are done to establish appropriate lag lengths for each commodity and the price models are estimated using a systems approach (i.e., using SUR). In the cases where there are both crops and beef enterprises, separate cropping and livestock price equation systems are estimated. Random error draws and price equation error correlations are used, along with the time series model parameter estimates, to calculate annual commodity prices used in the simulation analysis.

Additional Stochastic Considerations in Cortus (2005)

In the case of the Cortus study on wetland drainage, two additional stochastic parameters are incorporated into the Monte Carlo simulation analysis. The initial wetland configuration (i.e., number, size and shape of wetlands in each field) is changed for each iteration of the model. A "distribution" of potential wetland configurations is generated using GIS mapping for the study region. At the beginning of each model run, a different set of wetlands is drawn for use in defining the farm characteristics. This is done in order to avoid having a particular configuration influence the wetland drainage results.

The other additional stochastic parameter is time available for the producer to undertake drainage activities each year. An assumption in the analysis is that drainage activities are performed in the fall, after crop harvest is complete. In reality, the amount of time available for these activities will vary from year to year, depending on when harvest begins and ends. This, in turn, is dependent on weather conditions. Historical weather data and information from relevant literature are used to estimate a simple distribution of days available to do drainage

[13] It should be noted that using the "standard" Augmented Dickey Fuller test for stationarity, many of the crop price series used in the four studies are not stationary. As discussed by both Cortus (2005) and Koeckhoven (2008), there are limitations (related to both data and statistical power) associated with the use of this test. As well, using a different test (Kwiatkowski-Phillips-Schmidt-Shin or KPSS test) Trautman (2012) determined the majority of the price series in her study to be stationary.

activities. Annual draws are made from this distribution and used in the simulation analysis to determine feasibility of initiating or continuing wetland drainage.

Time Horizon

Many of the land use changes and management practices considered in these studies have a dynamic dimension to them; that is, implementation and effects of the changes occur over several years (e.g., establishment of a shelter belt, drainage of a wetland). As a result, a multi-year time horizon is used in the simulation models. In all studies but one, the time horizon is 20 years. This value is seen as sufficiently long to capture any effects of the changes. Trautman (2012) uses a 40 year time horizon. The longer time period is chosen because of the length of time required for shelter belt trees to mature.

Economic Relationships and Performance Measures

A modified net cash flow (MNCF) measure is calculated for each year within the simulation model for the representative farms. MNCF is the sum of farm revenues and business risk management (BRM) payments, minus the sum of variable expenses and annual machinery depreciation expense. Revenue includes returns from sales of crop and/or livestock. Variable expenses include those expenditures related to inputs used in production of crop and livestock commodities.[14] The machinery depreciation expense represents the annualized cost of maintaining the farm machinery complement at the initial book value. As such, it represents a proxy for machinery replacement expenditures. BRM program payments include crop insurance and AgriStability[15] payments.

Net present value (NPV) is the primary performance measure used in the four studies to evaluate the impact of land use/management practice change scenarios. NPV is used as a proxy for wealth and is compared between scenarios to determine whether a particular land use change has a positive or negative net private benefit. In general, NPV is equal to the present values of future cash flows summed over the relevant time horizon, minus any initial expenditure for the investment under consideration. In the case of the NPVs calculated in these studies, there is no initial investment expenditure. As well, the NPVs are calculated

[14] Debt servicing cash flows (i.e., principal and interest payments) are not included in these calculations and in fact no assumption is made in any of the studies concerning the capital structure of the representative farm businesses. This is done to avoid having capital structure decisions influence the model results with respect to the land use/management practice change scenarios. The analyses are all done on a before-tax basis.

[15] In the studies by Cortus (2005) and Koeckhoven (2008), participation in the Canadian Agricultural Income Stabilization (CAIS) program is modeled. CAIS preceded AgriStability as the main BRM “stabilization” program.

assuming an infinite time horizon; that is, NPV in perpetuity. This is done by taking the MNCF in the last year of the simulation and assuming that it occurs annually in perpetuity; that is, a perpetual annuity. The present value of this perpetual annuity is calculated and added to the NPV over the 20 or 40 year time horizon to obtain the NPV in perpetuity for each scenario.

Simulation Scenarios

Monte Carlo simulation methods are used to model various scenarios for the representative farms defined in the four studies. The farm models are built using @Risk software (Palisade Corporation 2010), which is a spreadsheet based program. Baseline scenarios for the representative farms are initially simulated in order to generate a starting point for comparison purposes. The various land use and management practice change scenarios are then simulated. The adjustments made to the baseline scenarios depend on the nature of the change being modeled. For each alternative scenario, information from existing literature and expert opinion is used to try and ensure that the impacts on relevant parameters (e.g., costs, yields, crop areas) are incorporated into the simulation models.

When modelling the alternative scenarios, in most cases the assumption is made that the representative producer implements the specific change starting in the first year of the simulation. As a result, some of the adjustments are "instantaneous"; for example, with changes to crop rotations it is assumed that the producer shifts to the revised rotation at the beginning of the time horizon. In other cases, the adjustments occur gradually over multiple years. An example of this is the establishment of shelter belts. It takes time to plant trees and so this is spread out over multiple years in the simulation to ensure feasibility.

In all but one of the studies, the decision to implement specific land use changes or revised management practices is exogenous to the model. There is no optimization done within the simulation analysis. The exception to this "rule" is the wetland drainage study by Cortus (2005). In Cortus' study, decisions to initiate drainage projects in any given year are made based on a calculation of the expected net returns; that is, the decisions are endogenous to the model. The expected return associated with all undrained wetlands on the representative farm is calculated in each year, taking into account the costs of drainage and the expected future crop returns that could be generated on the drained land[16]. If there are projects with positive expected returns and time available in the fall to undertake new drainage activities, the "best"

[16] An expected NPV for each project is calculated; a positive expected value represents a viable drainage project. Expected returns are calculated using historical average simulated yields and prices from the model.

project is initiated. If not, then no new projects are started. Once drainage projects are started, they are completed over a multiple year period, again depending on time available each year to do drainage.

Evaluation of Land Use/Management Practice Change Scenarios

Expected NPVs generated by the dynamic Monte Carlo simulation analysis are used to evaluate the alternative land use/management practice change scenarios. The expected NPVs for the scenarios to be compared, the baseline and the "change", are first converted to annualized values. Since the NPV is in perpetuity, the perpetual annuity formula is used for this purpose:

$$PV = \frac{A}{i} \tag{2}$$

where PV is the present value of the annuity, represented by the expected NPV in this case, *i* is the discount rate (equal to 10% or 0.10 in all studies) and A is the annuity. In this case, A is the annualized NPV to be calculated.

The difference between the annualized NPVs (i.e., NPV for change scenario – NPV for the baseline scenario) is calculated as the net impact of the change under consideration. A positive value represents a net positive private benefit associated with the change, with a negative value representing the opposite case. The change is then converted to a value on a per unit of land (i.e., hectare) basis. This is done to express the benefit (or cost) in terms (i.e., scale) consistent with what might be considered in policy instruments (e.g., a tax or subsidy per hectare). For changes that affect the whole farm (e.g., shifts in crop rotation), the area of the whole farm is used for this conversion. In the case of changes that affect part of the farm, the area affected is the basis for the conversion. For example, establishment of shelterbelts results in an area of land being removed from production to accommodate planting and growth of the trees. In the case of this change, the area removed from production is used as the basis for conversion.

IV. Results

As discussed earlier, the four studies model adoption for a range of changes in land use and/or management practices, all of which contribute to ES production. In this section, the results for the studies are reviewed and summarized. The results are grouped according to the nature of the land use or management practice. Specifically, four categories of practices are discussed; maintenance/restoration of natural areas (e.g., wetlands, riparian area), protection of natural areas (e.g., buffer strips, shelter belts), management of crops (e.g., changes in crop rotations), and management of pasture (e.g., rotational grazing). In each case, the practices are placed in context in terms of the types of ES being produced and subsequent human benefits, with a subsequent discussion regarding the range of private net benefits estimated from the farm level modelling.[17]

Maintenance/Restoration of Natural Areas

Natural areas on farm operations include wetlands and associated riparian areas, forested areas (e.g., woodlots), etc. As shown in Figure 2, these areas provide a number of ES (e.g., natural land cover, wildlife habitat) with associated human benefits. However, these areas also represent potentially productive agricultural land and so there are opportunity costs associated with maintaining them. Cortus (2005), Koeckhoven (2008) and Dollevoet (2010) all examine the direct farm level economics of maintaining these types of natural areas; specifically, wetlands, riparian area and forested area. Table 2 provides a summary of simulation results given a scenario of maintaining the areas assumed to be on the representative farms.[18]

As shown in Table 2, maintaining these natural areas is costly; the net private benefits for all scenarios are negative. In the Cortus study, the annual cost of maintaining wetlands for the Emerald representative farm ranges from $28 to $41 per hectare. Although not obvious from Table 2, the opportunity cost increases with the size of the representative farm. The cost arises from a combination of opportunity cost of foregone production and nuisance costs of maneuvering equipment around the wetland areas.

[17] The four studies were completed between 2005 and 2012. In order to allow consistent comparisons to be made between the estimates of private benefits, all values are converted to 2008 dollars and expressed on a per hectare basis.

[18] In fact, the scenarios modeled in both Cortus (2005) and Dollevoet (2010) are conversion of these natural areas, wetlands and forested area respectively, over to agricultural production. In both cases, the conversion results in positive net benefits, which may be interpreted as the opportunity costs of maintaining the original natural areas.

Figure 2: **Examples of Ecosystem Services and Human Benefits Derived from Maintaining Natural Areas**

Table 2: **Summary of Study Results - Net Private Benefits Derived from Maintaining Natural Areas**

Natural Area	Study	Net Private Benefits ($/ha/year)
Wetlands	Cortus (2005)	-$28 to -$41
Riparian Area	Koeckhoven (2008)	-$316 to -$533
Riparian Area	Dollevoet (2010)	-$197 to -$209
Forested Area	Dollevoet (2010)	-$111 to -$127

Both Koeckhoven (2008) and Dollevoet (2010) provide estimates of net private benefits associated with maintaining riparian areas. In the case of Koeckhoven, the annual costs per hectare maintained are significant, ranging from $316 to over $500. A significant portion of this cost is attributable to the requirement for fencing in Koeckhoven's analysis. The riparian area in question is adjacent to pasture and so fencing is required to exclude cattle either temporarily or permanently from this area. The range of costs also depends on the percentage of riparian area actually protected.[19] As the fencing cost is spread over a greater area, the cost per hectare decreases somewhat. The results from Dollevoet's analysis are somewhat different, with the net costs being significantly lower than Koeckhoven's estimates. The difference in the magnitude of the costs is primarily due to fencing not being required for riparian area protection on Dollevoet's representative farm. The riparian area in Dollevoet's study is adjacent to cropland, so no fencing is required. As a result, the range of values from Dollevoet's analysis is also much smaller.[20]

Dollevoet also provides results for maintaining forested areas on the representative farm in his study. The opportunity cost of maintaining forested area is modeled relative to conversion to cropland or tame pasture. In both cases there is a net cost, ranging from $111 to $127 per hectare per year, associated with the foregone opportunity for using the land in agricultural production.

The implication to be drawn from the results summarized here is that maintaining natural areas on cropping and beef operations in Western Canada is costly to producers. This is largely due to the opportunity cost, which varies depending on geographic location and type of agricultural enterprise involved. However, other factors (e.g., fencing requirements, nuisance costs) also play a role.

Protecting Natural Areas

Some of the land use changes modeled in the four reviewed studies involve taking land out of agricultural production in order to provide protection for natural areas. Buffer strips are areas of land that are set aside from production in order to provide a buffer for riparian and wetland areas. The buffer area serves to reduce runoff of sediment and agricultural chemicals. Permanent cover is similar to a buffer strip, except that the land is permanently seeded to

[19] For most of the BMP scenarios modeled by Koeckhoven (2008), alternative versions are modeled assuming differing degrees of protection, from 25% up to 100% of the total riparian area present on the farm.
[20] Dollevoet (2010) also modeled alternative degrees of conversion of riparian area to cropland.

perennial forage, which may be harvested and/or grazed. A shelterbelt is a line of trees planted to reduce wind erosion of soils. As illustrated in Figure 3, establishing buffer strips, permanent cover or shelter belts provide habitat for wildlife and/or waterfowl, improved soil and/or water quality, natural land cover, etc. All of these ES yield human benefits. As with the previous category of land use changes, however, there are also opportunity costs associated with the changes. The studies by Koeckhoven (2008) and Trautman (2012) examine the farm level economics of these types of practices. A summary of the results is provided in Table 3.

Both Koeckhoven and Trautman model adoption of buffer strips on their representative farms. As shown in Table 3, the net private benefits associated with these changes are strongly negative. In the case of buffer strips, the annualized cost per hectare converted ranges from $95 to $465. The net impacts estimated by Trautman are lower than those from Koeckhoven's study. Trautman models adoption by crop producers with the level and variability in net cost being attributable to differences in opportunity costs of foregone crop production across the soil zones. In the case of Koeckhoven's study, the cost has two components; the same type of opportunity cost, plus fencing cost due to the presence of a beef enterprise and the need to exclude cattle from the buffer strips.

The same two studies also model adoption of permanent cover. As is the case with buffer strips, the net private benefits for this land use change are negative (Table 3). Also similar to the previous land use change, and for the same reasons, the cost estimates from Trautman's study tend to be lower than those for Koeckhoven's study, although the "top end" of costs is higher in Trautman's study due to significant opportunity costs for some of her farms. The value of the additional hay production to the beef enterprise modeled in Koeckhoven's analysis dampens the net cost somewhat for that representative farm.

Trautman models the establishment of shelterbelts for all five representative farms in her study. As shown in Table 3 the net private benefits for this land use change are strongly negative, with the annualized cost per hectare converted to shelterbelt use ranging up to $411. As with buffer strips and permanent cover, there is an opportunity cost for the land used to plant the trees. In addition, there is a yield effect as the presence of a shelterbelt affects crop yields in adjacent fields. Close to the trees, the effect is negative (i.e., reduced yields) while

Figure 3: Examples of Ecosystem Services and Human Benefits Derived from Protecting Natural Areas

Table 3: Summary of Study Results - Net Private Benefits Derived from Protecting Natural Areas

Practice	Study	Net Private Benefits ($/ha/year)
Buffer Strips	Koeckhoven (2008)	-$450 to -$465
Buffer Strips	Trautman (2012)	-$95 to -$347
Permanent Cover	Koeckhoven (2008)	-$194 to -$198
Permanent Cover	Trautman (2012)	-$20 to -$280
Shelter Belts	Trautman (2012)	-$181 to -$411

further away the effect is positive.[21] The area affected by these yield effects changes with the growth in the trees, but the net effect on financial performance is negative.

As is the case with maintenance of natural areas, protecting natural areas present on cropping and beef operations in Western Canada is costly to producers. The explanation is also similar, as the effect is largely due to the opportunity cost. This makes it less likely, therefore, that there would be extensive adoption of these land use changes without policy intervention.

Crop Management Practices

Figure 4 provides generic examples of crop management practices that generate ES and associated human benefits. These types of changes, unlike the ones associated with natural areas, sometimes have direct positive effects on crop production (i.e., crop population ES and harvest benefits). Therefore, a priori expectations regarding net private benefits for these types of practices are not quite as clear.

Trautman (2012) models adoption for a number of crop management practices that contribute to ES production. These include changes in crop rotations (i.e., adding crops to the rotation) and adoption of residue management. The crop rotation changes modeled by Trautman include the addition of legumes (either alfalfa or field peas) to rotations for the representative farms. The agronomic impact of this type of change is to provide additional nutrients for subsequent crops through the ability of legumes to “fix” nitrogen in the soil. This direct benefit (i.e., more nitrogen available to support plant growth) should contribute positively to net private benefits for the change in management practice. An indirect effect of the change is that by reducing the need for chemical fertilizer (i.e., substitute fixed nitrogen for applied nitrogen) there is also a reduced chance of agricultural chemical runoff and leaching. This contributes positively to ES such as water quality. A similar effect may occur through the addition of oats to crop rotations, which is also modeled by Trautman. While not a legume, oats require lower levels of agricultural chemicals (especially pesticides) which in turn means reduced chance of chemical runoff.

The other change in crop rotation modeled in Trautman’s study is inclusion of a green manure crop. A green manure crop is a crop that is not grown for commercial production but instead is grown and then “worked under” and incorporated back into the soil. The purpose of growing a

[21] The pattern of yield effects used by Trautman is based on evidence from available literature on the use of shelterbelts.

green manure crop is to increase soil organic matter and thus improve soil quality. This has positive implications for future productivity of the soil. The tradeoff is that by growing a green manure crop, the producer is giving up the opportunity to produce a commercial crop on the field during that growing season. The green manure crop modeled by Trautman is red clover which is a legume. Besides the organic matter benefit, therefore, nitrogen is fixed by the clover for use in subsequent crops, similar to alfalfa and field peas.

The final crop management practice modeled by Trautman is residue management. For some crops (e.g., cereal grains) there is an opportunity to “harvest” and sell crop residue (i.e., straw). However, if residue for those crops is retained on the field there are potential benefits in terms of soil organic matter (i.e., ES such as soil quality). As well, crop residue also contributes to improved retention of soil moisture, another agricultural ES. In cases of excess moisture, however, this can actually have a negative effect on crop production (i.e., too much moisture or delayed warming of the soil). The residue management scenario modeled by Trautman involves retaining residue in these fields in some years.

The results for the various crop management practices modeled by Trautman (2012) are summarized in Table 4. The first pattern to note in this table is that the annualized effects of these practices are significantly smaller, in absolute value, than most of the costs associated with maintaining and protecting natural areas (i.e., results from Tables 2 and 3). There are a number of possible reasons for this difference, but it is probably at least partly due to the area over which the benefits are calculated. Unlike the previously discussed results for which the net impact is calculated per hectare maintained or converted, these effects are calculated per hectare of crop production. This is done because these changes (e.g., change in crop rotation) affect the entire farm area.

The second pattern of interest from the results in Table 4 is that there are positive net benefits in some cases. For the two crop rotation changes involving the addition of commercially produced legumes (i.e., alfalfa and field peas), there are (mostly) positive net benefits. This suggests that the combination of producing a crop for sale and the agronomic benefits of the legume crops results in positive benefits for the adopting producer.[22] Another factor contributing to the positive net benefits for the representative farms in southern Alberta

[22] Besides the benefit to subsequent crops of the fixed nitrogen, literature reviewed by Trautman (2012) also suggests that there is a separate positive yield benefit for crops following alfalfa or field peas in rotation. This yield benefit is also modelled in her analysis.

Figure 4: Examples of Ecosystem Services and Human Benefits Derived from Crop Management Practices

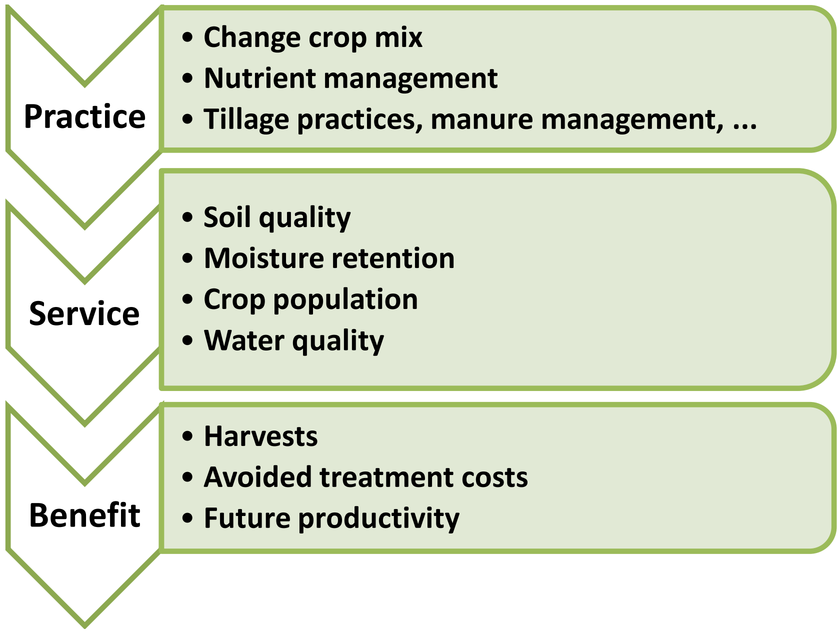


Table 4: Summary of Study Results - Net Private Benefits Derived from Crop Management Practices

Practice	Study	Net Private Benefits ($/ha/year)
Addition of Alfalfa to Rotation	Trautman (2012)	$64 to $32
Addition of Field Peas to Rotation	Trautman (2012)	$42 to -$2
Green Manure Crop	Trautman (2012)	-$5 to -$10
Addition of Oats to Rotation	Trautman (2012)	-$6 to -$29
Residue Management	Trautman (2012)	-$11 to -$26

(brown and dark brown soil zones) is that by adding another crop to the rotation, there is a reduction in the proportion of summer-fallow; that is, less land being idled and more used in commercial production. The one exception to this pattern of positive net benefits is for adoption of field peas by the farm in the black soil zone (i.e., net private cost of $2 per hectare), which is due to the opportunity cost of foregoing production of more highly valued cereal or oilseed crops.

Conversely, the other two crop rotation scenarios (i.e., addition of oats or a green manure crop) results in a negative net private benefit, although in some cases this is relatively small. In the case of green manure, the benefits of fixed nitrogen and associated reduction in chemical fertilizer requirements for subsequent crops are insufficient to offset the loss of a commercial crop for that year. The benefits of the additional soil organic matter with respect to increased soil productivity are long term in nature and are not modeled by Trautman. In the case of oats, the negative net private benefits are also due to the opportunity cost of growing this crop. Oats tend to be a lower profit crop and their addition to the rotation means reduced area for higher value crops (e.g., canola).

As shown in Table 4, residue management also represents a crop management practice associated with negative net private benefits. While straw is not a high value product, it can be harvested and sold by producers. Trautman's results suggest that the loss of this opportunity outweighs the potential benefits associated with retaining the residue. The short term impact on crop yields may be positive or negative depending on soil moisture conditions, as discussed earlier. Also, as is the case with the green manure crop, the effects of retaining residue on soil productivity through increased soil organic matter are long term in nature and not modeled in her study.

In summary, there is some potential for "win-win" situations with respect to the crop management changes reviewed here. In at least some cases, there may be opportunities for producers to achieve positive net benefits from making changes in crop management practices that also increase production of agricultural ES such as soil and water quality.

Pasture Management Practices

The final category of management practice change modeled in these studies is pasture management. Two of the studies, Koeckhoven (2008) and Dollevoet (2010), examine BMP scenarios for mixed cropping and livestock (i.e., cow-calf) operations. Both studies include changes in pasture management among the BMPs modeled in their analysis. Examples of pasture management strategies that potentially contribute to ES production are provided in Figure 5. These practices are generally intended to improve the quality and/or utilization of available pasture, which have direct effects on livestock enterprise performance and the potential for positive private benefits. Improved pasture quality also affects (positively) soil and/or water quality and, in the case of native pasture, biodiversity as well.

Koeckhoven (2008) examines two types of pasture management strategies. The first is establishment of off-stream watering (OSW) systems. This in itself is not "pasture management". However, one of the potential side effects of implementing OSW systems is that it may encourage cattle to make better use of more of the available pasture, versus congregating closer to streams or wetlands. This type of practice also potentially generates ES production through improved aquatic and riparian area "quality" as a consequence of the cattle spending less time in and around streams and wetlands. The second pasture management practice modeled by Koeckhoven is OSW combined with either temporary or permanent exclusion of cattle from riparian areas. This particular management practice change scenario has already been reviewed under "restoration of natural areas".

Literature reviewed by Koeckhoven suggests that there may be the possibility of greater rates of weight gain by cattle on pasture if they have access to OSW systems. The theory is that improved drinking water quality and pasture utilization would contribute to this result. Koeckhoven models alternative scenarios of his pasture management scenarios, incorporating increased calf weight gain on pasture, in order to examine the impact of these potential effects on the farm level economics.

Dollevoet (2010) models two pasture management scenarios in his study. The first scenario involves reducing stocking rates on tame and native pasture for the representative mixed farm operation. The other scenario modeled is rotational grazing. The purpose for both these changes is to allow pasture productivity to improve due to the reduced grazing stress. This has direct implications for the beef enterprise along with effects on ES production (e.g., improved biodiversity on native pasture if it is allowed to improve in "quality").

The simulation results for these pasture management scenarios are summarized in Table 5. For the changes involving establishment of OSW alone, decreased stocking rates or rotational grazing, the annualized change in NPV at the farm level is converted to a per hectare value by

Figure 5: **Examples of Ecosystem Services and Human Benefits Derived from Pasture Management Practices**

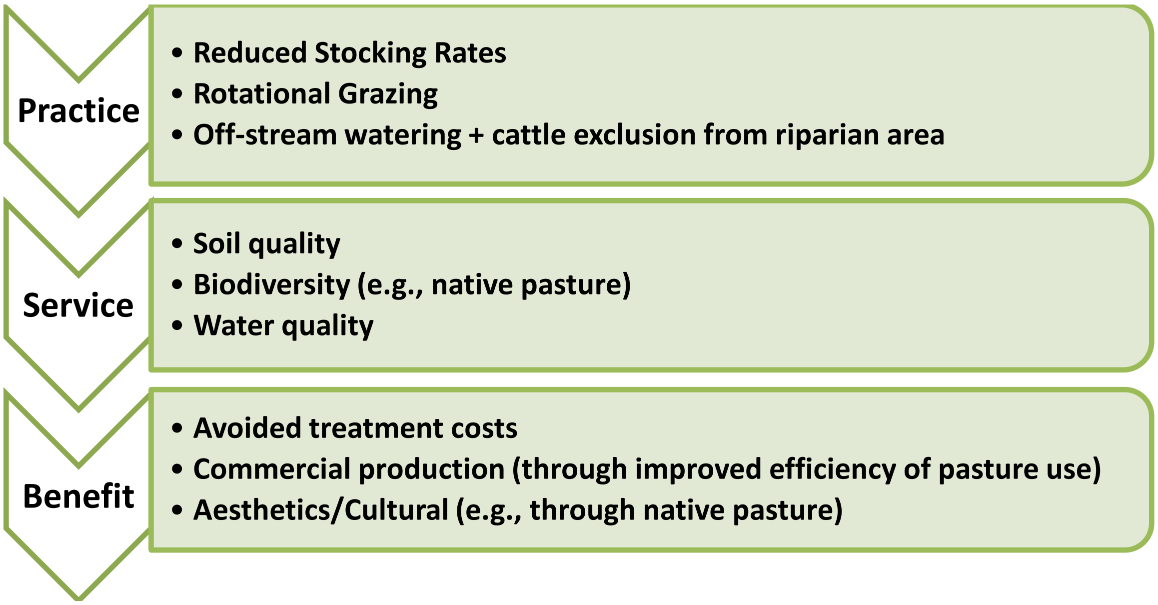


Table 5: **Summary of Study Results - Net Private Benefits Derived from Pasture Management Practices**

Practice	Study	Net Private Benefits ($/ha/year)
Off-Stream Watering[a]	Koeckhoven (2008)	-$0.96
Off-Stream Watering[a] (+ improved utilization)	Koeckhoven (2008)	$3.46 to $1.29
Decreased Stocking Rate[a]	Dollevoet (2010)	$0.88 to -$1.15
Rotational Grazing[a]	Dollevoet (2010)	$3.04 to -$4.47
Off-Stream Watering and Cattle Exclusion (temporary or permanent) from Riparian Area[b]	Koeckhoven (2008)	-$316 to -$533
Off-Stream Watering and Cattle Exclusion (temporary or permanent) from Riparian Area[b] (+ improved utilization)	Koeckhoven (2008)	$311 to -$309

[a]These values are on a per ha of pasture basis.
[b]These values are on a per ha of riparian area restored basis.

spreading the impact over the entire pasture area. This is done because the changes in question affect the entire pasture area. The results for the two scenarios from Koeckhoven's study that include temporary or permanent exclusion of cattle from riparian areas are expressed (as before) on a per hectare of riparian area basis.

The results shown in Table 5 are "mixed" in that the ranges include both positive and negative values. As well, some of the effects are large in magnitude while others are relatively small. The basic OSW strategy modeled by Koeckhoven results in a small negative net private benefit (i.e., net cost of $0.96 per year per hectare of pasture). If some degree of improved rate of weight gain is incorporated into the analysis (i.e., as discussed above), the results change such that establishing an OSW system actually generates positive benefits for the adopting producer, independent of the contribution to ES production. None of the per hectare values for the OSW scenarios are very large, in absolute terms.

The results for Dollevoet's pasture utilization change scenarios (i.e., reduced stocking rates or rotational grazing) range from small positive to small negative annual private benefits per hectare; +$0.88 to -$1.15 for decreased stocking rate, and +$3.04 to -$4.47 for rotational grazing. In other words, the effect on farm performance from adopting these practices is relatively minor when compared to some of the other scenarios reviewed in this paper. Whether the results for these scenarios are positive or negative depends largely on the degree and duration of resulting improvement in pasture quality and subsequent utilization by the beef enterprise on the representative farm operation. As the degree or duration of improvement increases, so do the net benefits.

For the last two pasture management scenarios from Koeckhoven's study, it is assumed that OSW is initiated along with construction of fencing to prevent cattle from having access to the stream and associated riparian area. The first set of results provided in Table 5 for this scenario (i.e., net annualized cost of $316 to $533) is identical to the results for restoration of riparian areas given in Table 2. As discussed earlier, the large net cost is attributable to the opportunity cost of lost returns from crop production in the riparian zone, and the cost of fencing required to exclude cattle from the area.

The second set of results for this management change incorporate varying degrees of improved rates of weight gain for the cattle on pasture. This improvement is attributable (as noted earlier) to benefits from better drinking water quality and improved pasture utilization. As shown in Table 5, depending on the degree of improvement modeled in the simulation analysis, the results for this scenario can be quite positive (i.e., up to $311 per hectare annually in private net benefits).[23]

In summary, the pasture management practices modeled in these studies (Koeckhoven and Dollevoet) are mixed in terms of providing net benefits to adopting producers. There are some opportunities for improved farm financial performance. However, generally speaking the degree of improvement is relatively minor and in at least some cases is not directly related to the BMP under consideration but instead is a side effect that might be achievable without implementing the BMP itself. Thus, caution should be taken in interpreting these results as indicating BMPs that hold potential for adoption independently of policy intervention.

V. Discussion and Policy Implications

From the study results reviewed in the previous section, a general pattern may be identified. It would appear that many if not most of the BMPs modeled in the four studies result in net costs for adopting producers. Thus, while their adoption may result in increased ES production they do not contribute positively to producer financial performance. For the most part, this is attributable to the opportunity cost associated with giving up commercial production. However, other factors do influence the overall effect, including farm size (e.g., wetland drainage results from Cortus suggest diseconomies of farm size in retaining wetlands), the degree of restoration or protection being considered (e.g., the impact of how much available riparian area is protected in Koeckhoven's BMP scenarios), and the cost of implementing the particular land use or management practice changes (e.g., cost of fencing or planting trees).

Having stated that, there are some exceptions to this general pattern. For some of the BMPs, it appears that there are "win-win" (or at least "win-no lose") opportunities. Some of the modeled BMPs do generate positive private net benefits along with the positive effects in terms of ES production. These include crop management practices such as addition of legumes to the rotation (Trautman 2012) and some of the pasture management practices from Koeckhoven (2008) and Dollevoet (2010). These positive net benefits are largely due to the side effects that "tag along" with the increased ES production. For example, in the case of incorporating legumes into rotations, there is a commercial product to be marketed (i.e., hay or peas) and

[23] It should be noted that this scenario only results in positive net benefits with a 25% protection rate for riparian area (i.e., only restoring 25% of the potential riparian area) and a 10% improvement in weight gain.

there are agronomic benefits for subsequent crops in the form of fixed nitrogen and yield effects. However, from a producer perspective it is likely the case that the ES production effects are secondary to the direct benefits associated with these practices and that the BMPs would not be considered for adoption without those direct benefits.

So what are the policy implications of these study results? As noted in the discussion of the economic problem for this paper, Pannell's (2008) policy analysis framework is one way to identify appropriate policy instruments to encourage adoption of land use or management practice changes that increase ES production. As per the discussion in Pannell (2008), the choice of policy response should be based on estimates of both public and private benefits associated with specific land use or management practice changes.

The results discussed in the previous section of this paper represent estimates of net private benefits associated with BMPs that enhance ES production within agriculture. As such, it is hypothesized that these practices provide positive public net benefits. This information can be used as the basis for an initial discussion about identifying potential instruments for policy makers to consider if interested in encouraging producers to adopt these types of changes or practices. Figure 6 illustrates how the practices may be "placed" within the Pannell policy framework.

All three rectangles in Figure 6 are in the top half of the graph, which is consistent with providing positive net public benefits. They are, however, distributed (horizontally) through both the left and right upper quadrants within the graph. As per the earlier discussion, maintaining or protecting natural areas (blue rectangle in Figure 6) tends to result in negative net private benefits (i.e., net costs to adopting producers). Conversely, evidence from the four reviewed studies suggests that some crop management BMPs can result in positive net private benefits (i.e., red rectangle in Figure 6). Finally, pasture management BMPs (brown rectangle in Figure 6) tend to provide relatively small positive or negative private benefits.

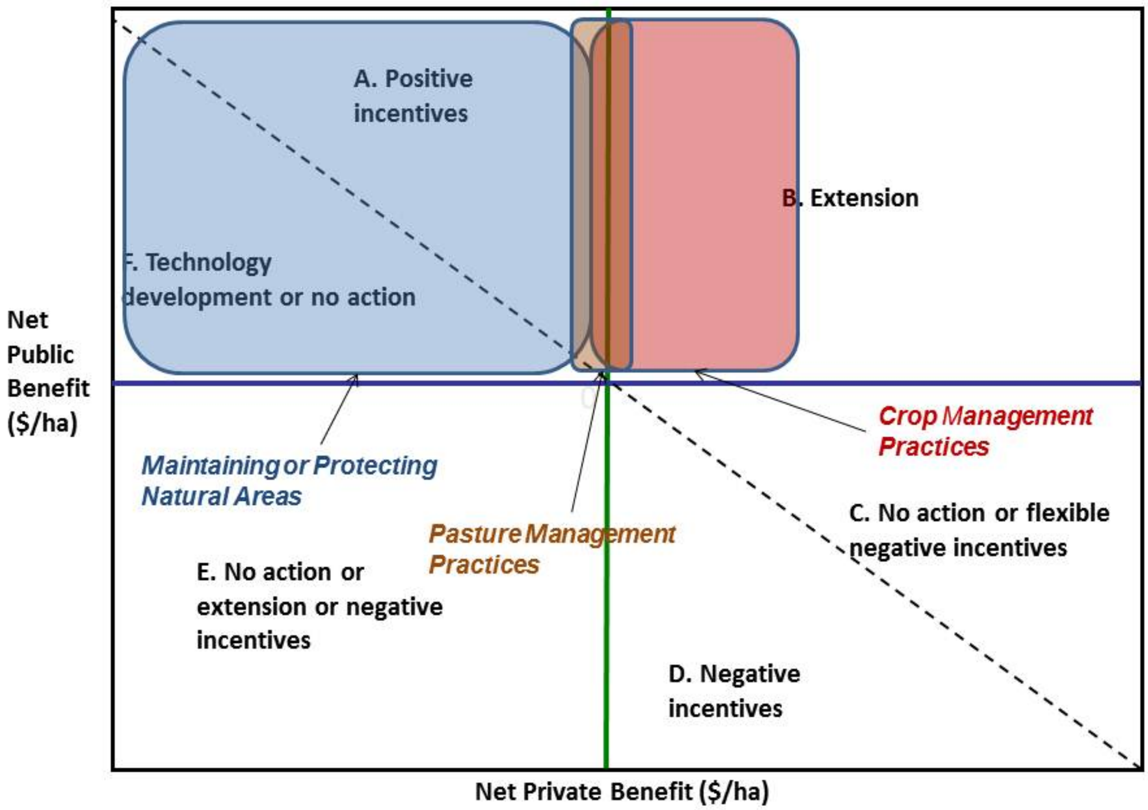
The four studies reviewed here focus on direct farm level financial impacts of adopting the various land use or management practice changes and do not attempt to quantify the effects on ES production or the value of any additional ES generated from adoption. As a result, it is not clear where these practices should be placed on the vertical scale within the Pannell framework other than being in the upper half (i.e., positive public benefits). This is indicated in Figure 6 by having the vertical extent of each rectangle (representing net public benefits) as the entire upper half of the policy framework graph.

The lack of information about public benefits associated with these practices/changes limits the ability to provide guidance to policy makers. For example, establishment of shelterbelts is shown by Trautman (2012) to result in significant negative net private benefits; as shown in Table 3, anywhere from $181 to 411 per hectare per year. This places the land use change clearly in the upper left hand quadrant of the Pannell policy framework (Figure 6). However, according to Pannell (2008) the appropriate policy response depends on the relative public benefits generated from this change. If the benefits are greater, in absolute terms, than the negative private benefits, the optimal policy response is to provide positive incentives to producers in order to encourage adoption of the change. This might entail the use of instruments such as subsidies, cost share programs, or perhaps market based instruments such as conservation auctions. However, if the public benefits from shelterbelts are more modest (i.e., less than the private costs of establishing them), the optimal policy response may be to do nothing; that is, recognize that it is not worthwhile to allocate public funds to support programs to encourage producers to plant trees. Instead, policy makers may wish to seek opportunities where public effort and funds would be more "efficiently" allocated (i.e., to support higher value ES production).

This dilemma is not true for all BMPs examined in these studies. An exception is the rectangle for crop management practices. For some of the BMPs within this category, both public and private benefits are positive. This suggests that an appropriate policy instrument is "extension". Providing information to producers about the potential direct benefits of adopting these practices is all that should be necessary. The same is true for some of the pasture management practices (i.e., part of that rectangle also overlaps with the upper right hand quadrant in Figure 6).

While there is a lack of information regarding public values associated with many of the BMPs examined in these studies, ongoing research by biophysical scientists (i.e., impact of BMPs on ES production) and environmental economists (i.e., value of ES) will be useful in filling the gaps. Wetland drainage is used here to illustrate how research results in those areas can be combined with the types of farm level analyses reviewed in this paper to inform policy development. In particular, Cortus' (2005) wetland drainage results are used as an example, as estimates of wetland values do exist in the literature.[24]

Figure 6: Pannell Policy Framework - Placement of Study Land Use/Management Practice Changes



As discussed earlier, Cortus' analysis indicates that annual net private benefits associated with wetland retention for the representative Saskatchewan cropping operation range from -$28 to -$41 per hectare of wetland. This places the land use change in the left hand half of the Pannell policy framework diagram. The implication is that the producer would otherwise have drained the wetlands, so the change is not doing so (i.e., retention).

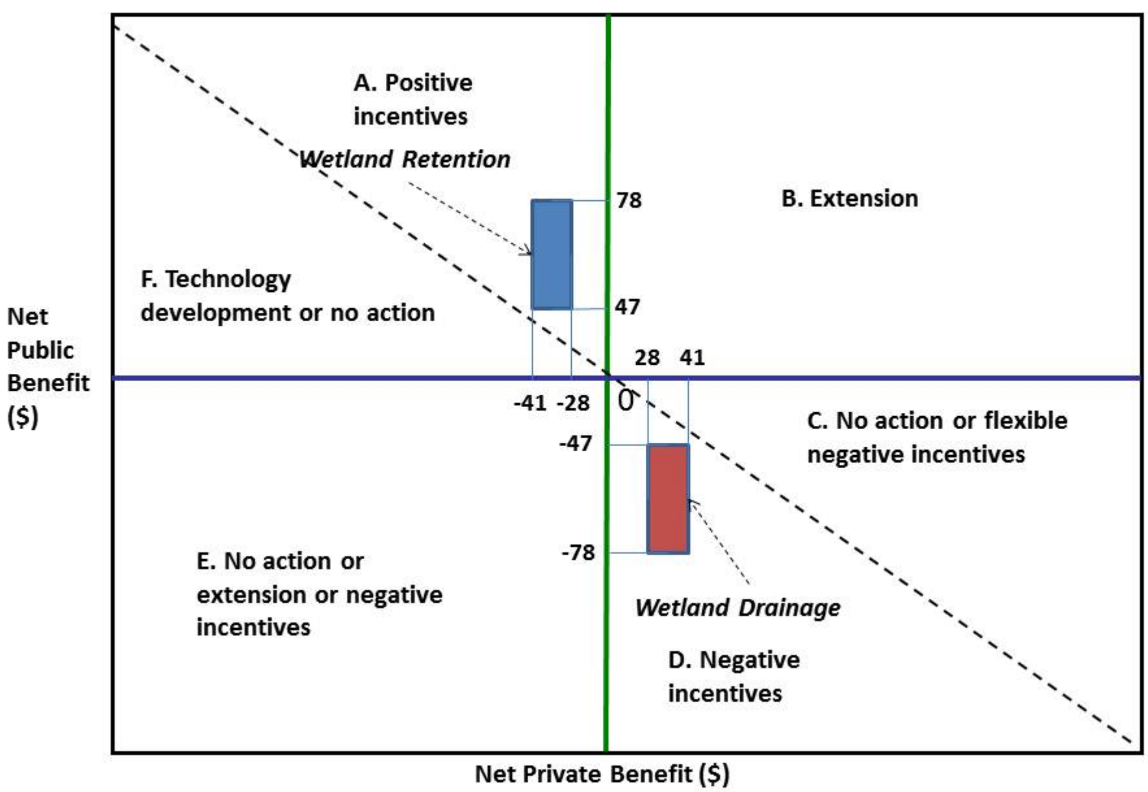
So what is the net public benefit associated with retaining a hectare of wetland? A number of wetland valuation studies have been completed, resulting in a wide range of values.[25] For the purposes of this example, estimates of wetland values from Belcher et al (2001), as reported in

[24] The example provided here is largely based on analysis and discussion in Cortus et al (2011).
[25] A summary of wetland valuation studies is provided by Brander et al (2006).

Olewiler (2004), are used. The Belcher et al study is not done in exactly the same location (i.e., watershed) as the Cortus analysis, but it is nearby. Belcher et al use a combination of estimation methods to develop a value for wetlands in the Upper Assiniboine River Basin, which is adjacent to Cortus' Emerald (SK) study region. Their "best" estimate of annual public benefits from retaining wetlands is approximately $78 per hectare. However, a significant portion of that estimate (i.e., approximately $23, the single largest component) is the value of carbon sequestration which assumes a given value of carbon. As discussed by Cortus et al (2011), there is uncertainty surrounding the likely value of carbon. Using the Belcher et al "best" estimate as the upper bound on the wetland value, a lower bound is estimated using a zero value of carbon. The resulting estimate of annual wetland value is $47 per hectare.

Figure 7 provides the resulting range of combinations of net public and private benefits for wetland retention. This is represented by the blue rectangle in the upper left hand quadrant of the Pannell policy framework diagram. Further, the rectangle lies entirely above the 45° diagonal, meaning that net public benefits from wetland retention are greater in absolute

Figure 7: Pannell Policy Framework Wetland Drainage/Retention



terms than the negative net private benefits. This implies that positive incentives (e.g., subsidies, conservation easements) represent an appropriate policy response to encourage land use changes of this type.

Before leaving this example, it should also be noted that an alternative interpretation of the wetland drainage/retention scenario may be defined within the policy framework. In particular, it is possible to consider the land use change as being wetland drainage; that is, if an area is currently undrained, the change would be to undertake drainage activities. Under this scenario, there are positive net private benefits associated with the land use change; the producer gains additional productive area as a result of draining the wetland, along with benefits from reduced nuisance costs. Obviously, the net public benefits for the change are negative. This scenario is also represented in Figure 7, by the red rectangle in the bottom right hand quadrant of the diagram. Given the estimates of public and private net benefits here, negative incentives should be used to discourage the land use change (e.g., regulations, taxes). This represents a very different policy prescription from the previous interpretation/scenario.

Whether wetland retention or wetland drainage should be considered as the relevant land use change may depend on an assumption regarding property rights. Under the first scenario (wetland retention = land use change), property rights are implicitly being assigned to the producer. Thus, society should "pay" to encourage the producer not to undertake the change. Conversely, with the second scenario (wetland drainage = land use change), property rights implicitly reside with the public and so a "polluter pay" principle takes precedence (e.g,. tax) or policy makers may decide to implement a command and control type of policy (i.e., regulation) to prevent limit or prevent drainage.[26]

VI. Conclusions and Future Work

The research results summarized in this paper illustrate the range of potential private benefits (or costs) associated with agricultural land use changes and management practice changes that contribute positively to ES production. Generally speaking the types of changes and scenarios modeled in the four studies result in net costs for adopting producers. Further, a significant proportion of those costs are attributable to foregone opportunities for commercial agricultural production (i.e,. opportunity costs), although in some cases (e.g., protection of natural areas by

[26] This discussion conveniently ignores the politics of policy making; that is, it tends to be more politically feasible to use positive incentives to encourage behavioural change, than to use negative incentives.

livestock producers) there are also significant implementation costs. The magnitude of these costs varies by size of operation, type of operation and also spatially (i.e., location of operation).

Despite the lack of information about net public benefits for these use and management practice changes, some general comments about policy may be made. In particular, it is likely that some type(s) of incentives (positive or negative, given the discussion for the wetland drainage example) are required to encourage producers to adopt BMPs such as buffer strips, shelter belts, wetland retention or restoration, etc. Alternatively, it may be the case in some instances that public benefits associated with the ES generated from these changes are sufficiently small that the optimal policy response is “no action”; that is, simply recognize that these types of changes are not going to occur and focus policy intervention in areas that appear to be more promising.

Results for some of the land use and management practice changes do suggest, however, that adoption of certain BMPs may occur without significant incentives being required. In particular, some crop rotation and pasture management related BMPs appear to have positive net private benefits. Combined with positive net public benefits, extension or information should be sufficient to encourage at least some producers to undertake these types of changes.

Finally, the results summarized here also highlight the need for additional research to be done. An obvious gap, noted earlier, exists in terms of quantifying public benefits associated with the types of changes modeled in the four studies reviewed in this paper. In order to develop estimates of public benefits, information is needed both for the quantity of ES generated by the changes and the value of those ES. This implies the need for both biophysical research (e.g., such as has been done through AAFC’s WEBs research program) and environmental economics research, as the values associated with many ES (e.g., biodiversity, aesthetics) are non-market in nature.

Related to this is the need for “scaling up” research to be undertaken. While adoption of practices such as buffer strips occurs at the farm level, the effects of these practices are often only measurable at a more aggregated level (e.g., watershed). This type of research often necessitates linking of farm level economic models with higher level biophysical models that relate factors such as soil type, topography and management practices to resulting environmental attributes (e.g., water quality) for a watershed.[27] There has only been a limited amount of this type of research done in a Canadian setting (e.g., De Laporte et al 2010; Dissart et al 2000; Yang et al 2007).

Finally, there is also a need to expand the set of estimates for farm level impacts of the types of changes reviewed here. As mentioned earlier, net costs and benefits from adoption of these types of practices can vary by farm type, size and location. Only a limited number of studies have been undertaken in a Western Canadian setting to estimate these values. However, if policy initiatives involving incentives of one type or another are undertaken to encourage producers to adopt environmental stewardship practices, it will likely be on a regional or national level. Additional research could be undertaken to expand the set of representative farms or to expand the set of land use and management practice changes considered in analyses. Another avenue of research may be to examine the feasibility of taking current estimates of private benefits or costs and transferring them to other sites. There exists a body of literature within the environmental economics discipline that examines transferability of willingness to pay (WTP) estimates for non-market values and it may be possible to adapt these methods for use in transferring costs and benefits of BMP adoption. Work is ongoing in both of these areas in order to try and extend the current set of estimates for net private benefits of BMP adoption in order to better inform environmental and agricultural policymaking.

[27] An example of this type of model is the Soil Water Assessment Tool (SWAT; Arnold et al 1998).

VII. References

Alberta Agriculture, Food and Rural Development (AAFRD). 2006. Beneficial Management Practices: Environmental Manual for Alberta Farmsteads. Edmonton, AB. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/ba3468a2a8681f69872569d60073fde1/610cf5ba8a00ff0f8725723d006f49ba/$FILE/090-1.pdf (accessed August 18, 2012).

Arnold, J.G., R. Srinivasan, R.S. Muttiah and J.R. Williams. 1998. Large Area Hydrologic Modeling and Assessment Part I: Model Development. *Journal of the American Water Resources Association*. 34:73-9.

Belcher, K., C. K. Edwards and B. Gray. 2001. Ecological Fiscal Reform and Agricultural Landscapes, Analysis of Economic Instruments: Conservation Cover Incentive Program. Background Paper, Ecological Fiscal Reform, National Roundtable on the Economy and Environment, Ottawa, ON.

Boyd, J. and S. Banzhaf. 2007. What are Ecosystem Services? The Need for Standardized Environmental Accounting Units. *Ecological Economics.* 63:616-26.

Boxall, P., M. Weber, O. Perger, M. Cutlac and A. Samarawickrema. 2008. Results from the Farm Behaviour Component of the Integrated Economic-Hydrologic Model for the Watershed Evaluation of Beneficial Management Practices Program. Project Report 08-01. Edmonton: University of Alberta, Department of Rural Economy. November.

Brander, L.M., I. Bräuer, H. Gerdes, A. Ghermandi, O. Kuik, A. Markandya, S. Navrud P.A.L.D. Nunes, M. Schaafsma, H. Vos and A. Wagtendonk. 2012. Usint Meta-Analysis and GIS for Value Transfer and Scaling Up: Valuing Climate Change Induced Losses of European Wetlands. *Environmental and Resource Economics*. 51:395-413.

Brander, L. M., R. J. G. M. Florax and J. E. Vermaat. 2006. The Empirics of Wetland Valuation: A Comprehensive Summary and a Meta-analysis of the Literature. *Environmental and Resource Economics.* 33:223–50.

Brethour, C., B. Sparling, B. Cortus, M. Klimas and T. Moore. 2007. An Economic Evaluation of Beneficial Management Practices for Crop Nutrients in Canadian Agriculture. Final Report to Crop Nutrients Council. George Morris Centre, Guelph, ON. January.

Cook, B.R. and C.J. Spray. 2012. Ecosystem Services and Integrated Water Resource Management: Different Paths to the Same End? *Journal of Environmental Management.* 109:93-100.

Cortus, B. 2005. The Economics of Wetland Drainage: A Case Study in Canada's Prairie Pothole Region. M.Sc. thesis. Edmonton: University of Alberta.

Cortus, B.G., S.R. Jeffrey, J.R. Unterschultz and P.C. Boxall. 2011. The Economics of Wetland Drainage and Retention in Saskatchewan. *Canadian Journal of Agricultural Economics*. 59:109-26.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton and M. van den Belt. 1997. The Value of the World's Ecosystem Services and Natural Capital. *Nature.* 387:253-60.

De Laporte, A., A. Weersink and W. Yang. 2010. Ecological Goals and Wetland Preservation Choice. *Canadian Journal of Agricultural Economics*. 58:131-50.

Dissart, J-C., L. Baker and P.J. Thomassin. 2000. The Economics of Erosion and Sustainable Practices: The Case of Saint-Esprit Watershed. *Canadian Journal of Agricultural Economics*. 48:103-22.

Dollevoet, B. 2010. Farm Wealth Implications of Ecological Goods and Services Practices and Policies. M.Sc. thesis. Edmonton: University of Alberta.

Fisher, B., R.K. Turner and P. Morling. 2009. Defining and Classifying Ecosystem Services for Decision Making. *Ecological Economics*. 68:643-53.

Gascoigne, W.R., D. Hoag, L. Koontz, B.A. Tangen, T.L. Shaffer and R.A. Gleason. 2011. Valuing Ecosystem and Economic Services Across Land-Use Scenarios in the Prairie Pothole Region of the Dakotas, USA. *Ecological Economics*. 70:1715-25.

Gómez-Baggethun, E., R. de Groot, P.L. Lomas and C. Montes. 2010. The History of Ecosystem Services in Economic Theory and Practice: From Early Notions to Markets and Payment Schemes. *Ecological Economics*. 69:1209-18.

Johnson, K.A., S. Polasky, E. Nelson and D. Pennington. 2012. Uncertainty in Ecosystem Services Valuation and Implications for Assessing Land Use Tradeoffs: An Agricultural Case Study in the Minnesota River Basin. *Ecological Economics*. 79:71-9.

Koeckhoven, S. 2008. Economics of Agricultural Best Management Practices in the Lower Little Bow Watershed. M.Sc. thesis. Edmonton: University of Alberta.

Ma, S. and S.M. Swinton. 2011. Valuation of Ecosystem Services from Rural Landscapes Using Agricultural Land Practices. *Ecological Economics*. 70:1649-59.

Marra, M. C. and B. W. Schurle. 1994. Kansas Wheat Yield Risk Measures and Aggregation: A Meta-Analysis Approach. *Journal of Agricultural and Resource Economics.* 19:69-77.

McMaster, G.S. and W.W. Wilhelm. 1997. Growing Degree-Days: One Equation, Two Interpretations. *Agricultural and Forest Meteorology*. 87:291-300.

Norgaard, R.B. 2010. Ecosystem Services: From Eye-Opening Metaphor to Complexity Blinder. *Ecological Economics*. 69:1219-27.

Olewiler, N. 2004. *The Value of Natural Capital in Settled Areas of Canada*. Ducks Unlimited Canada and the Nature Conservancy of Canada. http://www.ducks.ca/aboutduc/news/archives/pdf/ncapital.pdf (accessed February 28, 2010). Updated link: http://www.sfu.ca/mpp-old/04research/pdfs/natural_capital.pdf

Palisade Corporation. 2010. *Guide to Using @RISK, Version 5.7*. Ithaca, New York: Palisade Corporation.

Pannell, D. 2008. Public Benefits, Private Benefits, and Policy Mechanism Choice for Land-Use Change for Environmental Benefits. *Land Economics*. 84:225-40.

Sagoff, M. 2011. The Quantification and Valuation of Ecosystem Services. *Ecological Economics.* 70:497-502.

Trautman, D. 2012. The Economics of Beneficial Management Practices Adoption on Representative Alberta Crop Farms. M.Sc. thesis. Edmonton: University of Alberta.

Yang, W., A. Rousseau and P. Boxall. 2007. An Integrated Economic-Hydrologic Modeling Framework for the Watershed Evaluation of Beneficial Management Practices. Journal of Soil and Water Conservation. 62:423-32.

Zaimes, G.N. and R.C. Schultz. 2012. Assessing Riparian Conservation Land Management Practice Impacts on Gully Erosion in Iowa. *Environmental Management*. 49:1009-21.

Zhang, X. and M. Zhang. 2011. Modeling Effectiveness of Agricultural BMPs to Reduce Sediment Load and Organophosphate Pesticides in Surface Runoff. *Science of the Total Environment*. 409:1949-58.

Appendix: Pannell Policy Framework

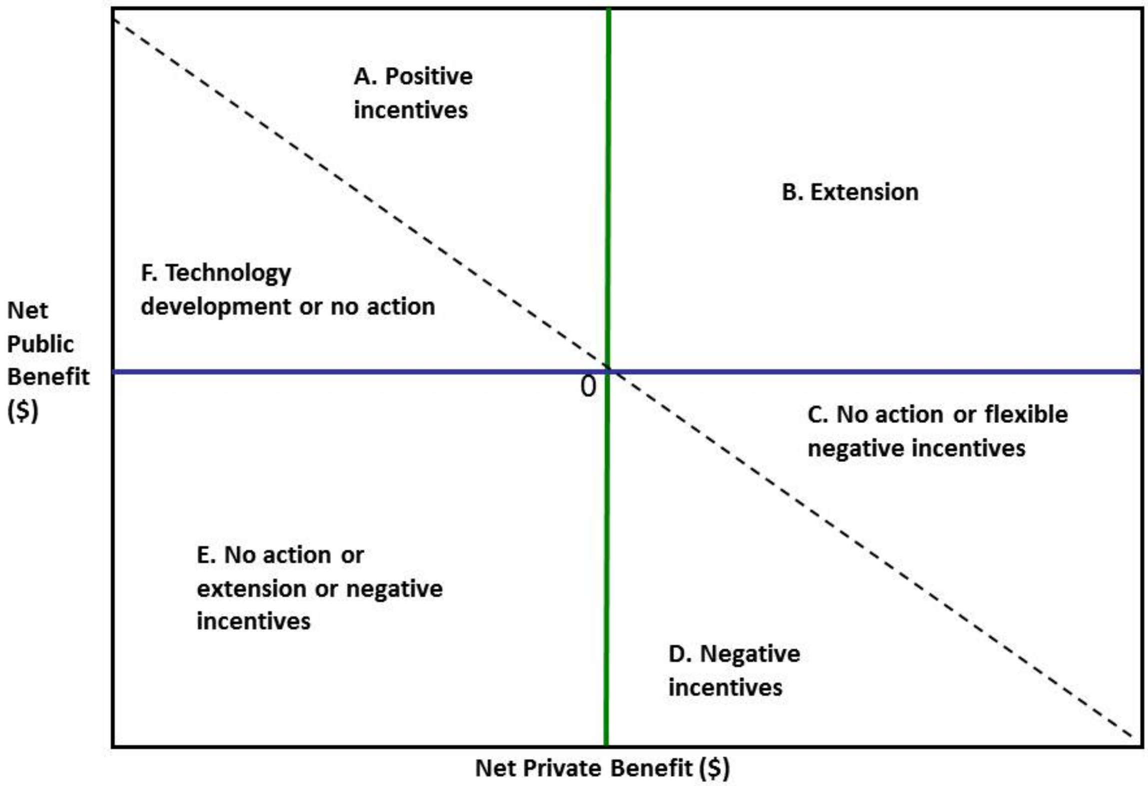
In considering options for policy intended to encourage increased production of ecosystem services within the agricultural sector, a number of different instruments are available to policy makers. These include regulations, taxes, subsidies, market based instruments such as conservation auctions or easements, etc. Often, policy decisions are made based on previous experience (e.g., what has worked in the past), taxpayer cost considerations and political expediency. However, from the perspective of economics, it would seem to make sense to think about policy options and policy making in terms of costs and benefits. What are the costs associated with implementing a particular policy instrument versus the resulting benefits? What type of instrument will be most effective in terms of generating benefits and most efficient in terms of value obtained from public funds invested?

Pannell (2008) incorporates relevant economic considerations into a policy analysis framework that may be used to consider BMP scenarios examined in this paper. Pannell's framework uses the relative magnitude of net public and private benefits resulting from a land use change as the basis for identifying an appropriate policy instrument. As per Pannell's (2008) discussion, "private benefits" are the benefits (net of costs) that accrue to the land owner or decision maker as a result of making a land use change. "Public benefits" are the benefits (again net of costs) that accrue to everyone else, and may be considered as a proxy for the value to society of the change. For the purposes of the discussion in this paper, the framework is "extended" to include consideration of not only land use changes (e.g., wetland drainage or restoration) but also changes in agricultural management practices (e.g., implementation of zero tillage or nutrient management planning).

The result is a mapping of alternative policy options to combinations of relative public and private benefits. This is illustrated in Figure A1. For example, if a land use change generates both positive private and public benefits (i.e., the upper right hand quadrant, or area A, in Figure A1), Pannell's framework indicates that the appropriate policy instrument is extension or information. In other words, all that should be necessary for private landowners to make the land use change is awareness of the practice and the nature of the private benefits associated with that change. Alternatively, if a land use change results in negative public benefits but positive private benefits, this places it in the lower right hand quadrant (i.e., either area C or D) in Figure A1. In this case the appropriate policy response is dependent on the relative absolute magnitude of the public and private benefits associated with the change. If the public "cost" is greater, in absolute terms, than the positive private benefit (i.e., the public is "harmed" from the change to a greater extent than the landowner benefits), negative incentives (e.g., taxes or regulations) should be used to discourage producers from undertaking the land use change. This is represented by area D in Figure A1 (i.e., below the 45° diagonal). Conversely, if the net benefits to the landowner outweigh the public costs from the land use change (i.e., area C in Figure A1), then the appropriate response is either to accept that the land use change will occur (i.e., no action) or use flexible negative incentives (e.g., taxes). Corresponding interpretations may be made for the other quadrants and subquadrants in Figure A1. Pannell (2008) goes on to expand and extend the policy analysis framework to incorporate factors such as transaction and learning costs, adoption lags and benefit-cost considerations. However, for the purposes of the discussion in this paper, the simplified version presented here is used.

In order to make empirical use of a framework such as the one developed by Pannell (2008) to inform agricultural/environmental policymaking, estimates of both public and private benefits for alternative land use practices (i.e., BMPs) are required. For example, suppose that a government agency is interested in encouraging agricultural producers to establish buffer strips around wetlands on their properties. Buffer strips have been shown to improve filtration of sediments and chemicals, thereby improving the quality of water in wetlands (e.g., Zaimes and Schultz 2012; Zhang and Zhang 2011). This suggests that buffer strips increase ecosystem service production (e.g., recreation, drinking water quality), resulting in positive net public benefits. The private benefits associated with this land use change would likely be negative, due to the opportunity cost of removing land from production to create the buffer strips. This would place the land use change in the upper left hand quadrant of the Pannell framework (i.e., Figure A2). However, the appropriate policy response depends on the relative magnitude of those benefits. If public benefits outweigh negative private benefits (i.e., area A in Figure A2), positive incentives to encourage adoption of the BMP are warranted. Conversely, if the private costs outweigh the public benefits from buffer strips, then perhaps taking no action is appropriate; that is, accept that buffer strips will not be widely adopted and focus attention on other management practices.

Figure A1: Pannell Framework for Agricultural/Environmental Policy Analysis[a]



Source: Pannell (2008)

[a] Net Private Benefit refers to the benefits (positive or negative) received by the private individual making a change in land use or management practice; Net Public Benefit refers to the benefits (positive or negative) received by society as a result of the change.

Figure A2: Pannell Policy Framework – Buffer Strip BMP Example