Tradeoffs Between Environmental Quality and Economic Returns from Agriculture: A Case Study of the Lower Little Bow Watershed, Alberta

by

Marcus Becker

A thesis submitted in partial fulfillment of the requirements for the degree of

Master of Science

in

Agricultural and Resource Economics

Department of Resource Economics and Environmental Sociology
University of Alberta

Abstract

The aim of this study is to evaluate both the costs and benefits of implementing changes to agricultural practices, with the goal of identifying cost effective means of achieving environmental targets and providing supplementary information to policymakers. A suite of agricultural land use scenarios are developed to assess tradeoffs between economic returns from agriculture and indicators of environmental quality in the Lower Little Bow (LLB) watershed of southern Alberta. These scenarios, 11 in total, feature a range of beneficial management practices (BMPs) designed to improve select environmental criteria. Building upon research done previously on BMP adoption in the LLB watershed, the BMPs featured include alternative crop rotations, manure management strategies, and various land use conversions. The environmental factors of interest in this analysis are water quality and soil carbon levels, as represented by nutrient loads and changes to carbon sequestration rates, respectively. Results indicate that implementing BMPs in the LLB watershed to achieve environmental benefits will have negative impacts on economic performance and that policy changes may be necessary to induce land use changes. The results also demonstrate that certain land uses in the LLB watershed can achieve greater environmental benefits for less cost, which has important policy implications in an agricultural context.

Acknowledgements

First, I would like to thank my supervisor, Dr. Scott Jeffrey, for his insightful guidance, patience, and confidence in me throughout the duration of this project. He managed to see past our differences in hockey team preference and provide me with a constant source of encouragement throughout this endeavor, for which I am tremendously grateful.

I would also like to thank Dr. Jim Miller, research scientist with Agriculture and Agri-Food Canada's Lethbridge Research Station, for his support and helpful suggestions throughout the project.

My thanks are also extended to Dr. Peter Boxall, member of my advisory committee, and Dr. David Chanasyk, external examining committee member, for their input and thoughtful comments.

I also gratefully acknowledge the funding that was provided for this research project by Agriculture and Agri-Food Canada (AAFC) through the Watershed Evaluation of Beneficial Management Practices (WEBs) research program. Additionally, I am very appreciative of the support of the Department of Resource Economics and Environmental Sociology (REES), the Faculty of Agriculture, Life, and Environmental Sciences (ALES), and the University of the Alberta.

To my friends, roommates, fellow students, and work colleagues: thank you. It's been the best of times.

Finally, to Mom and Dad. I am forever indebted to you both for your love and support throughout the entirety of my university education. Thank you. This could not have been possible without you.

Table of Contents

List of Tables

List of Figures

Chapter 1: Introduction

1.1 Background

Management of the complex interactions between agricultural production and the surrounding environment is a current focal point of research, policy, and debate. While the responsibility for this management has historically been left to the private producer or land-owner, the residual impacts of these management decisions are often felt at the societal, or public, level. The quality and health of the surrounding environment is a significant way in which these impacts are experienced by society.

Beyond simply producing marketable products such as food, agricultural land can provide society with a multitude of other goods and services. These include the continued maintenance of water quality, the storage of carbon in soil, the provision of wildlife habitat and recreational opportunities, as well as the continued cycling of nutrients throughout the ecosystem (Dale and Polasky, 2007). Along with many others, these beneficial impacts are termed ecosystem services (ES). Boyd and Banzhaf (2007) define ES as "components of nature, directly enjoyed, consumed, or used to yield human well-being" (p.619). On land used for agriculture, the preservation, enhancement, or degradation of these ES can be greatly impacted by management practices and land use decisions. The value of these ES, coupled with increasing demand for food and other agricultural products from pressures of growing populations and continued economic growth, make the efficient and sustainable use of this land an essential goal. As such, both farmers and society have an important responsibility to manage these agri-environmental interactions with care.

One way that positive environmental outcomes can be achieved in conjunction with agricultural production is via the implementation of Beneficial Management Practices (BMPs). BMPs are practices that either enhance the provision of ES and/or mitigate potential adverse impacts to existing ES attributable to agricultural activities. Three general categories of BMPs include the reduction of inputs, control of runoff or erosion, and the use of barriers or buffers to contain or intercept contaminants from reaching the surrounding environment (AAFC, 2000). For instance, improved nutrient management that reduces the amount of fertilizer or manure applied on or lost from cropland can help ensure that water quality is not adversely affected via leaching or surface runoff of excess nutrients. The use of a buffer strip around a nearby water body can help safeguard against the movement of these nutrients into water resources. Since preservation and enhancement of environmental quality is often a significant focus of policy-makers, provision of ES through BMPs represents an important channel of empirical investigation to help improve policy formation.

Conversely, certain practices can facilitate the depletion of ES, such as the continued cultivation of marginally productive lands, the drainage of wetlands, and the intensive use of chemicals (DUC, 2006). However, many of these practices contribute positively to the economic viability of a farm operation, creating a tradeoff between the benefits derived from ES and the private returns from agricultural production. Understanding and quantifying these tradeoffs is essential to development and formation of policy targeted at improving economic and environmental outcomes. Studying the implementation of BMPs represents an effective way to evaluate these tradeoffs since both the farm economy and the quality of surrounding environment have the potential to be impacted.

The Lower Little Bow (LLB) watershed is an agriculturally-intensive region of southern Alberta, Canada, where the interaction between agricultural activity and the surrounding environment requires significant management and attention. Within immediate proximity of the LLB watershed are a number of intensive livestock operations, which create significant waste management issues for the region. Additionally, intensive annual cropping is common across the landscape, leading to high use of chemical inputs, manure, fossil fuels, and water resources. Both the quality and quantity of water supplies have been identified as areas of concern for the region (OWC, 2010). In the LLB watershed, the Little Bow River is vulnerable to both the runoff and subsurface flow of nutrients from nearby agricultural fields. As such, the LLB watershed is the study site for this analysis, which will focus on the tradeoffs between economic returns from agriculture and environmental quality within the watershed.

1.2 Economic Problem

In economics, producers are assumed to be rational agents who seek to maximize their returns, or profit, given a set of constraints. Examples of constraints in an agricultural production context include limits on land, labour, and/or financial capital. Given this, producers will select a set of land uses and associated management practices that optimize their objective. Often, however, this does not include the provision of a socially optimal level of ES, since these benefits accrue to society as a whole and not directly to the producer. In such cases ES provision is lower than what may be optimal from society's point of view.

Voluntary adoption of BMPs by producers that would enhance ES provision is unlikely to happen in circumstances where the private costs associated with doing so outweigh the private benefits. For example, abstaining from draining wetlands on agricultural fields decreases the land base available for cultivation and consequently reduces potential revenue (e.g., Cortus, 2005). While this decision enhances the ES provided to society by the wetland, the private producer bears the full cost. Other BMPs may involve increased ongoing management costs (e.g., Koeckhoven, 2008). Conversely, producers have an incentive to adopt practices that increase returns or maximize their objectives. This is the case regardless of whether or not the practice or land use in question is considered a BMP. For instance, practices that erode the stock and provision of ES on agricultural land may be adopted if economic returns are improved (e.g., draining a wetland). In some cases, these actions impose costs on society; increased water treatment costs may be a consequence of draining nearby wetlands, for example. Conversely, a BMP may benefit both the producer and society concurrently if the change in practice increases both private and public benefits.

Pannell (2008) developed a policy framework for deciding between different actions to be undertaken by policy makers in the context of land use and management of private land. Pannell (2008) suggested that the appropriate policy response depends on the sign (either positive or negative) and magnitude of both private (i.e., producer) and public (i.e., societal) benefits. For instance, in cases where the public benefits of BMP adoption are positive and outweigh the costs to the private producer (i.e., private benefits are negative), the appropriate policy mechanism according to Pannell's framework is a positive incentive, such as a subsidy. This would allow for compensation to the producer, thereby encouraging adoption, with increased public benefits as a result. Alternatively, when both private and public net benefits of a BMP are positive, extension and education is the appropriate response. This is because increased private benefits should provide an incentive to a producer, negating the need for financial compensation.

The difficulty in applying this framework to BMP adoption in agricultural production is the uncertainty regarding both private and public impacts. Previous research has been undertaken to quantify the private impacts of BMP adoption for agricultural producers (e.g., Trautman, 2012; Xie, 2014). However, less is known about the public benefit side. This is primarily due to a lack of markets, meaning that the true value is not revealed by market prices (unlike the case for private benefits). Without this monetary value it is challenging for policy makers to make informed decisions regarding policy for BMP adoption. This research seeks to address this economic problem by quantifying both public and private impacts of BMP adoption in a Canadian agricultural context.

1.3 Research Objectives

The purpose of this study is to assess the private economic impacts of BMP adoption and land use change in the LLB watershed of southern Alberta. Changes in economic returns from agriculture will be assessed for a suite of BMPs identified as feasible for the study area and able to improve upon current environmental concerns. In conjunction with private evaluation, resulting changes to specific parameters of environmental quality will also be evaluated in order to quantify certain elements of public impacts. The aim of the study is to have quantifiable estimates of both public and private impacts. The results will assist policy-makers by highlighting the various tradeoffs that exist between economic profitability and environmental quality, thus helping to improve the policy selection and decision making process.

1.4 Organization of the Thesis

This thesis is organized into seven chapters. Following this introductory chapter, Chapter 2 introduces in more detail the environmental quality outcomes of interest in this study, including water quality and storage of organic carbon in soil, and how agriculture impacts these outcomes. This chapter also provides an overview of the BMPs considered for inclusion in this study, including both the biophysical and economic impacts of these management practices.

Chapter 3 introduces the study area as well as agricultural activity more broadly in southern Alberta. Data from the most recent Census of Agriculture (as well as other sources) are presented, as well results from a survey of producers in the LLB watershed. Past land uses, typical crop mix, and historical soil test results are discussed. A summary of pertinent environmental issues in the study area are also provided.

Chapter 4 provides a discussion of the methodological approach employed in this study. Specifically, alternative modeling techniques are introduced, including capital budgeting techniques and simulation models. The nutrient budget balance approach is introduced as a method to model the risk posed by excess nutrients from agricultural production pose to water quality. Lastly, different approaches to quantify changes in SOC storage from changes to agricultural land management practices are discussed.

Chapter 5 introduces the baseline and BMP scenarios used in the analysis, and details the specific methods used to quantify economic and biophysical impacts. Chapter 6 presents the results from both baseline scenarios and the eleven BMP scenarios of land use in the LLB watershed. Tradeoff curves between environmental quality metrics and private economic returns are developed, and non-market valuation approaches are used to monetize the public benefits of environmental quality improvements.

The final chapter of the thesis (Chapter 7) draws conclusions from the findings of the study and discusses implications for future environmental and agricultural management in the LLB watershed. The chapter concludes with a discussion of the study limitations as well as important areas of future research to refine our understanding of the tradeoffs between agricultural profitability and environmental quality.

Chapter 2: Agricultural Beneficial Management Practices and the Environment

Agricultural activity interacts with the surrounding environment in significant and dynamic ways. Among the elements of the environment at the forefront of this interaction are soil health, water quality, nutrient cycling, and biodiversity (Dale and Polasky, 2007). Accounting for and managing this interaction in a sustainable and responsible manner is fundamentally important to the balance of economic, environmental, and social objectives. Further expansion and intensification of agricultural production will likely increase pressures on the surrounding environment, necessitating the need for improved environmental management.

The implementation and adoption of agricultural BMPs is one way in which these issues can be addressed. BMPs are alternative farming practices that can be employed by producers to enhance the quality of the surrounding environment and the provision of public ecological goods and services. BMPs are also often used to minimize the potential harm that farming can sometimes impose on natural systems. A BMP may be defined as a management practice that "ensures the long-term health and sustainability of land related resources used for agricultural production, positively impacts the long-term economic and environmental viability of the agricultural industry, and minimizes the negative impacts and risk to the environment" (Boxall et al., 2008, p.5). Agriculture and Agri-Food Canada (AAFC) categorizes BMPs into three general types: those that control runoff and erosion, those that reduce inputs, and those that act as barriers and buffers (AAFC, 2014). The Canadian Farm Stewardship Program outlines several categories of activity specific BMPs, including livestock site management, manure management, land management, irrigation management, precision farming, and agricultural waste (SAFRR, 2016).

Certain agricultural BMPs can positively impact farm profitability through environmental improvements which enhance productivity or reduce costs. For instance, BMPs that improve soil quality and therefore crop yield potential may increase economic returns if the cost of implementation is less than the benefits received. However, other

BMPs involve private costs to producers and do not increase profitability. Rather, the benefits of BMP implementation accrue to society. An example of this would be the biodiversity, nutrient cycling, and carbon storage benefits provided by preserving a wetland on an area used for annual cropping. In these cases, as discussed in Chapter 1, public policy intervention may be necessary to ensure that these services are provided and environmental quality is preserved.

The adoption of BMPs by producers in the LLB watershed is one way in which positive environmental quality outcomes can be achieved. The following sections describe both the environmental quality outcomes and BMPs of interest in this study.

2.1 Environmental Quality Outcomes of Interest

The focus of this study is on water quality and the storage of carbon in agricultural soil, and how they are impacted by agricultural production practices. Water quality has been identified as an area of concern due to the high intensity of agricultural operations in the study region and the frequency of impairment to the quality of surface water bodies (OWC, 2010). As a strategy to mitigate climate change, the carbon storage potential of agricultural soils has increasingly garnered attention among researchers and policy-makers (McConkey et al., 2014; Smith et al., 2001).

2.1.1 Water Quality and Nutrient Cycling

The transfer and loss of nutrients to the surrounding environment has become a prevailing issue in modern agriculture. The availability of chemical fertilizers at relatively low cost has greatly reduced the agronomic constraint of low nutrient availability in soil (Havlin et al., 2014). However, agricultural practices that involve use of these inputs (e.g., intensive annual cropping on nutrient deficient soils) are of primary environmental concern, specifically as they relate to water quality. When the import of these nutrients into the soil via fertilizer or manure exceeds the removal by plants, the surplus is retained in the agro-ecosystem.

A surplus of residual N can be transferred to the environment in a number of ways. Residual inorganic N can be lost to the atmosphere as nitrogen gas (N_2), nitric oxide (NO), or as nitrous oxide (N_20), the latter of which is a potent greenhouse gas (GHG). Alternatively, residual N can remain in the soil as nitrate (NO_3^-), nitrite (NO_2^-), or ammonium (NH_4^+) (Fetter, 1993). NO_3^-, in particular, is at risk of leaching into groundwater or being lost in runoff into surface water as it is highly mobile in aqueous environments, and can be easily transported into ground or surface water bodies (Eilers et al., 2010). Organic N, which is present in manure as amino acids and urea compounds, is generally more stable and less at risk of transport away from the soil profile. However, mineralization of organic N into more mobile inorganic forms (NO_3^-, NO_2^-, etc.) does occur naturally over time. The process of denitrification, where NO_3^- is transformed to N_2 gas, helps to naturally attenuate the accumulation of NO_3^- in groundwater and the soil profile. Occurring in conditions of restricted oxygen availability, denitrification is a bacterially-mediated process involving the reduction of an N oxide (e.g., NO_3^-) by either organic or inorganic electron donors, such as sulfur, iron, or organic carbon (McCallum et al., 2008).

Residual P can also be transferred and lost to surrounding water bodies. However, P is generally considered less mobile than inorganic forms of N, and therefore less at risk of leaching into groundwater. This immobility is due to the adsorptive capacity of calcareous soils, which are common in southern Alberta (Lutwick and Graveland, 1978). Therefore, surface flow (i.e., runoff) or soil erosion are the dominant P transport mechanisms, both of which are impacted by several factors including climate, agricultural practices, and soil type (Liu et al., 1997).

NO_3^- pollution of ground and surface water is problematic from both an environmental and health perspective. Methemoglobinaemia (blue-baby syndrome) is caused by elevated levels of NO_3^- in drinking water (Johnson et al., 1987). Other human health-related impacts of consuming NO_3^- polluted water include increased prevalence of carcinogenic nitrosatable compounds linked to gastrointestinal cancer risk, reproductive and developmental toxicity, and brain and urinary tract tumours (WHO, 2011; Health Canada, 2013). As such, the maximum allowable NO_3^- concentration for drinking water in Canada is set to 45mg per L, or 10mg per L when considered as NO_3^--N (Health Canada, 2013). Regarding NO_2^-, a maximum of 1mg per L is recommended.

High levels of NO_3^- and/or P in surface water bodies can contribute to excess algae growth and eutrophication (Eilers et al., 2010). Eutrophication can have detrimental impacts to fish and other organisms as oxygen levels become depleted in the presence of rapid plant and algae growth (ECCC, 2010). This excessive biologic production also leads to increased water temperatures, sedimentation, and can impede water flow and navigation. Cyanobacterial (blue-green algae) blooms are also common in eutrophic waters. These blooms increase the incidence of odour and taste problems with drinking water, as well as produce trichloromethane during the release of hepato- and neuro-toxins upon bacterial death. Alberta Environment and Park's Surface Water Quality Guidelines for the Protection of Aquatic Life stipulate that NO_3^--N and P levels should not exceed 10 mg and 0.05 mg per L, respectively (AEP, 1999).

Although traditionally thought of as a local or regional issue, the vastly increased supply of biologically available N and P created synthetically for agricultural production purposes has led to alteration of nutrient cycles at a global level (Vitousek et al., 1997; Galloway, 2008). Improved nutrient management is of critical importance to the ongoing sustainability of agricultural systems, and water resources in particular.

2.1.2 Storage of Soil Organic Carbon

Soils are an important carbon sink and will play an important role in development of strategies to mitigate future climate change (Smith et al., 2001). The sequestration of atmospheric CO_2 in agricultural soil represents a significant public benefit, as CO_2 is a potent GHG which contributes to the warming of the planet (IPCC, 2007). The cultivation of native grassland for the purposes of crop production releases the carbon stored in the soil into the atmosphere, and through this process the expansion of modern agriculture in the 20th century contributed significantly to the proliferation of GHGs in the atmosphere (Smith et al., 2001). However, these losses can be reversed with appropriate soil management. In their current state, agricultural soils of Canada have significant carbon sink capacity (Paustian et al., 1997). Certain land management changes to conventional cropping systems have been suggested as means to foster the storage of soil organic carbon (SOC), including the reduction or elimination of tillage practices, decreases in the use of summerfallow, or inclusion of more perennial vegetation.

Although there are certain private benefits to SOC accumulation, the bulk of the benefits accrue to society. For instance, in a study modeling the SOC storage and economic impacts of various cropping rotations in the Canadian prairies, Belcher et al (2003) found that the private, on-site benefits of each tonne of SOC ranged from $0.20 to

$2.10 per hectare per year. However, these values are dwarfed by literature estimates of public benefits of carbon sequestration, which are often greater than $100 per tonne (e.g., ECCC, 2016; Nordhaus, 2007).[1] Therefore, as the LLB watershed has a long history of cultivation, the enhancement of SOC storage capabilities is an environmental quality outcome of interest in this study.

2.2 Beneficial Management Practices of Interest

Several BMPs were identified for their potential to positively impact the environmental quality outcomes outlined in the previous section, as well as their feasibility of adoption in the study area. These BMPs include the introduction of alfalfa, legume green manures, and field peas to annual crop rotations, the management of livestock manure, and the conversion of annual cropland to permanent forage.

2.2.1 Introduction of Alfalfa

Alfalfa (*M. sativa*) is a perennial legume that is grown primarily for hay feedstock, silage, or grazing as a forage source for livestock. In Alberta, alfalfa is one of the most common hay crops as it is extremely productive when managed properly and a suitable variety is selected (AAF, 2005d). It also responds well to irrigation conditions, and generally yields a productive second cutting within a growing season (AAF, 2001). Irrigated alfalfa can produce between 4.5-5 tonnes of dry matter per acre annually when sufficient nutrients are received (AAF, 2005d).

Growing alfalfa can be a considered a BMP for a multitude of reasons, several of which are relevant to this study. First, as a legume, alfalfa plants have the ability to symbiotically fix nitrogen (N_2) from the atmosphere for purposes of plant growth. This feature is made possible due to the presence of *Rhizobium* bacteria in the root nodules of the crop.[2] The bacteria form a symbiotic relationship with the crop by making N available in a more useable (i.e., mineral amino acid) form (PSE, 2015). This biological fixation is the source of several potential environmental and economic benefits. Because N is taken

[1] A detailed discussion of these estimates of public benefits is provided in Chapter 6, section 6.5.1.
[2] Crop producers can augment the amount of bacteria present in the roots of legume crops via the practice of inoculation.

directly from the atmosphere, the need for supplemental chemical fertilizer inputs is reduced, and often eliminated, relative to a cereal or grass crop. This reduction provides direct private benefits to the producer in terms of time, energy, and input cost savings. By reducing the application of chemical fertilizer, the risk to ground and surface water posed by the buildup of excess nutrients in the soil is reduced. Alfalfa will generally absorb available residual N from the soil before utilizing fixed N (Russelle, 2004). As such, the inclusion of alfalfa in annual crop rotations can be a beneficial remediation strategy for areas with soil impacted by high residual N levels (Entz et al., 2001). Because alfalfa is a perennial crop, its root system is able to effectively penetrate deep into the soil zone and extract deeply embedded residual N more proficiently than would a more shallow rooted annual crop. However, proper management of an alfalfa stand is necessary to ensure that these environmental benefits are not squandered. For instance, Entz et al (2001) documented in field trials throughout western Canada that keeping alfalfa stands longer than four years may increase the risk of nitrate pollution as the plant begins to fix N at a rate higher than it uses for growth. Russelle (2004) also warns about the potential for N losses following stand termination, as large pools of plant-available N can be left in the soil which can be lost to the surrounding environment via leaching or runoff. When utilized properly, however, this source of N can form the basis of N credits available to subsequent crops. Therefore, a benefit of the alfalfa BMP is the reduced need for chemical fertilizer application on annual crops following the termination of the alfalfa stand (Entz et al., 2002). This benefit has been documented to last as long as seven years following alfalfa termination (Entz et al., 1995).

Other benefits of adding alfalfa to an agricultural production system include improved soil quality, structure, and organic matter content (Putnam et al., 2001). A key reason for these impacts is the enhanced protection of soil from erosion. As a perennial crop, a well-established alfalfa stand will have a dense canopy relative to an annual crop, which protects the soil from the erosion inducing energy of wind and rainfall (Sturgul et al., 1990). Reduced erosion not only maintains soil quality for agricultural production, but also limits the loss of sediment into waterways, thus improving water quality. Increased storage of SOC is also promoted by perennial crops such as alfalfa, due to a larger litter base and root system compared to annuals (Janzen et al., 1998; Entz et al., 2002; VandenBygaart et al., 2008). Mapfumo et al (2002) estimated this total C contribution to be 2.7 times greater for a perennial like alfalfa compared to an annual crop. In a study of the short-term (two years) net C impacts of adding alfalfa in an annual crop rotation in Manitoba, Maas et al (2013) found that a recently established alfalfa stand was almost double the sink for atmospheric CO_2 than a control annual crop (spring wheat and canola). This finding was attributed to the longer growing season period for net CO_2 uptake, expanded root system and C contribution from litter deposits, as well as reduced N_2O emissions (Maas et al., 2013).

A well-documented yield benefit to crops grown after an alfalfa stand has been found, likely due to the combination and cumulative effects of the afore-mentioned environmental impacts. Based on field experiments conducted in Alberta, Hoyt (1990) was among the first to report this yield benefit. Significant yield increases to wheat crops were found for up to thirteen years following the termination of an alfalfa stand, and were most pronounced in the first eight years (ranging from 66 to 114% greater than continuous wheat – fallow). An alfalfa/grass (bromegrass) mix stand also led to yield increases in wheat yield, but not bromegrass alone. Other long-term studies, including Campbell et al (1990) and Ellert (1995) have reported similar results. However, it should be noted that yield reductions have also been reported in cases where the relatively moisture-intensive needs of an alfalfa system induce soil moisture shortages in the following year when precipitation levels are sufficiently low (Brandt and Keys, 1982). This yield benefit, along with the fertilizer reduction benefits described earlier, are a significant reason why the inclusion of alfalfa in annual crop rotations can be economical to a private producer. For instance, taking these benefits into account, both Trautman (2012) and Xie (2014) found that the implementation of the alfalfa BMP by representative cropping operations in Alberta (both irrigated and dryland) can increase overall economic returns in some situations. Zentner et al (1992) also suggest that the inclusion of a perennial forage like alfalfa can reduce economic risk to a producer, resulting in long-term gains to profitability. Importantly, their analysis took place in the Dark Brown soil zone of the Canadian prairies, of which the current study area is also a part. In this analysis, the economic and environmental implications of growing alfalfa are both taken into account to assess the private and public impacts of adopting this particular BMP.

2.2.2 Legume Green Manuring

The practice of green manuring involves plowing a crop into the soil instead of harvesting it at the end of the growing season (AAF, 1993). Instead of being used to generate direct revenue for the producer, the green manure crop is instead incorporated back into the soil system. This practice can be considered a BMP as it provides several environmental benefits, including improved soil physical properties, increased soil organic matter, disruption of disease cycles and weed growth, as well as influx of nutrients into the soil for use by subsequent crops (MAFRI, 2012; Cherr et al., 2006). In principle, any crop can be used as a green manure source. However, because they have the ability to fix N_2 from the atmosphere, legumes are used most often (Cherr et al., 2006). Since a producer loses the ability to generate revenue when growing a crop designated for green manuring, legumes are a more cost-effective choice since they do not require inputs of supplemental chemical N fertilization.

Using an annual legume such as field peas, clover, lentil, vetch, tangier, or fababean as a green manure may be considered as a substitute for the practice of summerfallow in semi-arid regions of the Canadian prairies. Although the practice of leaving fields fallow for a growing season to retain soil moisture has been declining in popularity over the last several decades, a significant portion of farms in the study region still employ summerfallow. According to the 2011 Agricultural Census, over 20,000 acres of cropland in Lethbridge County were fallowed on an annual basis, accounting for nearly 5% of total cropland (Statistics Canada, 2011). The benefit of leaving a field fallow for a growing season is that soil moisture reserves can be increased, thus improving the chances of adequate moisture availability the following season. However, there are several long-term environmental and economic implications of this practice, including declines in soil organic matter, increased soil salinization, increased erosion from exposure to wind and precipitation, as well as depleted soil fertility (AAF, 2008; Zentner et al., 2004). The crop producer also loses the income that would otherwise be generated by growing a crop. It has been proposed that the full or partial replacement of summerfallow with the growing of an annual legume green manure crop for the purpose of green manuring may offset these detrimental impacts and over time prove to be an economic alternative for producers (Zentner et al., 2004). While more costly in terms of seed and management, long-term benefits such as improved soil health and fertility may contribute to increased crop yields and reduced fertilizer costs (St.Luce et al., 2015). When compared to wheat as the preceding crop, O'Donovan et al (2014) found that fababean green manure produced significant positive effects on both canola and barley yields.

However, the evidence is mixed regarding the long-term benefits of legume green manuring in dryland conditions. Zentner et al (2004), in an assessment of green manuring in the Brown soil zone of Saskatchewan, found that the management of a legume green manure crop is vital to the success of the BMP and the cropping system. Specifically, the timing of plowdown for the green manure crop affected both soil water depletion and subsequent N fertilizer savings. Benefits from the green manuring took several years to materialize, however, and Zentner et al (2004) suggested that it may take several rotation cycles until the benefits generated a meaningful economic impact for crop producers. The results of this study imply that modifying the practice in certain ways to minimize the impacts of soil moisture depletion can in fact make this BMP a viable management option in the Canadian prairies (Zentner et al., 2004). Other techniques, such as effective snow management, have also been documented to improve the green manure BMP (Brandt, 1990; Brandt, 1999).

In a more recent study, a series of three-year long field experiments involving legume green manuring was conducted across six locations in the Canadian prairies by St. Luce et al (2015). In this study, precipitation levels were above average every year in all but one of the locations. With this alleviation of moisture constraint, St. Luce et al (2015) found that using fababean as a green manure produced significant yield increases and reduced the economic optimum N rate for subsequent annual crops. Under irrigated conditions, where the moisture constraint is eliminated, green manuring has been documented to generate a consistent yield benefit (Walley et al., 2007).

The economic implications of this BMP to crop producers have also been studied. Khakbazan et al (2014) investigated the economic effects of green manure crops and found that although net revenue was highest in the year following the green manure (relative to other preceding crops like canola and wheat) it was insufficient to compensate for the loss of revenue during the green manure year. Trautman (2012) modeled the implementation of a green manure BMP on representative dryland crop farms in the Brown and Dark Drown soil zones of Alberta, and found that this BMP involved a net cost to producers in both regions. This equated to an 8% decrease in annualized returns in the Brown soil region and a 12% decrease in the Dark Brown region. Xie (2014), modeling a representative southern Alberta irrigated cropping operation, also found there were private costs to producers from adoption of this BMP.

Green manuring was selected as a BMP of interest for the current analysis because adoption of the practice will likely impact nutrient management at the field and farm level, thus having the potential to affect water quality. A more complete understanding of the tradeoffs between reduced chemical fertilizer application on subsequent crops, import of nutrient into the soil system from plowing down the legume green manure crop, and increased export of nutrients due to crop yield increases is essential in order to fully assess the value of this BMP in addressing the environmental concerns of interest, namely water quality.

2.2.3 Introduction of Field Peas

Another BMP considered in this analysis is the introduction of field peas to annual crop rotations. Field peas are an annual legume, and therefore also have the ability to fix nitrogen directly from the atmosphere to use for plant growth. Because of this ability, field peas have been documented to improve the soil fertility and nitrogen economy of cropping operations (e.g., Walley et al., 2007). Additionally, interest among crop producers in growing field peas has risen in southern Alberta over the last fifteen years as market demand has improved. According to AFSC (2014), the total number of insured acres devoted to field peas in Lethbridge County has increased from 1,209 in 1998 to nearly 18,000 in 2013. Similarly, the number of farms reporting growing field peas has increased from 10 to 51 over that same timespan. Field peas are nutritionally valuable to both humans and animals, as they are a good source of protein, lysine, and starch (Lafond et al., 2007). As such, field peas have increasingly been seen as a viable option for producers.

The inclusion of a pulse crop such as field peas in rotation with cereal-based cropping systems can produce several important environmental benefits. Firstly, the ability to derive nitrogen from the atmosphere reduces producer dependence on chemical fertilizer application (Zentner, 2002; Walley et al., 2007). Field peas have been shown to have relatively high nitrogen fixation rates when compared to other legume species. According to SAFRR (2005), field peas derive approximately 80% of total plant N from the atmosphere, which equates to about 178 pounds of N per acre under dryland conditions. This level of fixation is higher than lentil, soybean, chickpea, and dry beans, but lower than alfalfa, sweetclover, and fababean. At that level of N fixation, field peas are generally assumed to be able to cover their N requirements over the course of the growing season; moreover, research has found that growing field peas can increase the N supplying capacity of soils to subsequent crops (Walley et al., 2007). As such, this BMP has the potential to create private benefits to a producer in the form of chemical fertilizer reductions for other annual crops. In turn, lower fertilizer applications reduce the risk of nutrients leaching out of the soil and into groundwater.

Other environmental benefits also accrue to cropping systems that employ annual legumes in rotation with cereals. SAFRR (2005) describes legumes as "soil building" crops, which have beneficial effects on soil biological, chemical, and physical conditions. For instance, these benefits can include increases in soil organic matter, improvements to soil structure, reduction of soil erosion, and better soil moisture holding capacity (SAFRR, 2005). AAF (2008), reporting on a long-term study conducted in southern Alberta under dryland conditions, found that the diversification of cropping activities by adding field peas reduced the incidence of harmful weed species and was helpful in breaking up crop disease cycles. For example, controlling weed species with different herbicides reduces the risk of developing herbicide resistance (AAF, 2004c).

Lastly, many studies have investigated the yield impacts of including legumes species such as field peas in crop rotations and have found positive effects (Entz et al., 1995). In a recent study, St. Luce et al (2015) looked at the effects of preceding legumes, including field peas, on a spring wheat – canola rotation at six different sites in western Canada. The authors found evidence that both canola and wheat yields increase when following field peas relative to other annuals. O'Donovan et al (2014) corroborated these findings, showing that on average canola yield increased by 10% when following field peas rather than spring wheat. Stevenson and van Kessel (1996) found a higher impact on spring wheat yields, citing increases up to 43% after field peas. In Saskatchewan, Adderley et al (2006) compared the yield impacts to spring wheat of both field peas and lentils. They found that in conditions of low soil fertility, field peas were able to improve soil nitrogen levels and subsequent wheat yields more than lentils or continuous wheat. However, if soil nitrogen levels were already high, no significant yield impacts were found. Other studies, including Campbell et al (2011) and Williams et al (2014), have also documented positive yield effects in cropping systems due to pulse crops in a variety of Canadian agricultural contexts and conditions.

The economic implications of this BMP have also been investigated. Zentner et al (2002) concluded that, despite higher initial costs associated with seed, inoculants, and other inputs, including a pulse crop such as field peas in a cereal-based rotation can lead to higher farm incomes and reduce production and marketing risk. Khakbazan et al (2014) concluded that using field peas in crop rotations with canola or wheat can be economically beneficial for a producer. Averaged across several sites in western Canada, the net revenue in a 3-year cropping sequence featuring canola increased by 22% when field peas were added. Net revenue in a barley rotation increased by 20%. Trautman (2012) found that geographic location matters for this BMP, as positive economic impacts were found in the Brown, Dark Brown, and Dark Grey soil zones of Alberta, but not the Black zone. Trautman (2012) hypothesized that, because the Black soil zone is generally regarded as more productive, the displacement of higher-value crops such as canola in the crop rotations offset the benefits of the field pea BMP.

2.2.4 Manure Management

The application of livestock manure to cropland is a long-standing agricultural practice. Before the use of chemical fertilizers became widespread, manure was a valuable, and often the only, source of nutrients to promote crop growth. Although displaced somewhat by the proliferation of chemical fertilizer, the use of manure continues to provide many benefits to agricultural producers. Other than being a source for both macro (e.g. N, P, K, and S) and micro (e.g. Fe, Zn, Mn) nutrients, manure application can improve soil structure and tilth, increase soil organic matter, and develop drainage and water-holding capacity (Miller et al., 2002; Sharpley et al., 1998). Other biophysical benefits include enhanced soil enzyme activities (Lalande et al., 2003) and increased soil microbial biomass carbon (Lupwayi et al., 2005). Manure application is often suggested as an important remediation practice to address the long-term consequences of annual cropping systems. For instance, the degradation of soil quality, reduction of organic matter, and increase in topsoil erosion are all issues that can be partially addressed through the replacement of chemical fertilizer with manure as a crop nutrient source (Caldwell, 1998; Mozumder and Berrens, 2007).

However, manure can also be a source of environmental pollution when not managed properly. Although valuable as a nutrient source, manure is also expensive to transport and apply to cropland (Smith and Miller, 2008). This creates incentives for agricultural producers to disproportionately apply manure on cropland closer to the manure source (e.g., a feedlot) in order to minimize hauling costs. When manure is applied at a rate exceeding the nutrient requirements of crops the leftover nutrients may potentially build up in the soil and be at risk of migrating to ground and surface water (Sharpley et al., 1998). Excessive nutrient levels in water can lead to eutrophication, which is harmful to ecosystem health (Sharpley and Moyer, 2000). As such, regulations in Alberta stipulate that manure application must only be done at rates at or below the annual crop N requirements (AAF, 2008b). Other manure application regulations specified in the *Agricultural Operation Practices Act* (AOPA) include manure incorporation requirements, setback distances, record keeping requirements, and soil test nitrogen thresholds (AAF, 2008b). These regulations were designed to minimize the impact and risk that manure application can have on the surrounding environment. For instance, the requirement that manure be incorporated into the soil within 48 hours is made so that ammonium-N (NH_3) losses via volatilization into the atmosphere are minimized. Similarly, regulations on the minimum distance to a water body (setback distance) manure can be applied help to minimize the risk of nutrient runoff into surface water. The AOPA also specifies nitrate limits for soils receiving manure. These limits are variable, depending on the texture of the soil, distance to the water table, soil zone of the province, and farming method (i.e., dryland or irrigated). Nitrate limits are lowest for soils that are sandy (coarse-textured, which increases the risk that nitrate may be transported downwards into groundwater), less than 4 m to the water table, and dryland. On irrigated soils, nitrate limits are highest, and range from 180 to 270 kg per hectare.

Despite these regulations, concerns persist over the environmental implications of manure application. One issue arises when manure is applied at crop N requirements, P tends to accumulate in the soil. This occurs because the ratio of N to P used by most major grain and hay crops is 4:1, which exceeds the ratio of N to P (2:1) typically found in manure (Sharpley et al., 1998). Therefore, if N requirements of a crop are met at a certain manure application rate, the amount of P applied will be double that needed by the crop. Over time and with repeated applications, the excess P will accumulate in the soil (Olson et al., 2010). Another issue with using manure as a nutrient source is that the availability of nutrients is not consistent over time and sometimes will not coincide with crop needs. The majority of N and P in manure are in organic form, and thus not available to plants. Once these nutrients are converted to mineral form, plants are able to absorb and use them for growth. However, the mineralization of organic nutrients occurs at varying rates (e.g., 70% of organic P in manure is mineralized in the first year after application, versus only 25% of organic N) and is subject to a variety of environmental factors (e.g., precipitation, temperature).

Lastly, unlike chemical fertilizer, which is made to exact nutrient content specifications, the nutrient content of manure is generally quite variable. Unless a laboratory analysis is performed, nutrient levels can be uncertain and dependent on animal type, diet, storage methods, type and amount of bedding used, transport, and climate conditions (particularly moisture) (Eghball and Power, 1994). This uncertainty can lead to inadvertent over-application of manure and buildup of residual nutrients in the soil.

In a series of field experiments conducted in southern Alberta, Olson et al (2009) found that groundwater beneath coarse-textured irrigated soils is particularly vulnerable to nitrate leaching under high annual manure application rates. Four rates of annual cattle manure application (20, 40, 60, and 120 tonnes per hectare) were tested at two different irrigated sites, one with coarse-textured soil and the other with medium-textured soil. Nitrate accumulation in the soil profile increased linearly by application rate at each site, and significant movement of nitrate into shallow groundwater was found at the coarse-textured site under the highest application rates. Olson et al (2009) also caution that large precipitation events may incite leaching of residual nitrate into groundwater when substantial nitrate accumulation in the soil profile occurs. Chang and Janzen (1996), however, found that under dryland conditions excess nitrate is more likely to remain in the root zone of the soil. Olson et al (2010) also studied the risk of P loss under a range of manure application rates on irrigated fields. Generally speaking, P is more immobile and thus less prone to downward movement through the soil than N, due to the high P adsorptive capacities of calcareous soils found in southern Alberta (Lutwick and Graveland, 1978). Specifically, P has a tendency to adsorb to Al and Fe oxyhydroxide coatings on sediment and calcite surfaces in soil (Sims et al., 1998). However, when cattle manure is applied at high rates (between 20 and 120 tonnes per hectare per year), Olson et al (2010) found that a significant percentage of P inputs from the manure could not be accounted for in soil testing or crop removal, suggesting that P loss via leaching does occur. Similar to NO_3^- loss, coarse-textured soils are more at risk.

The application of manure is considered as an alternative/beneficial management practice in this analysis. When compared to chemical fertilizer application, several potential environmental and economic benefits may be realized. In particular, the results of several studies suggest that applying manure may result in a crop yield benefit relative to just chemical fertilizer application. Lupwayi et al (2005) found that canola, wheat, and barley yields at a northern Alberta site were significantly higher when cattle manure was applied on an N basis when compared to chemical fertilizer and control treatments. Yield increases between 25-75% were observed. Other studies have suggested that the application of manure in conjunction with chemical fertilizer can increase crop yields (Reddy et al., 2000). The positive soil quality impacts of manure described earlier (e.g., increased soil organic matter, improved structure) have been cited as reasons for this yield benefit (Black and White, 1973). Xie (2014), assuming a non-nutrient yield benefit between 1-5%, found that the application of cattle manure on a representative irrigated cropping farm in southern Alberta can increase economic returns. Not all studies have found evidence of this crop yield benefit, however, including some conducted in Alberta (Miller et al., 2002; Olson et al., 2009). Miller et al (2002) found improvements in several soil properties, but limited crop yield response in barley silage relative to chemical fertilizer treatments.

Alternative methods of manure application have been proposed to limit the environmental concerns associated with conventional N-based applications. Annual P-based manure application has been proposed as a BMP to address these concerns (Miller et al., 2011b). This strategy would reduce the overall amount of manure applied to cropland, and necessitate the use of supplemental chemical fertilizer to meet crop N requirements. However, by crop P requirements not being exceeded, P would not accumulate in the soil under this practice. A suggested variation of this BMP strategy is the application of manure at a rate of three times the annual P requirement, but done only once every three years (triennial). This change in strategy would help to alleviate the increased application costs to crop and livestock producers (Smith and Miller, 2008; Miller et al., 2011b). In a study conducted in southern Alberta, Miller et al (2011b) found that both annual and triennial P-based cattle manure application rates resulted in significant reductions of dissolved P fractions in runoff compared to annual N-based application. Importantly, no significant difference with respect to P loading in runoff was found between annual and triennial P-based application, suggesting that both practices can be implemented to generate improved environmental outcomes.

Manure management is considered as a BMP in this study due to the prevalence of manure use as a nutrient source on cropland in the LLB watershed. As will be discussed further in Chapter 3, the density of livestock operations within Lethbridge County and immediately adjacent to the study watershed make manure a common resource for crop producers in the area. As such, the implementation of a manure management BMP is especially pertinent to environmental and farm management.

2.2.5 Conversion to Permanent Forage

The conversion of annual cropland to permanent cover is an alternative land use that involves the cessation of annual cropping activities in favour of establishing permanent forage, grass, or tree cover. Many of the environmental benefits discussed in the section describing the alfalfa BMP (2.2.1) apply in this case as well, including increased soil organic matter and carbon storage, reduced erosion, and improved wildlife habitat, and enhanced biodiversity (AAF, 2004c; Jefferson et al., 2004; AAFC, 2007).

Often, areas that are marginal for crop production, or are either of environmentally sensitive or of high ecological value, are those targeted for this land use change. For example, land that is at high risk for soil erosion may be better suited to the cover of permanent vegetation instead of continuous cropping because the soil is protected year-round by fully developed vegetation. Areas with steep slopes or poor quality soils are also often better suited to forage production (AAF, 2004c). Whether it is for hay production, livestock grazing, or complete retirement from agricultural activities and into natural uses, the marginal quality of these targeted areas can lessen the private economic impacts.

Conversion to permanent forage can generate several significant environmental benefits. For instance, perennial vegetation improves the capacity of soil to store organic carbon when compared to annual crops (Desjardins et al., 2005). This is partly due to the high root to shoot ratios developed in perennials, resulting in increased biomass stored below ground level, as well as a higher return of plant residues to the soil (Dyck et al., 2015). The lack of soil disturbance in areas devoted to permanent cover compared to annual cropping activities also contributes to increased carbon storage (Boehm et al., 2004). According to Smith et al (2001), in the western Prairie provinces this benefit is most pronounced in the Black and Dark Gray soils of the sub-humid prairie region and lowest in the comparatively drier semi-arid Brown and Dark Brown soil zones. Another benefit vis-à-vis annual crop production is the drastically reduced levels of inputs into the agro-ecosystem. For instance, fertilizer and pesticide use is often minimized or eliminated on fields devoted to permanent cover, which reduces the risk of soil and water contamination. For these reasons, this BMP is relevant to the present analysis of land use in the LLB watershed.

Several policies and programs in Canada and the United States have been designed to incentivize the conversion of marginal agricultural land to permanent cover. In the 2000s, the Government of Canada initiated the Greencover Canada program, which provided funds to farmers who were willing to retire environmentally sensitive and/or marginally productive annual cropland. Other national programs have more recently taken up this strategy, including the Alternative Land Use Services (ALUS) program, which is administered on a more local and regional level (ALUS, 2016). A long-standing program in the United States is the Conservation Reserve Program (CRP), which also compensates farmers and landowners financially for leaving land in natural states (USDA, 2016).

2.3 Chapter Summary

This chapter explores the relationship between agricultural beneficial management practices (BMPs) and impacts to the surrounding environment. Two environmental quality outcomes are the focal point of this analysis: water quality and the storage of organic carbon in agricultural soil. A suite of BMPs were presented for analysis in this study, primarily for their potential to have positive impacts on the two environmental metrics of interest. These BMPs include the introduction of alfalfa, legume green manures, and field peas to annual crop rotations, improved management of livestock manure, and the conversion of annual cropland to permanent forage.

Chapter 3: The Study Area

The purpose of this chapter is to introduce the Lower Little Bow watershed, which is the spatial area of analysis in this study, as well as provide context regarding the typical agricultural practices, land uses, and environmental quality concerns that are prevalent in the southern Alberta region. Agricultural Census data from the County of Lethbridge, where the LLB watershed is located, are presented to provide the reader with a representative profile of agricultural activity in southern Alberta. A description of the LLB watershed follows, including a discussion of past land uses and management practices based on a survey of producers in the area. Lastly, an overview of current environmental quality concerns in the region is presented, including a summary of past research and initiatives conducted to address these issues. The motivation for this current research is guided by these issues, and the agricultural practices typical of the region form the basis of the modeling used.

3.1 Agriculture in the County of Lethbridge

The County of Lethbridge is located in southern Alberta within Agricultural Region #2[3], and is encompassed by the Oldman River Basin. The County of Lethbridge is an agriculturally prominent region of the province, with a diverse number of activities populating the landscape. These activities include cow/calf operations on pastureland, dryland and irrigated cropping, as well as intensive livestock operations in confined areas. According to the 2011 Census of Agriculture (Statistics Canada, 2011), the County of Lethbridge contains 933 total farms, the majority of which are used either for oilseed and grain farming (325) or cattle ranching and farming (285). When classified by industry, 230 farms were involved in beef cattle ranching and farming (including feedlots) and 55 farms were reported as dairy cattle or milk production. The total number of farms reporting area in some form of crop production, including summerfallow and hay farming, was 803. Table 3.1 displays the distribution of farm size (in acres) found in the county in 2011. Farms vary in size, although a majority (62%) are less than 400 acres.

[3] Agriculture Region #2 consists of four counties: Lethbridge, Warner, Taber, and Newell.

Table 3.1. The Distribution of Farm Size in the County of Lethbridge, 2011.

Acres	**Number of Farms**	**Percent of Total**
< 130	297	32%
130 - 399	282	30%
400 – 1,119	209	22%
1,120 – 3,519	116	13%
>3,520	29	3%
Total	**933**	**100%**

Source: Statistics Canada (2011).

The 933 farms consisted of a total of 696,670 acres of farmland, which was primarily used for crop production (514,337 acres), native range for pasture (79,393 acres), or tame or seeded land for pasture (60,873 acres). A total of 91 farms reported using a summerfallow practice, comprising 20,333 acres in 2011. Chemfallow was the most common form of weed control used on summerfallow land (14,817 acres). Irrigation was used by 558 farms on a total of 215,201 acres, the vast majority of which was used for annual field crops (173,033 acres) or alfalfa, hay and pasture (40,665 acres).

Regarding the production of hay and field crops, wheat, barley, and canola were the three most commonly grown annual crops in the County of Lethbridge in 2011. Less prominent crops receiving a significant share of farming acreage were oats, corn, field peas, dry beans, potatoes, and sugar beets. Alfalfa (and alfalfa mixtures) was the most common perennial forage crop. Table 3.2 summarizes the number of acres devoted to the production of various notable annual and perennial crops.

Table 3.2. Crop Production in the County of Lethbridge, 2011.

Crop	Number of Acres
Spring Wheat	123,197
Barley	115,228
Canola	101,032
Durum Wheat	24,539
Corn	20,595
Field Peas	16,045
Winter Wheat	9,309
Sugar Beets	6,079
Oats	6,027
Dry Beans	2,361
Potatoes	1,366
Other	15,281
Alfalfa and Alfalfa Mixtures	41,233
Other Tame Hay	20,731

Source: Statistics Canada (2011).

Commercial chemical fertilizer was used by 538 farms on 386,176 acres of farmland. A similar number of farms, 533, reported either producing or using manure. Of the 156,564 acres receiving manure application in the county, 57% of the area received manure that was spread naturally by grazing animals, 31% received solid or composted manure incorporated directly into the soil, and 6% received liquid manure incorporated directly. Manure that was not incorporated into the soil was applied to small percentage of the acreage (5.78%).

Other notable land use practices found in the county include the use of nutrient management planning (247 farms), the use of windbreaks or shelterbelts (178), and the planting of buffer zones around water bodies (96). Less common practices include the use winter cover crops (50) and green manure crops (29).

According to census data, gross farm receipts vary widely across the County of Lethbridge, ranging from less than $25,000 to greater than $2,000,000 per year. Table 3.3 displays the range and distribution in total gross receipts.

Table 3.3. Distribution of Farm Total Gross Receipts in the County of Lethbridge, 2011.

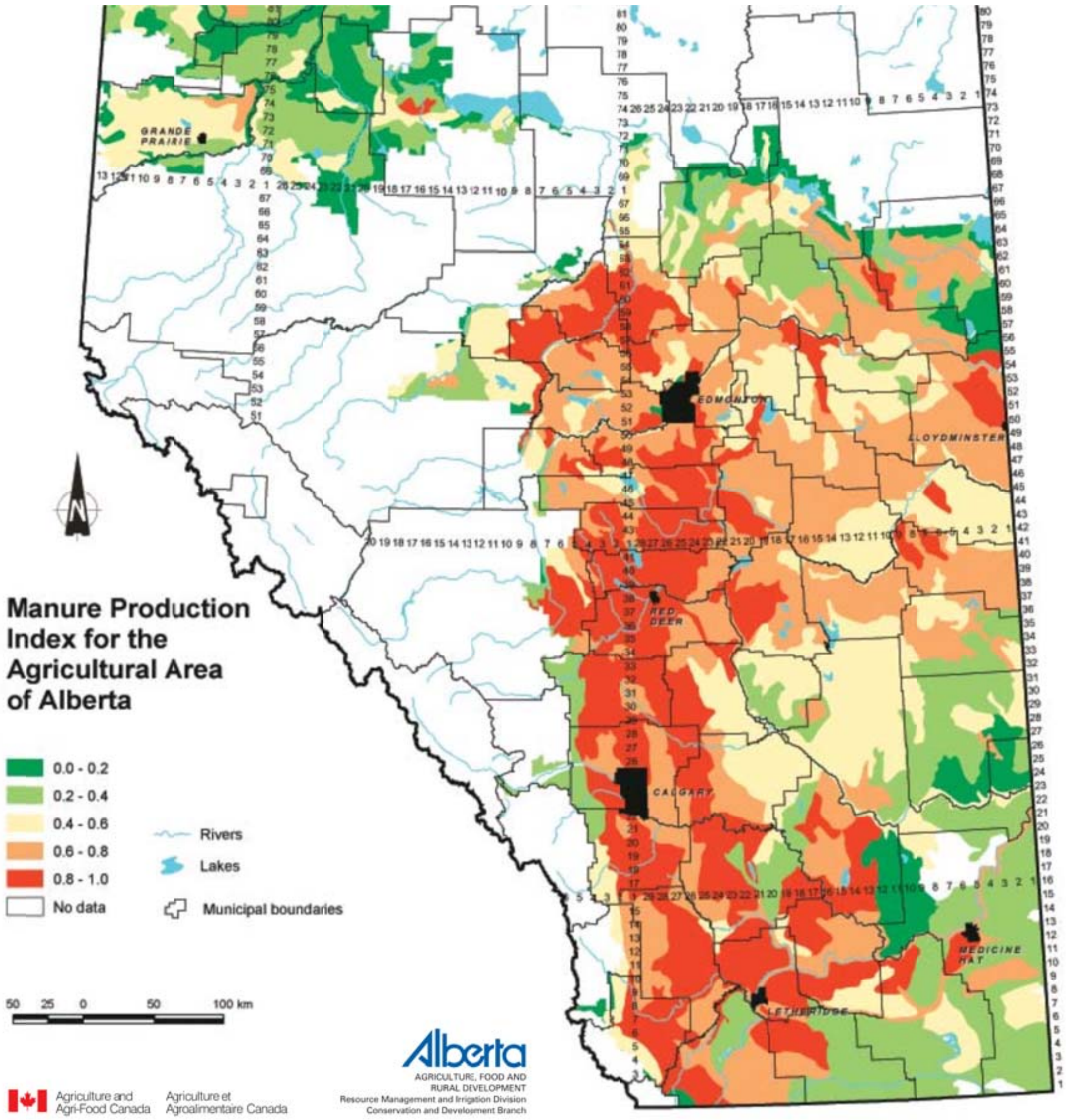
Total Gross Receipts	Number of Farms	Percent of Total
< $25,000	212	23%
$25,000 - $99,000	217	23%
$100,000 - $999,999	350	38%
$1,000,000 - $1,999,999	77	8%
> $2,000,000	77	8%
Total	**933**	**100%**

Source: Statistics Canada (2011).

Intensive livestock operations are densely located in the County of Lethbridge. In 2011, The County of Lethbridge ranked number one in the province of Alberta for both total cattle production (428,000 head) and total chicken production (1,330,000 birds), and had the fifth greatest number of hogs (66,000 head). Most of the intensive operations, including feedlots, are located within the vicinity of the cities of Lethbridge and Fort Macleod, with the highest density occurring to the north and east of Lethbridge (OWC, 2010). Less intensive operations, such as cow-calf facilities and range cattle, are found throughout the county. The prevalence of livestock in the area results in a high level of manure production. The concentration of these industries has the potential to create long-term environmental issues in the region, which are discussed further in section 3.2.1.

Figure 3.1 displays a map of a manure production index for agricultural regions of Alberta (AAF, 2005b). The index is based on the amount of livestock manure produced per square kilometer, which reflects the concentration of manure concentration found throughout the province. The index provides an estimate of the degree to which livestock operations may contribute to nutrient loading of soil and water, pathogens, and odour (AAF, 2005b). The region to the north and east of Lethbridge is coloured red, indicating an index score between 0.8 and 1; this score places the region in the upper range of manure production in the province.

Figure 3.1. Livestock Manure Production Index for the Agricultural Areas of Central and Southern Alberta.



Source: AAF (2005b).

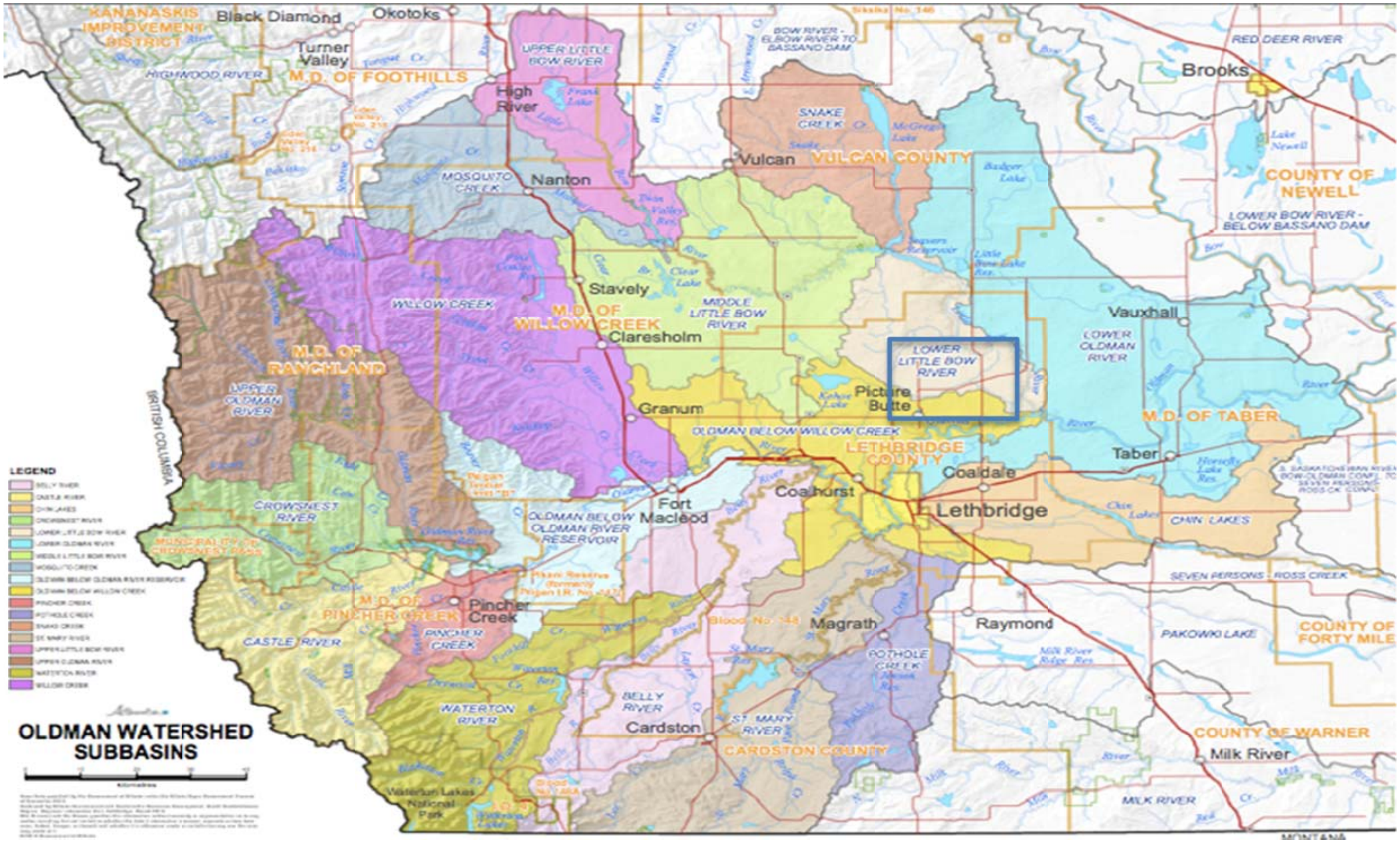
3.2 The Lower Little Bow Watershed

The LLB watershed is a 6,680 acre micro-watershed located within the prairie sub-basin of the Oldman River Basin, approximately 40 km northeast of the City of Lethbridge. The lower portion of the Little Bow River, flowing southeast from the Travers Reservoir north of Lethbridge to the Oldman River, runs through the watershed area. Agriculture is the dominant land use in the watershed, and primarily includes dryland and irrigated cropping along with livestock operations on tame and native range. Barley and wheat have historically been the two most common crops grown in the study area (Bremer et al., 2008; Rahbeh et al., 2013). The LLB watershed is contained within the mixed grass and dry mixed grass natural sub-regions (PEWC and AAF, 2014), and native grassland makes up a significant share of the land base.

The climate of this region is semi-arid, and receives relatively less moisture but more heat compared to the rest of the province (Rahbeh et al., 2011). Precipitation levels decline with distance from the Rocky Mountains, reaching a low for the Oldman River Basin at the confluence of the Bow and Oldman Rivers (east of the LLB watershed) at less than 300 mm per year. The LLB watershed averages approximately 380 mm of precipitation per year, about one-third of which is deposited as snow (AAFC, 2013). Closer to Lethbridge, average annual precipitation levels are higher and average about 450 mm annually (OWC, 2010). Irrigation is common in the watershed, and nearly 50% of the area is irrigated on a regular basis. Irrigation in the Prairie sub-basin of the Oldman River Basin is managed by several irrigation districts, including the Lethbridge Northern Irrigation District and the Bow River Irrigation District, the former of which provides a portion of the irrigated water supply to producers in the LLB watershed. The remainder is sourced directly from the Little Bow River (Rahbeh et al., 2011). Producers in the LLB watershed have historically followed a fixed irrigation schedule, with irrigation taking place on a weekly basis to depths of 38 mm and 44 mm for cereals and grasses, respectively (Rahbeh et al., 2011). The high level of heat found in the region results in both a high number of growing degree days and rate of evapotranspiration compared to the rest of the province (Rahbeh et al., 2011). The combination of a lack of precipitation and high temperatures explain the popularity of barley and wheat in the LLB watershed[4] (Koeckhoven et al., 2008). Both of these crops can be grown in a wide range of climatic zones, and are relatively resistant to low levels of moisture (AAF, 2006).

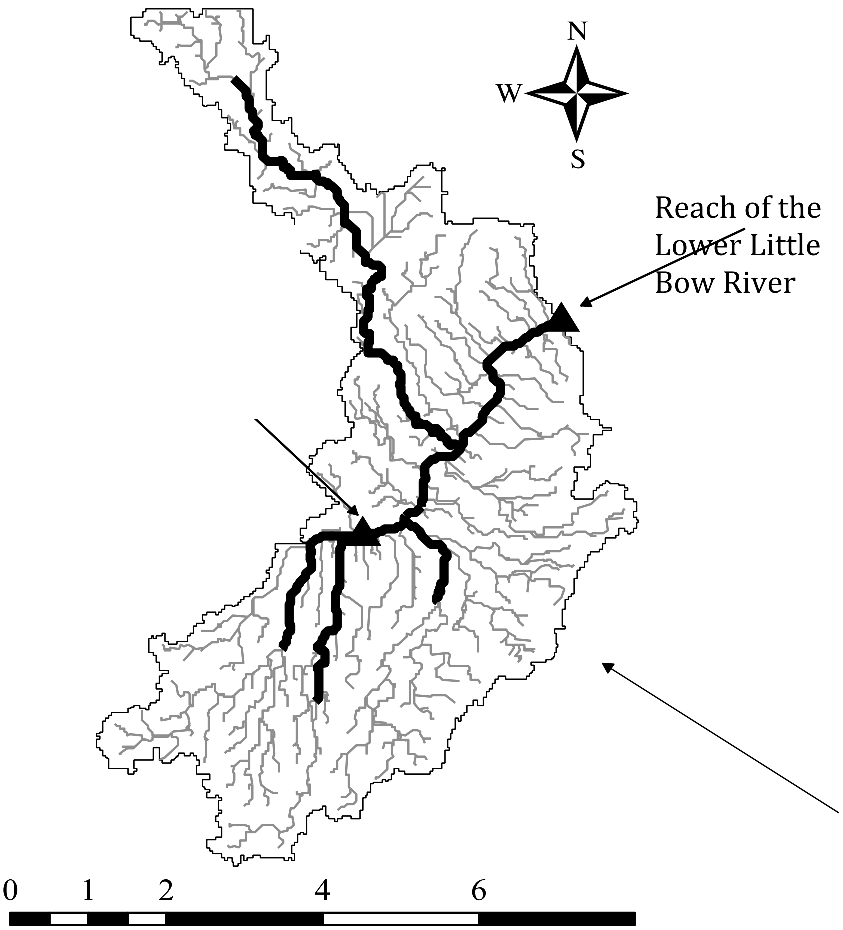
[4] See Table 3.4 for a snapshot of crop acreage in the Lower Little Bow watershed.

Figure 3.2. Spatial Extent of Sub Basins in the Oldman River Basin of Southern Alberta. [a]



[a] The Lower Little Bow micro-watershed is located within the Lower Little Bow sub-basin pictured above, and within the blue square to the northeast of the municipality of Picture Butte.
Source: OWC (2010).

Figure 3.3. Spatial Extent of the WEBs Lower Little Bow Watershed, Alberta.



Adapted from: Rahbeh et al (2011).

3.2.1 Environmental Issues and Previous Research

The diversity of agricultural land use in the LLB watershed make it a suitable site to investigate and conduct research into the impacts of agriculture on the environment and the potential effectiveness of BMPs in mitigating these impacts. These land uses, as well as the semi-arid climate, are broadly representative of common agricultural activities in southern Alberta. Additionally, the presence of the Little Bow River provides a natural laboratory for the analysis of agricultural BMPs with respect to impacts on ground and surface water quality, making this region a focal point for research into agri-environmental interactions.

As mentioned earlier, the intensity of agricultural activity in this area has had detrimental impacts on environmental quality. Concerns over water quality have been a long-standing issue for the region, and thus a featured point of extensive monitoring and research effort. At the forefront of this effort has been the Oldman Watershed Council (OWC), a non-profit organization that has partnered with Alberta Agriculture and Forestry (AAF) and various local communities to improve management and promote good stewardship of water resources in the Oldman River Basin. Founded in 2004 as a merger between the Oldman River Basin Water Quality Initiative and the Oldman Basin Advisory Council, this organization (or previous incarnations) has led water quality monitoring of the Little Bow River since 1999 (OWC, 2010). Monitoring has occurred along various stretches of the river, including at the mouth (point of exit from the Travers Reservoir) and near the confluence with the Oldman River (south of the current study area).

In a recent report regarding the overall health of the Oldman River Basin, water quality in the Prairie sub-basin (where the LLB micro-watershed is located) was rated as Fair to Poor (OWC, 2010). High and increasing annual loads of nitrogen, phosphorus, fecal coliforms (bacteria), and total suspended solids (TSS) in river flows over the last ten years were the primary reasons for this rating. Total Nitrogen (TN) in the Little Bow River exceeded the Alberta Environment Surface Water Quality Guidelines for Protection of Aquatic Life threshold (1.0 mg per L) in past years, including most recently in 2003 and 2006 (OWC, 2010). These instances coincided with high river flow years, due to higher than average levels of precipitation in the area. High levels of precipitation can increase surface runoff from farmland, which will increase the transport of nutrients (such as N and P) to surface water bodies. Drier conditions reduce the prominence of this nutrient loss pathway, and therefore residual N and P will tend to remain longer in the soil (Little et al., 2003). Monitoring also indicated that the surface water quality thresholds of Total Phosphorus (TP), which is 0.05 mg per L, were frequently exceeded between the years 1997 and 2009. For instance, this threshold was exceeded by up to 0.5 mg TP per L in every year between 2003 and 2009 at monitoring sites upstream of the LLB micro-watershed (OWC, 2010).

Little et al (2003) studied the relationship between water quality in the Little Bow River and land use in the 55,000 ha Lower Little Bow sub-basin between 1999 and 2002. The proportion of cereals, irrigated land, and intensive livestock feeding operation density were all significantly and positively related to a suite of poor water quality indicator variables (Little et al., 2003). These variables included concentration of TN, NO_3^-, and TP in the Little Bow River. Conversely, the same variables were inversely related to the proportion of land used for native range. Scott (2012) analyzed water quality data from samples collected during 2004-2007 in the Little Bow River, and found that water quality was adversely affected by exceedances in fecal coliform, dissolved oxygen, and total suspended solids thresholds.

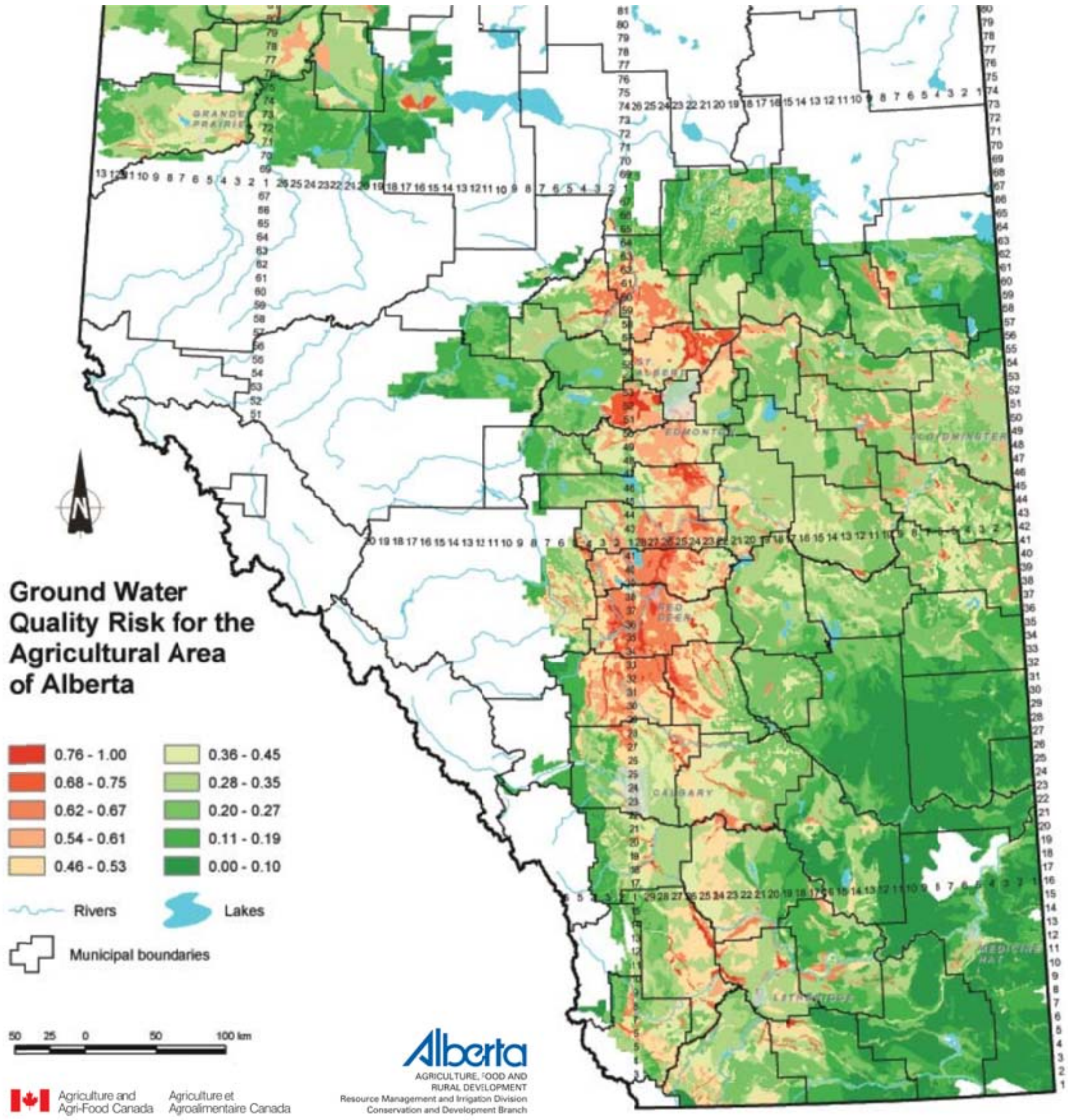
Another significant pathway of nutrient loss from agricultural land is via leaching through the soil profile and into groundwater. The contamination of groundwater by water-mobile forms of nitrogen, such as NO_3^-, has been studied extensively in the LLB watershed and adjacent regions of the Prairie sub-basin. Rodvang et al (1998) investigated the vulnerability of shallow, unconfined aquifers below seven irrigated and manured cropland sites of Lethbridge County. NO_3^- from excess fertilizer and the mineralization of organic N in manure was especially prone to leaching through sandy, coarse-textured soils above shallow groundwater. In several fields, Rodvang et al (1998) found the concentration of NO_3^- from fertilizer in shallow groundwater was between two and three times the maximum acceptable level for human consumption.[5] At the highest

[5] The widely accepted threshold of NO_3^- for safe human consumption is 45 mg per L, or 10 mg per L if measured as NO_3^--N (Health Canada, 2013; WHO, 2011). The human-related health effects were discussed in Chapter 2, section 2.1.1.

fertilizer and manure application levels (approximately 90 kg N per acre per year), nearly 65% of the N applied was available for leaching after crop use. In a continued evaluation of the same sites, Rodvang et al (2004) found that NO_3^- concentrations in groundwater continued to significantly increase over time, and that the subsurface flow of groundwater into lower-lying areas would transport and eventually discharge NO_3^- into surface water. Across the study area, average NO_3^--N concentrations in groundwater increased from 12.5 mg per L between the 1994-1996 period to 17.4 mg per L in the 1999-2001 period. This trend suggests that denitrification has not been entirely effective in the attenuation of NO_3^- concentration in groundwater. Under a range of groundwater recharge rates and nutrient removal rates via riparian areas, Rodvang et al (2004) predicted that NO_3^- concentration in the Oldman River could rise by a factor of at least 4.3 and eventually exceed water quality guidelines. Not surprisingly, the authors also found that NO_3^- concentrations were significantly lower in shallow aquifer water below less intensive agricultural activities, namely pasture or native range. Similarly, fields with fine textured soils were less at risk of NO_3^- leaching through the soil profile.

In a more recent study, Kohn et al (2015) assessed the long term temporal trends in groundwater NO_3^- concentration in an area near the town of Picture Butte, known as the Battersea drain, between 1994 and 2014. This area is located immediately south of the LLB watershed, and north of the City of Lethbridge. The Battersea features many of the same agricultural activities and land uses as the LLB watershed, including extensive irrigation of cropland and a proximity to a high density of livestock operations. On a regional basis, average NO_3^- concentrations were found to be slightly increasing over time. However, the authors found that the effects of land-use practices to have a more prominent impact at the local, field-level scale. The fields most at risk of contamination from agricultural sources are those with subsurface geology favourable to the downward movement of pollutants (i.e., coarse-textured soils, shallow aquifers).

Figure 3.4. Groundwater Quality Risk for the Agricultural Areas of Central and Southern Alberta.



Source: AAF (2005c).

Figure 3.4 displays a map of central and southern Alberta indicating the level of risk to groundwater quality (AAF, 2005c). Areas of red indicate a higher risk. This risk was calculated from an assessment of the abundance, mix, and intensity of agricultural activities (e.g., livestock operations, crop production, chemical use) in a particular area, coupled with physical characteristics of an area such as aquifer vulnerability and available moisture. The drier climate of southern Alberta makes the region comparatively less vulnerable to groundwater contamination than areas of central Alberta. However, the dry climate characteristic can be offset by the relative level of intensity of agricultural operations northeast of Lethbridge (including the LLB watershed), elevating risk levels (De Jong et al., 2009).

3.2.2 The Watershed Evaluation of Beneficial Management Practices (WEBs) Project

The LLB watershed was selected as one of nine representative agricultural watersheds throughout Canada for the Watershed Evaluation of Beneficial Management Practices (WEBs) project undertaken by AAFC. Broadly, the goal of the WEBs project was to determine the water quality and private economic impacts of implementing agricultural BMPs, and ultimately contribute to and improve land-use decision-making at farm, regional, and provincial levels (AAFC, 2013). A total of five BMPs were selected for evaluation at the LLB watershed study site between 2004 and 2008: streambank fencing of livestock, off-stream watering of livestock with no fencing, conversion of annual cropland to greencover, manure management, and buffer strips. Various biophysical metrics were tracked during the assessment of these BMPs.

Cattle grazing on land adjacent to streams or rivers can negatively impact water quality and riparian health. When cows use surface water bodies as a drinking water source, they contribute contaminants such as sediment, nutrients, and pathogens directly to the water in the form of fecal deposition (Miller et al., 2010). Additionally, soil compaction and disproportionate grazing pressure due to increased cattle traffic can have an adverse impact on riparian vegetation, which influences nutrient runoff from adjacent agricultural land. Fencing of the streambank, which excluded cattle in an adjacent native range pasture from access to the river, was introduced as a BMP along an 800 m stretch of the Little Bow River as part of the WEBs project (Miller et al., 2010). Rainfall simulations were conducted in order to measure various runoff variables. An improvement in environmental quality due to the BMP was found in some areas, namely with respect to riparian health and reductions in several runoff variables such as mass loads of TN, total dissolved N, and TP in some years. Miller et al (2010) suggested that improved riparian health led to an increased capacity to buffer the river from nutrient runoff, compared to the case with more bare soil and higher compaction when cattle were present.

Off-stream watering is another management technique that can help mitigate pollution of streams and rivers from livestock grazing on nearby pastureland (Miller et al., 2009). Drinking water troughs strategically placed away from a natural water body will reduce the incidence of nutrient deposition (via fecal matter). However, shifting the distribution of cattle may increase nutrient contamination of soil and groundwater below water troughs. The application of this BMP to the LLB watershed was studied by Miller et al (2009), who found that shifting the distribution of nutrients in this way was an effective means of reducing N levels in soil adjacent to the Little Bow river. However, accumulation of N in the soil subsurface around off-stream watering sources indicated that leaching of NO_3^- may eventually be of concern. Rainfall simulations were again conducted to determine the impacts on water quality in the LLB river, which revealed little to no improvement in a majority of runoff variables. Miller et al (2009) suggest that a longer-term evaluation may be necessary to elicit the true BMP response in terms of water quality impacts.

Another BMP in the LLB watershed investigated as part of the WEBs project was manure management. The water quality impacts of annual and triennial P-based cattle manure applications were compared to the standard practice of annual N-based application. Both of the alternative application methods significantly reduced dissolved P fractions found in runoff (Miller et al., 2011b). However, no environmental benefit was found with respect to annual P-based application over triennial P-based application, suggesting that either BMP may be a viable management option for producers in the region that utilize manure on cropped fields (Miller et al., 2011b). The implementation of a version of this practice is modeled for cropped fields across the LLB watershed in this analysis, and is discussed further in Chapter 5.

In 2005, buffers strips were established at the base of an irrigated annually cropped field next to the LLB river (Kalischuk et al., 2008). A range of vegetative species were used, including tame grass and alfalfa mix, barley, and a mix of shrubs and native grass. Results showed that, in years with extreme precipitation levels, NO_3^- loss was highest in the barley treatment. However, because only small amounts of surface runoff generally occur in the study region, buffer strips less than 6m may sufficiently reduce the risk of NO_3^- loss from a fertilized field (Kalischuk et al., 2008). It was recommended that the best vegetative cover from a producers perspective may be a tame grass mix that included deep-rooted alfalfa. This particular type of buffer strip would be both economical (i.e., generate some revenue for a producer) and effective against sub-surface movement of nutrients into surface water (Kalischuk et al., 2008). More recent studies in the LLB watershed have also examined the effectiveness of vegetated buffers to retain nutrients and reduce transport of sediment (Miller et al., 2015; Miller et al, 2016).

Lastly, the conversion of annual crops to forages was studied as a BMP in the LLB watershed as part of the WEBs project as well. Two fields initially used for annual crop production (barley) were converted to alfalfa in 2006 and 2007. Rainfall simulation was used to measure impacts to runoff water quality and quantity. Mixed results were found, and, depending on the field and year of study, certain water quality variables were improved. However, the authors concluded that the BMP did not improve the overall water quality in runoff (Miller et al., 2008b). This BMP is also investigated further in this analysis, and the modeling strategy is discussed further in Chapter 5.

Other research done in the LLB watershed as part of the WEBs project conducted by AAFC included a spatial analysis of land use and topographic characteristics and construction of nutrient balance budgets (both N and P) based on reported land management practices (see section 3.2.2). Hydrologic modeling utilizing the Soil and Water Assessment Tool (SWAT) was also undertaken (Rahbeh et al., 2011; Rahbeh et al., 2013). Finally, economic analysis was conducted on the adoption of several of the BMPs studied, including streambank fencing and off-stream watering, on a representative farm in the watershed (Koeckhoven, 2008). Koeckhoven (2008) concluded that BMP implementation for riparian and water quality protection is costly to producers and economic incentives would be necessary to encourage adoption. Smith and Miller (2008) also investigated the economic implications of various manure management scenarios, including annual and triennial application to adjacent cropland based on crop P-requirements, to a representative feedlot operator. Increased private costs were found to be associated with P-based application methods relative to N-based, although annual P-based application was more costly than triennial (Smith and Miller, 2008).

3.2.2.1 Producer Survey

A survey of producers in the LLB micro-watershed was undertaken in 2007 as part of the WEBs project (Bremer et al., 2008). Of the sixteen producers in the region, fifteen took part in the survey representing 59 of the 60 total quarter sections used for agricultural activity. Several quarter sections were sub-divided further based on differing land use in different areas, resulting in a total of 65 'fields' as the unit of analysis. A questionnaire was developed to assess a range of agricultural practices, including typical crop rotations, fertilizer use, manure application practices, crop yields, and stocking rates for each field.[6] The questionnaire was administered by the County of Lethbridge soil conservation technologist via phone or in-person interviews. For the purposes of the WEBs project, this information was used to construct an N and P balance budget for the watershed. The producer survey was used in the present analysis to establish a baseline of land use in the watershed for comparison with BMP adoption outcomes, including typical crops grown, fertilizer use, and manure application practices. More detail on the use of the producer survey results is provided throughout Chapter 5.

Table 3.4 provides a snapshot of land use in 65 fields of the LLB watershed based on answers given in the producer survey. The reported land use is for the year 2006. A total of 3,040 acres are irrigated, 2,720 of which were used primarily for annual cropping. Other reported crops grown in the watershed included sugar beets, corn (for silage), and triticale.

[6] For a more detailed description and discussion of the survey, including questions presented in the questionnaire, see Appendix 1 of Bremer et al (2008).

Table 3.4. 2006 Land Use and Crop Acreage in the Lower Little Bow Watershed.

Land Use		**Total Acres**	**Percentage**
Annual Cropping	Barley	2,360	35.33%
	Canola	920	13.77%
	Spring Wheat	320	4.79%
	Durum Wheat	80	1.20%
	Oats	240	3.59%
	Rye	320	4.79%
Perennial Cropping	Alfalfa	120	1.80%
	Timothy	80	1.20%
Pasture		2,240	33.53%
Total		**6,680**	**100.00%**

According to the producer survey, manure application on fields of the LLB survey is common. However, a range of practices with respect to manure application were reported, differing in amount, frequency, and type of livestock manure used. For instance, application levels as high as 45 tonnes of cattle manure every four years were reported on certain irrigated cropping fields. In contrast, levels as low as 15 tonnes every five years were also found. In general, lower levels were applied to non-irrigated cropping fields, and areas used for pasture rarely received manure. Of the 39 fields designated for annual cropping, 25 received manure on a regular basis. Both hog and cattle manure were utilized, although cattle manure tended to be more common. Additional fertilization with chemical fertilizer in intervening years between manure applications was also a common practice.

Stocking rates on pasture fields used for grazing were also reported, revealing a wide variety of use and livestock type. Cattle were the most common livestock grazed in the watershed, however sheep and horses were also present on some fields. A high level of ambiguity was found in the responses to stocking rate, preventing a more accurate estimation of average and typical practices in the area (Bremer et al., 2008). For instance, the duration of grazing was often not reported, or reported in vague terms.

3.2.2.2 Soil Testing

Soil testing was conducted during the WEBs project as part of the spatial analysis of land use and topographic characteristics in the LLB watershed (Miller et al., 2008). GIS (geographic information systems) software was used to map landforms, land use, and various hydrologic features along with physical and chemical soil properties in order to test for significant relationships.

Surface soil samples were collected from 251 locations within and adjacent to the watershed in the summer of 2006 (Miller et al., 2008a). Among other variables, NO_3^- (in ppm) and P (mg per kg) test levels were reported, along with a percent sand, silt, and clay texture. Calibrated for sample depth and collection time of year using methods provided by Kryzanowski et al (1988), NO_3^- and P results from the sampling were converted to a lbs per acre measure. For the present analysis, the samples were spatially referenced to the 65 agricultural fields in the LLB watershed, resulting in three or four samples for each field on average.

Table 3.5 displays the distribution of soil test N levels across the fields of the watershed, and Table 3.6 displays the corresponding distribution of soil test P levels. Generally speaking, soil N levels in the LLB watershed in 2006 were not a significant reason for concern. According to the limits set out in the Agricultural Operation Practices Act (AOPA), the upper limit of NO_3^- levels in soils of the Dark Brown region can range from 100-150 lbs per acre depending on the soil texture and subsurface distance to the water table (AAF, 2008b). However, soil P levels are a more significant concern, as nearly of a majority of fields (49%) had P levels exceeding 80 lbs per acre. Previous research in Alberta has indicated that soil test P levels exceeding 100 lbs per acre (200 mg P per kg of soil) pose high risks of P losses in runoff (AAF, 2007b). In 2006, 25 fields exceeded this threshold.

Table 3.5. Distribution of Soil Test N Level Across 65 Fields in the Lower Little Bow Watershed in 2006.

Soil Test N [a] *(lbs acre^{-1})*	**Number of Fields**	**Percentage**
< 20	29	45%
21 - 40	18	28%
41 - 60	6	9%
61 - 80	7	11%
> 80	5	8%
Total Fields	**65**	**100%**

[a] Per two feet of depth.

Table 3.6. Distribution of Soil Test P Level Across 65 Fields in the Lower Little Bow Watershed in 2006.

Soil Test P [a] *(lbs acre^{-1})*	**Number of Fields**	**Percentage**
< 20	6	9%
21 - 40	7	11%
41 - 60	10	15%
61 - 80	9	14%
> 80	32	49%
Total Fields	**65**	**100%**

[a] Per two feet of depth.

3.3 Chapter Summary

In this chapter, an overview of agricultural activity both in Lethbridge County and the LLB watershed was provided. In addition, previous research on and monitoring of environmental issues in the LLB watershed was discussed, along with a summary of the WEBs project undertaken by AAFC. Finally, findings from a survey of agricultural producers in the LLB watershed conducted in 2007 were presented, along with results of soil sampling done throughout the watershed in 2006.

Chapter 4: Methodological Approach

In this study, economic returns from agriculture in the Lower Little Bow watershed are evaluated in conjunction with certain indicators of environmental quality. Alternative agricultural land use (BMP) scenarios in the LLB watershed are developed for comparison with a baseline scenario in order to assess these tradeoffs.[7] The methodological theory and approach employed in this study to measure the private and public benefits (and costs) of each scenario is outlined in this chapter, along with a discussion of alternative modelling approaches. Private benefits, which are represented by the watershed-level economic wealth impacts of implementing a BMP scenario, will be assessed using an enterprise budgeting technique, where modified net cashflows are built and analyzed within a Net Present Value (NPV) framework. Public benefits, which encompass the environmental quality impacts of BMP and alternative land use adoption, are determined via the construction of watershed-level nutrient balance budgets and calculation of changes to soil organic carbon storage using published estimates of carbon factors in agricultural soils. To conclude the chapter, a summary of the methodologies used and an introduction to the land use scenarios is provided.

4.1 Approaches to Modelling Agricultural Systems

Complex agricultural systems, such as the LLB watershed, require sophisticated modelling techniques in order to accurately represent the dynamic interaction between economic and environmental parameters. Agricultural production processes are subject to a wide variation in biophysical factors, which introduces uncertainty and risk in decision-making. Similarly, the impacts of agricultural activity on environmental quality are frequently subject to variable elements and are often difficult to predict. For instance, a rare high intensity precipitation event may cause surface runoff of previously applied fertilizer P from the soil into adjacent water bodies. Additionally, the prices of both inputs and outputs of the production process are subject to change, which influences the production decision. Several methodological approaches have been used extensively in

[7] A detailed discussion of the alternative land use scenarios is provided in Chapter 5.

the literature to capture these interactions within agricultural systems, including mathematical programming, simulation analysis, and capital budgeting. These approaches are discussed here, followed by a description of the methodological approaches used to track changes in the environmental parameters of interest.

4.1.1 Mathematical Programming

Under a specified set of requirements or constraints, mathematical programming can be used to determine an optimal outcome. Given a mathematical objective function and a set of constraints, a 'best', or most efficient, solution can be identified. An objective function can be either maximized or minimized, depending on the problem. For instance, in an agriculture context, mathematical programming can be used to maximize farm profit under a set of land, labour, and cost constraints. Additional constraints may be added depending on the nature of problem; for example, the decision-maker may also have to follow an environmental quality regulation where only a certain amount of manure can be applied to a given area. Smith and Miller (2008) used mathematical programming to determine the costs to livestock producers near the LLB watershed associated with different levels of manure application. The BMP of P-based manure application was introduced as an additional constraint in the optimization problem. In general, the coefficients of the objective function are fixed parameters that give the contribution of each activity to the value of the overall objective function. The constraint set defines the technical relationship between resource usage for each activity and the resource endowment available to the decision maker. Mathematical programming can be either linear or non-linear, depending on the assumptions about model relationships.

This approach has been used extensively in the agri-environmental system context. De Laporte et al (2010) developed a mathematical programming model to examine the tradeoffs between agricultural returns and environmental benefits in the Eramosa watershed in southwestern Ontario. The agri-environmental relationship was explicitly defined using a GIS (Geographic Information System) based hydrologic model, which was able to capture and quantify the environmental impacts in terms of sediment abatement from preserved wetlands and areas of wildlife habitat. These were used in the mathematical programming model to develop the objective function coefficients and constraints. Yang and Weersink (2004) used a similar approach to investigate the cost-effective targeting of land retirement for establishing riparian buffers in another agricultural watershed (Canagagigue Creek, Ontario). The use of mathematical programming in this study allowed the researchers to identify where on the landscape a BMP (riparian buffers in this case) may be implemented to best achieve the greatest environmental benefits for the least cost. Rivest (2009) also investigated the tradeoffs between environmental quality and agricultural profitability in a Canadian context using mathematical programming models.

This methodological approach was deemed unsuitable for this study. The primary reason for this is a lack of identifiable landscape heterogeneity that would impact agricultural productivity potential. The watershed encompasses a small area (6,680 acres) with little to no variation in biophysical factors (e.g., soil type, precipitation). The attractiveness of a mathematical programming model is then reduced without this variation as it would be difficult to distinguish between different fields in terms of the economic impact of particular land uses.[8] This increases the potential for "corner solutions" which involve complete specialization in one particular land use. These types of solutions are not realistic but without further information and model complexity (e.g., incorporating risk and risk aversion) are difficult to avoid.

4.1.2 Simulation Analysis

Simulation modeling is another quantitative tool that can be used to assess agricultural systems. This technique involves the construction of a model that is designed to encompass the variables and relationships that make up a real world system. Specifically, simulation can be thought of as the "process of building a mathematical or logical model of a system or decision problem, and experimenting with the problem to obtain insight into the system's behaviour or to assist in solving the decision problem" (Evans and Olson, 2002, pg. 2). In this sense, the model takes specified input parameters and converts that into a set of predicted output measures (April et al., 2003).

In an agricultural context, a simulation model can be used to forecast outcomes of farm economic performance based on a combination of either decision variables (e.g.,

---

[8] In Scenario 4c of the analysis, certain fields are characterized as 'marginal' based on the Land Suitability Rating System (AIWG, 1995) to reflect the potential for decreased productivity in these areas. A discussion of this decision is provided in section 5.2.5.3.

fertilizer use, management practices) and random variables (crop and input prices, crop yields). Stochastic simulation models are those that incorporate one or more random variables, whereas a deterministic model contains only specified, non-random variables (Carson, 2003). Incorporation of stochastic elements into a simulation model provides a potentially useful way to model the production and/or market risk inherent in agricultural production. A specific form of stochastic simulation that has been used in previous studies is Monte Carlo simulation, which involves the generation of a large number (often thousands or hundreds or thousands) of possible outcome paths and can be used to incorporate random variation, lack of knowledge, or error in the evaluation of system performance and reliability (Mun, 2006). Monte Carlo simulation methods were used in Trautman (2012), Xie (2014), and Cortus (2005).

Previous studies on farm-level economic performance in the LLB watershed have incorporated stochastic simulation elements into the evaluation (Koeckhoven, 2008; Xie, 2014). The focus of these studies was on farm-level outcomes and decision making, which limited the scope of the analysis to private benefits and costs. Instead of modeling a representative farming operation, a watershed-level analysis is the focus of this study and certain public benefits (i.e., improved environmental quality) are incorporated. While adoption of various BMP practices occurs at the farm level, the impacts of these practices are often only measureable at a more aggregated level such as the watershed (Jeffrey et al., 2012). As such, this study represents a first step toward investigating impacts of watershed-wide BMP adoption. Although certain random variables are included in the model structure[9], a less complex (i.e., deterministic) approach is utilized in the modeling of basic economic and biophysical relationships. Similar to Koeckhoven (2012) and Xie (2014), economic returns are assessed using an enterprise budgeting technique, where modified cashflows are built and analyzed in a NPV framework (see the following section). However, various stochastic elements present in those studies, such as the variability of crop prices and yields, are instead treated as deterministic.[10] The sensitivity of the model results to deterministic elements is then investigated through sensitivity

[9] For example, the N contributions of biological fixation in leguminous crops such as alfalfa are represented by a range of possible values (see section 5.2.2.4).
[10] See Chapter 5 for a discussion of how crop price and yield parameters are specified.

analysis (see Chapter 6). As such, the model constructed for this study contains elements of a simulation analysis, which are used in conjunction with capital budgeting techniques.

4.1.3 Capital Budgeting and Net Present Value Analysis

The costs and benefits associated with BMP implementation in the LLB watershed are evaluated using a capital budgeting technique. A theoretical discussion of capital budgeting and an empirical description of the model used in the analysis is provided in this section.

4.1.3.1 Capital Budgeting

Capital budgeting is a planning tool used to evaluate long-term investment decisions. The business of farming requires agricultural producers to make a multitude of long-term investment decisions, including whether or not to adopt a BMP in their operation. Assuming that decisions are made consistent with wealth maximization, the capital budgeting technique can inform the producer as to which investment decisions are the most potentially beneficial (i.e., increase wealth). To be consistent with the objective of wealth maximization, all operational cashflows must be considered and evaluated over a period of time, discounted at the opportunity cost of capital (Copeland and Weston, 1988). Investments, or projects, that increase the wealth of the operation are considered beneficial and should be undertaken. Conversely, those that decrease wealth should be rejected.

Commonly used capital budgeting techniques include internal rate of return (IRR), payback period (PP), accounting rate of return (ARR), and net present value (NPV). A discussion of the relative merits of each of the above four techniques is provided by Copeland and Weston (1988). Both IRR and NPV use discounted cashflow calculations, which are suitable for evaluation of separate investments (or projects) over a period of time (Ross et al., 2003). Typically, an investment decision involves an initial capital outlay followed by a stream of cashflows (both positive and negative) generated over time. Both the timing and magnitude of cashflows, as well as the time preferences of the decision-maker, must be taken into account by a capital budgeting technique. While decisions such as the implementation of crop rotation BMPs do not require an initial capital outlay, benefits and costs are borne out over the project period. Thus using a capital budgeting technique such as NPV or IRR is important. In this study, NPV is used to assess the potential private profitability vis-à-vis the opportunity cost of capital for BMP implementation decisions in the LLB watershed. NPV is chosen due to certain conceptual and computational advantages, one of which being that it is a metric consistent with the assumption of wealth maximization (Copeland et al., 2005; Ross et al., 2003).

4.1.3.2 Net Present Value Analysis

NPVs are calculated by discounting future cashflow streams to a present value. Present value (PV) is a method to express the value of future cash streams or payments in terms of their worth at the present time. PV is a useful method to reduce complex cashflows to a simple value for purposes of investment (or project) comparison. Taking into account the time value of money, the PV of a stream of future cashflows is calculated using the following formula (Ross et al., 2003):

$$PV = \frac{C_1}{(1+r)^1} + \frac{C_2}{(1+r)^2} + \frac{C_3}{(1+r)^3} + \dots \frac{C_n}{(1+r)^n} \tag{4.1}$$

where C_t is the net cashflow in time t (t=1, 2, … n), r is the discount rate, and n is the number of years considered for the investment. In the current analysis, each investment (watershed-wide BMP implementation) is considered under a 20-year timeframe (i.e., n equals 20).

NPV accounts for any initial investment cost involved in the implementation of a project or investment decision that impacts future cashflows. The formula for NPV is as follows (Ross et al., 2003):

$$NPV = \sum_{t+1}^{n} \frac{C_t}{(1+r)^t} - I_0 \tag{4.2}$$

where I_0 is the initial investment cost. If a project does not require an initial investment, then the value of I_0 is 0. For instance, the addition of a perennial forage crop to annual crop rotations is an "investment" for which the full benefits and costs are incorporated into future cashflows and there is no initial investment cost. Several past studies have analyzed the feasibility of BMP implementation in a NPV framework, including Trautman (2012) and Xie (2014).

A NPV can be converted to an annual benefit or cost using the following amortization formula:

$$A = NPV\left[\frac{r}{1-\frac{1}{(1+r)^n}}\right] \tag{4.3}$$

where A is an annualized value representing the net benefit (or cost) per year. The formula is utilized in the calculation of annual economic benefits from each of the alternative BMP scenarios.

4.1.3.3 Choosing a Discount Rate

A discount rate is used to convert future cashflows to their present value. The choice of discount reflects several factors, including the time value of money, the riskiness of the project or investment, and the opportunity cost of the allocated capital. The time value of money involves an assumption regarding investor preferences: earlier returns (positive cashflows) are more preferable to later returns, all things being equal. The discounting done in NPV calculations therefore is done as a method to evaluate cashflows associated with an investment on an equivalent time basis (Ross et al., 2003). The relative riskiness of an investment should also be reflected in the discount rate (Copeland and Weston, 1988). When risk is high, the adoption of a project will not occur unless the expected return is high enough to compensate the decision-maker. As such, a risk premium is often added to the choice of discount rate. Lastly, the discount rate should reflect the market-determined rate of return of the best alternative opportunity for using the initial capital outlay (Ross et al., 2003).

One approach to determining the appropriate discount rate is to use a rate similar to those used in the evaluation of projects with a similar level of risk. For example, Xie (2014) studied the private impacts of adopting BMPs for a representative southern Alberta irrigated farm. The discount rate used was 10%. This rate was also used by Koeckhoven (2008), who evaluated BMP adoption on a representative dryland farm in the LLB watershed region. Another approach that may be used is the theory of Capital Market Line (CML), where the unique risk of an investment can measured and taken into account. The discount rate, or required rate of return, for an investment depends on a risk-free rate of return (commonly represented by a government bond), expected market return (commonly represented by the return on an index of stocks, such as the Toronto Stock Exchange), the standard deviation of the market portfolio, and standard deviation of returns or cash flows for the investment (Sharpe et al., 2000). Cortus (2005) utilized this approach to determine the discount rate for grain production as an investment on a representative farm in Saskatchewan. The calculated discount rate was 13.91%, which Cortus (2005) considered to be a maximum given other discount rates used for similar projects (e.g., Miller, 2002). Although discount rates in livestock production are often higher, Cortus (2005) settled on a rate of 10%, arguing that crop production is inherently less risky than livestock production due to a greater diversification of products being produced (multiple crops in a single year). Given this information, and because a diverse array of crop-related production is being considered in this study, the discount rate chosen for the NPV analysis is 10%.

4.1.4 Nutrient Balance Budgeting

In this analysis, the calculation of private economic impacts through a capital budgeting technique is paired with the construction of watershed-level nutrient balances in order to evaluate the tradeoffs between economic returns and environmental quality. Constructing nutrient balance budgets for agricultural systems is a method that can be used to provide a quantitative estimate of excess nutrients at a given site (e.g., a field, farm, or watershed). This information can be used to identify areas of potential environmental concern and to evaluate alternative nutrient management practices (Meisinger and Randall, 1991).

The theory behind this method is based upon the general conservation of mass equation for a soil-crop system: the change in total nutrient level for a system (e.g., P in the soil of a particular field) is equal to the difference between the sum of inputs (imports) and the sum of outputs (exports) of the particular nutrient. A nutrient surplus, where inputs exceed outputs, may indicate a potential risk to the environment; conversely, a nutrient deficit may signal to a producer that system productivity is being adversely affected by a shortage of required nutrients. For instance, excess N inputs can lead to the buildup of highly soluble NO_3^- in soil, which is at risk of leaching into groundwater and negatively impacting surrounding water quality (e.g., Rodvang et al., 1998). However, a lack of sufficient N in the soil can result in decreased crop yields. The goal of the system manager is therefore to achieve a balance of nutrient inputs and outputs over the long-term.

Meisinger and Randall (1991) defined the 'long-term potentially leachable total N' (LPLN) as the difference between total N inputs and total N outputs, minus any changes in soil N storage through other natural processes (e.g., denitrification). The term LPLN is used because it emphasizes how an N surplus *could* leach or be removed via runoff in drainage events and thus may *potentially* impact water quality (Meisinger and Randall, 1991). The same is true of P, although different chemical interactions with the soil will influence the rate of P loss and thus risk to water quality. In essence, calculation of nutrient balance provides the manager (or researcher) with a proxy of potential risk to water resources. The extent that a nutrient surplus will impact water quality depends on local climatic and hydrologic conditions; for instance, in dry areas (like the LLB watershed), LPLN may not leach into groundwater for many years. However, the buildup of nutrients in the soil over time still presents an environmental risk and even organic factions will mineralize over time and eventually become mobile in the soil system.

The calculation of a net nutrient balance requires the estimation of both inputs into and outputs out of the system. Meisinger and Randall (1991) divided the flow of nutrients (both inputs and outputs) into primary and secondary processes, based upon the magnitude and proportion of total nutrient level in the system.[11] Primary processes are those that are either large in magnitude (e.g., addition of manure) or constitute a relatively major proportion of the total nutrient flow. Meisinger and Randall (1991) list the following as primary N input processes: chemical fertilizer, manure application, biological N_2 fixation, and irrigation water. Secondary processes include atmospheric deposition (dry), crop seed, and non-symbiotic N_2 fixation. Major inputs of P include

[11] The dual classification is necessary due to the diversity in agricultural systems. For example, in a system with low levels of inputs and outputs, such as native grassland used for pasture, N import via precipitation can be a proportionately significant source.

only chemical fertilizer and manure application. The primary output of both N and P is harvested crop material, although certain secondary processes such as weight gain in livestock and losses via natural pathways such as denitrification and ammonia volatilization (in the case of N) may be significant depending on the system.

This method has been used for the evaluation of a range of agricultural systems, from specific fields units up to the watershed level. Karlen et al (1995) used individual field-level analysis to determine differences in nutrient balance from various management practices on two different fields. Koelsch (2005) assessed the overall nutrient balance of a farm-level livestock operation, which included a 2500-head cattle feedlot and three separate fields grown to various livestock feed (corn, alfalfa, etc.). Barry et al (1993) also used a whole-farm nutrient budget, with the aim of estimating the effect of different farming practices on NO_3^- concentrations in groundwater. Other studies (e.g., Nord and Lanyon, 2003) scale up the analysis to the watershed level because impacts of individual farm nutrient management decisions are most readily observable at a more aggregated level. Yang et al (2011) go even further, estimating an annual N budget for Chernozemic soils in Canada (primarily in the prairie regions of Alberta, Saskatchewan, and Manitoba) using census level agricultural data.

Given that water quality in the LLB watershed region is one of the primary environmental issues of concern in the current study, estimating nutrient balances is an appropriate approach to assess how agricultural activity may impact this particular environmental parameter. Buczko et al (2010) assessed the predictive quality of simple N loss indicator approaches, including the calculation of N balance, the exchange frequency of soil solution, and potential NO_3^- concentration in leachate. Using both field and published data of NO_3^- leaching across sites in both Europe and North America, the authors determined that N balance is a relatively superior indicator of N loss over the long-term (but poor over a one-year timeframe). In situations where a complete accounting of the N cycle via direct measurement is not possible, calculation of N balance provides a reasonable indication of N loss risk for certain systems over longer assessment periods (Buczko et al., 2010). As such, nutrient balances of both N and P are calculated for each land use scenario in this analysis.

For this analysis, chemical fertilizer and manure application were considered as major inputs of both N and P, with biological fixation an additional major input of N. Following Bremer et al (2008), N contribution from irrigation water was assumed to be negligible, but the import of N via atmospheric deposition was considered due to the high density of livestock operations in the area (Hao et al., 2006). The export pathways considered for both N and P were removals from harvested crop materials and animal weight gain on fields devoted to pasture activities. The latter was considered because on pastureland, where nutrient additions and removals are relatively low, removal via animal weight gain makes up a proportionately significant part of the overall nutrient flux. A detailed description of the calculation strategy for each pathway is provided in Chapter 5, sections 5.6.1 and 5.6.2.

Given the lack of precise on-site measurements, the estimation of nutrient balance is subject to a range of uncertainty. For instance, nutrient input from manure is difficult to estimate due to variation in N composition, uncertainty in loading rates, spatial variability of application, and losses after excretion (Meisinger and Randall, 1991). Without specific manure analysis, reliance on published nutrient content values ('book values') will inevitably produce some error. Meisinger and Randall (1991) provide a range of uncertainties for individual input and output components. As discussed in Chapter 5, uncertainty in nutrient balance calculations is incorporated for N, to a limited degree.

4.1.5 Estimation of Changes to Soil Organic Carbon Storage

In addition to water quality, the depletion of soil organic carbon (SOC), or soil organic matter (SOM), is another key environmental issue of interest in this study. SOM is an important component of soil health, and influences a number of important aspects of soil quality (e.g., stabilization of soil structure, nutrient provision). Agricultural soils in Canada have lost about 25-35% of their stored C since land was first being brought into cultivation, a process which has contributed to the buildup of greenhouse gases in the atmosphere (Smith et al., 2001). However, these losses can be reversed, and in their current state most agricultural soils now have a significant C sink capacity (Paustian et al., 1997; Janzen et al., 1998). Because the selection of agricultural practices has traditionally been motivated by a desire to increase yields, a number of potentially beneficial alternatives have been ignored. These alternative practices have garnered attention in recent years as ways to enhance the capacity of soils (and the grassland ecosystem more broadly) to store C and contribute to both local soil health and mitigation of global climate change. For instance, Lal (2004) estimated that as much as one-third of the annual increase in atmospheric CO_2 can be offset by targeted agricultural land management. In Canada, the practices most commonly identified by research as having the greatest potential to contribute to soil C accumulation are the reduction or elimination of summerfallow, conversion of annual cropland to either native grassland or perennial cropping, inclusion of perennial forages in annual crop rotations, and adoption of reduced or no tillage practices (Vandenbygaart et al., 2008). Other practices, such as fertilizer application, irrigation, manure management, and use of cover and green manure crops are generally thought to have some influence on soil C, but the effects are difficult to measure and sufficient research does not exist to produce estimates for Canadian agricultural land (Vandenbygaart et al., 2008).

4.1.5.1 Literature Estimates and Modeling Approaches

Smith et al (2001) were among the first researchers to study management practices useful in enhancing the storage of SOC in agricultural soils in Canada. They used the CENTURY model (Parton et al., 1987) to simulate changes in agricultural management across a 70-year time period and then looked at changes in C flux in soils, calculating soil C sequestration coefficients (in Mg C ha^{-1} yr^{-1}) for a set of management practices. These included conversion to permanent cover (grass), addition of forage, no tillage, minimum tillage, addition of a cereal to a cereal-fallow rotation, removal of fallow periods, and several different fertilization strategies.

Boehm et al (2004) went on to use the C sequestration coefficients developed by Smith et al (2001) to measure the total sink potential of Canadian agricultural soils from the incorporation of zero tillage, reduction of summerfallow, and conversions to permanent cover. The coefficients derived by the CENTURY model were compared to empirical estimates of SOC change from studies across Canada in a compendium gathered by Vandenbygaart et al (2003). The CENTURY model proved to be an accurate, albeit conservative, estimator of SOC changes.

Vandenbygaart et al (2008) also utilized the CENTURY model to estimate changes in SOC after the implementation of a land use change. The authors used model simulations to develop a generalized set of SOC storage factors unique to each land management practice, agricultural reporting zone (within Canada), and soil texture. They then developed a series of empirical equations to fit these factors, which can be used to predict SOC changes over a specified period of time for a particular land management practice and set of biophysical conditions. Because SOC dynamics are governed by first-order kinetics, where the rate of soil C gain or loss following a land use change decreases over time, the equations assume an exponential functional form. The Canadian government's National Inventory Report 1990-2009 on greenhouse gas sources and sinks (ECCC, 2009) utilizes this methodology, which was recently updated and formalized by McConkey et al (2014).

A different model was proposed by Campbell et al (2000). Using a 30-year crop rotation experiment on a Brown Chernozem in Saskatchewan, the researchers studied the influence of cropping frequency, fertilizers, and type of crop on SOC. From measurements of SOC in the soil, they developed an empirical equation to estimate SOC dynamics in the rotations, taking into account crop residue decomposition, C mineralization, and crop yields. In this equation, SOC is primarily a function of organic C additions as plant residue, which can be estimated as a function of crop yield, harvest index, and the straw/root ratio.

This empirical model was tested against the CENTURY model in two separate studies (Campbell et al, 2007a; Campbell et al., 2007b). The authors showed how grain yields can be used, together with coefficients of conversion of C inputs from crop residues to SOC, to estimate changes in SOC over time and across different treatments (minimum tillage, conventional tillage) of cropping systems. The authors concluded that when crop yields are known, this empirical model can be more accurate in predicting SOC dynamics than the CENTURY model in the semi-arid prairies.

The above Campbell et al studies (2000; 2007a; 2007b) only investigated the SOC dynamics in rotations featuring cereal crops. Gan et al (2009) determined carbon allocation coefficients for common oilseed (canola and flax) and pulse (field pea and lentil) crops in grain, straw, roots, and rhizodeposits. These values are important tools for modeling and quantifying SOC dynamics and CO_2 sequestration in more diverse cropping systems. A range of moisture regimes were examined to produce robust values.

Bolinder et al (2007) proposed a method for predicting SOC dynamics based upon estimates of net primary productivity (NPP) and annual C inputs to the soil for various Canadian agro-ecosystems. The method for estimating C accumulation and distribution in crop plants involves summing the C added in the agricultural product, post-harvest residue, root tissue, and extra-root material. The C contained in the latter three elements is assumed to be added to the soil. Bolinder et al (2007) report C allocation coefficients for many different systems (including perennial forages, grass species, legumes, and native grassland), as well as the relative proportion of C returned to the soil.

The calculation of SOC sequestration values in this study is performed using the methods outlined in McConkey et al (2014) and used in ECCC (2009). Of the four primary land management changes considered in McConkey et al (2014), three are relevant to the analysis of the LLB watershed: reduction of summerfallow practices, introduction of perennial crops to annual crop rotations, and conversion to permanent cover with perennial vegetation. Each of these land management changes are modeled within one or several of the BMP watershed scenarios constructed. The reduction or elimination of tillage practices is not considered. A detailed description of the methods used is provided in section 5.6.3.

4.2 Chapter Summary and Introduction to Watershed Scenarios

This chapter discusses a variety of techniques used to model the economic and biophysical elements of agricultural systems. The goal of this analysis is to evaluate the watershed-level economic and environmental outcomes arising from the implementation of various BMPs in the LLB watershed. A comparison of these two sets of outcomes will help to illuminate the tradeoffs inherent in land use decision-making. A capital budgeting technique, specifically NPV analysis, is selected as the most appropriate method to measure the watershed-level economic impacts of BMP implementation (i.e., investment). To capture the watershed-wide environmental impacts of BMP implementation, two additional modelling approaches are used in conjunction with NPV analysis: nutrient (N and P) balance budgets and the estimation of SOC changes using published estimates of carbon change factors in agricultural soils. The former allows for the evaluation of risk to water sources from nutrient leaching or runoff. The estimation of SOC changes allow for the calculation of public benefits in terms of increased carbon sequestration.

These methodological approaches are brought together via the development of watershed-level land use scenarios. A baseline scenario is constructed to be representative of current agricultural land use and management practices in the LLB watershed. A suite of BMP scenarios are also built and evaluated for their impact on economic returns and environmental performance. These scenarios feature the BMPs discussed in Chapter 2, which are designed to improve either water quality (through the reduction of excess nutrient inputs) or rates of SOC storage. Each watershed scenario is evaluated over a 20 year period. Total cashflow, involving the calculation of both revenues and costs from all agricultural activity in the watershed, is assessed for each year and discounted using the NPV framework to obtain a PV of total economic activity. In conjunction, nutrient balance budgets of both N and P are constructed for each scenario to evaluate the impacts of different land uses and management practices on residual soil nutrient levels. Finally, where a scenario involves the implementation of one of the three land use changes deemed to impact SOC levels (e.g., reduction of summerfallow), the resulting change in SOC storage is calculated and aggregated across the watershed. Table 4.1 lists the scenarios and the BMPs evaluated in each.

Table 4.1. List of Watershed Scenarios and Corresponding BMPs for Evaluation.

	BMP Evaluated / Change in Land Use
Scenario 1a	Introduction of alfalfa to crop rotations on all cropped fields
Scenario 1b	Introduction of alfalfa to crop rotations on dryland cropped fields
Scenario 2a	Introduction of legume green manure (irrigated) and field peas (dryland)
Scenario 2b	Introduction of legume green manure crop
Scenario 2c	Introduction of legume green manure crop and field peas
Scenario 3	Manure management
Scenario 4a	Conversion of all cropland to permanent forage
Scenario 4b	Conversion of only dryland cropland to permanent forage
Scenario 4c	Conversion of only marginal dryland cropland to permanent forage
Scenario 5a	Conversion of all pastureland to annual cropping
Scenario 5b	Conversion of all pastureland to annual cropping, with alfalfa
Scenario 5c	Conversion of all pastureland to annual cropping, with alfalfa and manure

Chapter 5: Study Methods

Agricultural activity within the Lower Little Bow watershed has a unique set of past production decisions, biophysical limitations, and economic dynamics that impact future environmental and economic outcomes. When modeling watershed-level outcomes for the LLB watershed, it is important to take this set of characteristics into account. The following sections within this chapter describe the procedure employed to categorize agricultural fields of the watershed, establish the various land use scenarios, estimate the relevant economic parameters, and determine resulting environmental impacts.

5.1. Classification of Fields in the Lower Little Bow Watershed

A survey of agricultural producers was undertaken as part of AAFC's WEBs (Watershed Evaluation of Best Management Practices) project in order to obtain information on past and typical land uses. The information elicited by the questionnaire primarily included typical crop rotations, yields, fertilizer use, manure applications, and stocking rates for land under production within the LLB. The questionnaire was administered by the County of Lethbridge soil conservation technologist via phone or in-person interviews in March and April of 2007 with producers in the study region.

A total of 60 quarter sections were identified as being used for agricultural purposes within the study area. However, the acreage used for production varies across the quarter sections, ranging from 40 to 160 acres (Table 5.1). As such, each productive quarter section will be referred to as 'fields' in the analysis and the corresponding acreage of each individual field will be reflected in the management decisions and calculated returns from production. Additionally, several quarter sections were split into two units to reflect different uses in different areas. Therefore, there are a total of 65 fields that are used as units of analysis in the current study. Refer to Appendix A for detailed information regarding each of the fields, and section 3.2.2 for a discussion of typical practices.

5.1.1. Land Uses of the Lower Little Bow Watershed

A wide range of agricultural production systems are used in the LLB. Of the 6,680 acres of agricultural land, irrigated annual cropping systems account for the largest portion with 2,720 acres (41%) making up 24 fields. Dryland annual cropping is employed on 1,320 acres (20%), comprising 15 individual fields. Tame and native pastureland used for livestock grazing (primarily cow-calf operations) also constitutes a significant share, with 2,280 (34%) acres over 21 fields. Irrigated fields used for perennial vegetation (e.g., bromegrass, timothy hay, or mixed alfalfa-grass) account for 360 acres (5%) across five individual fields.

Further distinction can be made in categorizing fields used for pasture. In addition to dryland and irrigated, pastureland can be broken down into two types: tame and native pasture. This distinction is significant due to the differences in recommended stocking rates and relative productivity. Native pasture, which has not been seeded or altered in any way, typically grows more slowly than tame pasture (AAF, 2005). Examples of native species used for grazing in southern Alberta include vetch, hairy wild rye, rough fescue, and native wheat grasses (AEP, 2009). Tame pasture, however, consists of vegetative species not native to the area that are purposefully seeded. Examples of these introduced species include clovers, red fescue, kentucky blue grass, bromegrass, and timothy grass. Native range occupies 2,000 acres (30%) within the LLB watershed and tame pasture just 640 acres (9.5%), of which 360 acres are irrigated.

Another distinction to be made among pastureland is whether it is located on riparian versus upland area. Riparian areas tend to be more productive than upland areas, producing a greater amount of forage due to the higher water table (Fitch and Adams, 1998; Bork et al., 2001). Inspection of aerial photographs of the watershed resulted in the categorization of 920 acres of pastureland as riparian with 80 and 840 acres of that being tame and native pasture, respectively. The assumption was made that fields directly adjacent (bordering) the Lower Little Bow River are considered riparian. This is supported by the digital elevation model (DEM) used in Miller et al (2008a) which revealed that the lowest areas of the watershed are adjacent to the river, and that surface flow was generally directed that way.

Table 5.1 summarizes the acreage of land use categories ascribed to the fields of the LLB watershed, based on the producer survey and inspection of aerial photographs. For purposes of this analysis, irrigation infrastructure is assumed to remain fixed; that is, removal or addition of irrigation from individual fields is not included in any of the proposed scenarios.

Table 5.1. Land Use Categories and Corresponding Acreage of the Lower Little Bow Watershed.

	Activity/Use	Area (Acres)	Number of Fields	Percent of Total Area
Irrigated	Annual Cropping	2,720	24	41%
	Tame Upland Pasture	280	4	4%
	Tame Riparian Pasture	80	1	1%
	Total Irrigated	3,080	29	46%
Dryland	Annual Cropping	1,320	15	20%
	Tame Upland Pasture	280	3	4%
	Native Upland Pasture	1,160	12	17%
	Native Riparian Pasture	840	6	13%
	Total Dryland	3,600	36	54%
	Total	6,680	65	100%

5.1.2. Soil and Topographic Characteristics

Agricultural productivity and land use is influenced by various soil properties and topographic characteristics of the landscape (Noorkabhsh et al., 2008). Soil properties and landscape characteristics were obtained for each of the 65 fields of the LLB watershed using the Agricultural Region of Alberta Soil Inventory Database (AGRASID).[12] The AGRASID is a spatial inventory of soil landscape polygons within the Alberta white zone (Alberta Soil Information Centre, 2016). Certain attributes, such

[12] The AGRASID was accessed via the online Alberta soil information viewer (http://www4.agric.gov.ab.ca/agrasidviewer/) in November, 2014.

as drainage, landform profile, and soil series are reported. The properties deemed to be useful for the purposes of this analysis and consequently extracted from the database included type of soil, land suitability rating, and the presence of various limitations (primarily moisture or slope) on agricultural activities.

In many cases, multiple soil polygons will overlap within an identified field in the study area. In these cases the dominant soil polygon, defined as the polygon which covers the majority of the area of the field, is chosen as representative of the field's soil properties. Multiple soil polygons were found in 27 of the 65 fields (42%).

Soil within the LLB watershed is predominantly Orthic Dark Brown Chernozemic (55%), which includes the Lethbridge, Readymade, and Whitney soil series. Other classifications included ZUN (miscellaneous undifferentiated mineral soil), Orthic Dark Brown with Regosolic profiles, and Orthic Brown Chernozemic. For each soil polygon, the AGASID provides the land suitability rating system (LSRS) value for the production of spring-seeded small grains (e.g., wheat, barley, oats). The LSRS for agricultural crops rates individual components of land productivity (climate, soil, landform) separately and under explicitly defined conditions. Based on this framework, an area of land can be placed into one of seven basic classes to reflect its potential for crop production. Class 1 indicates areas that are most suitable for cropping, and each subsequent class reflects a higher degree of limitation and a lower score along the LSRS.[13]

The predominant land suitability rating of fields in the LLB watershed were either Class 4 or Class 5. According to the AIWG (1995), Class 4 lands are those defined as having "severe limitations that restrict the growth of specified crops" and "are marginal for sustained production of specified crops". Class 5 lands are those defined as having "very severe limitations for sustained production" and "annual cultivation using common cropping practices is not recommended".

Of the 65 fields in the LLB watershed, 37 (57%) were rated Class 4 and 28 (43%) Class 5. In some cases, the rating value may include more than one class in a polygon, depending on the soils present. For instance, an LSRS rating of 4(8)-5(2) indicates that 80% of the land in the polygon is classified as Class 4 and 20% as Class 5. In these cases,

[13] Given the parameters and criteria of the LSRS, there is no Class 1 land in Alberta.

the dominant rating (Class 4 in this example) was used as the overall class for the field in the analysis.

A second component of the LSRS are the subclasses assigned to an area of land based on the type of limitation. The basic subclasses are climate, soil, and landscape. Within the climate subclass, temperature and moisture limitations are two specific designations. A number of different types of limitations may be specified for the soil subclass, including water holding capacity/texture, soil structure, organic matter, depth of topsoil, soil reaction, salinity, sodicity, organic surface, drainage, rock, degree of decomposition or fibre content, and depth and substrate. Several specific limitations may be implicated in the landscape subclass, including slope, landscape pattern, stoniness or coarse fragments, wood content, or inundation.

Fields of the LLB were primarily limited by two specific conditions: water holding capacity/texture (denoted 'M', within the soil subclass) and slope (denoted 'T', within the landscape subclass). All 65 fields were within soil polygons featuring a subclass M limitation, highlighting the semi-arid climate of the LLB watershed region. However, only 39 fields were classified as possessing a subclass T limitation, thereby further providing a potentially useful means of grouping fields according to productivity potential (discussed further in section 5.2.5.3). Of these 39 fields (3,760 acres, 56% of the total acreage), 19 were utilized for cropping activity between 2006-2011 according to the producer survey.

Soil sampling was undertaken in July 2006 as part of the WEBs project. A total of 251 samples were obtained across the watershed, with three or four samples from each field. From each sample, a % clay, sand, and silt content was determined and a textural class was established, as per Figure 5.1. Dominant textural classes were loam, sandy loam, and sandy clay loam. For purposes of this analysis, the textural class was translated to a textural category of either 'Coarse' or 'Medium' based on Kryzanowski et al (1988) (Table 5.2). The distinction between coarse and medium is relevant vis-à-vis certain management practices, such as the recommended amount of fertilizer application. A total of 3,960 acres, corresponding to 38 fields, were classified as having 'Coarse' textured soil; conversely, 2,720 acres, over 27 fields, were classified as having 'Medium' textured soil. The relevant biophysical properties of each field, as well as available information on typical yields, crops grown in the past, fertilizer practice, and manure application from the producer survey, is reported in Appendix A.

Table 5.2 Soil Texture Category Based on Textural Class.

Texture Category	**Textural Classes**
Very Coarse	Sands, Loamy Sands
Coarse	Sandy Loam, Fine Sandy Loam
Medium	Loam, Sandy Clay Loam, Sandy Clay, Clay Loam
Fine	Silt Loam, Silty Clay, Silt
Very Fine	Clay, Heavy Clay

Source: Kryzanowski et al (1988).

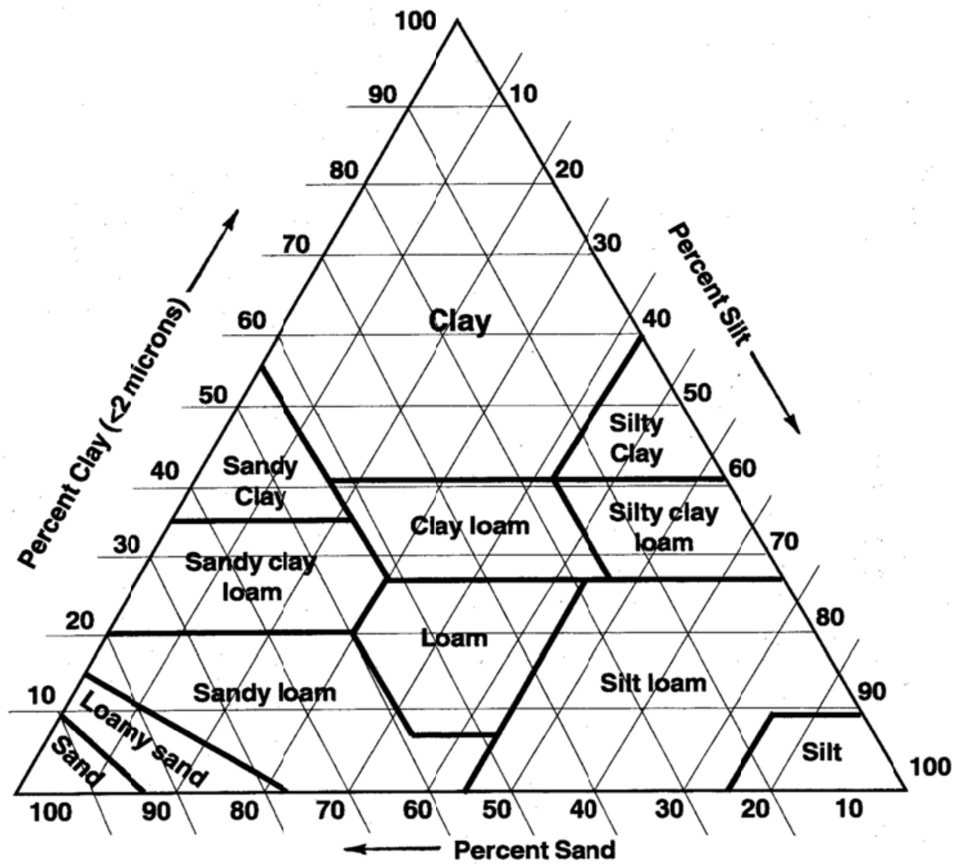


Figure 5.1. Soil Textural Triangle and Textural Classes. (source: Plant and Soil Sciences eLibrary http://passel.unl.edu/pages/informationmodule.php?idinformationmodule=1130447039&topicorder=2&maxto=10. Last Accessed February 4, 2016.)

5.2. Lower Little Bow Watershed Land Use Scenarios

Evaluation of tradeoffs between economic returns from agriculture and certain metrics of environmental quality is done by modelling watershed-level land use scenarios featuring the implementation of various BMPs. There are a total of twelve distinct land use scenarios modelled, embedded within five scenario classes that each feature a different possible BMP for the area. The BMPs investigated as part of this analysis include the addition of a perennial legume forage to annual crop rotations, the use of legume green manures, the inclusion of field peas in annual crop rotations, the implementation of a livestock manure management BMP on cropped fields, and the conversion from annual cropping to permanent cover using perennial forage crops. As a counterpoint, the conversion of pastureland to annual cropland is also investigated as an intensification of land use possibility. The economic and environmental outcomes of each of the alternative land use (BMP) scenarios are compared to a baseline scenario of land use in the LLB watershed.

5.2.1 The Baseline Scenarios

For the purposes of this analysis, two baseline scenarios were constructed in order to compare economic and environmental outcomes arising from the adoption of various BMPs modeled in the alternative land use scenarios. Both scenarios are intended to emulate the current and projected future land use found in the LLB watershed and as such, feature the same allocation of land use as presented in Table 5.1. Each of the 65 fields are assigned to that specific corresponding activity (i.e., irrigated annual cropping, dryland annual cropping, tame grass used for pasture, etc.).

One baseline scenario was developed to represent 'business as usual' (BAU) agricultural practices specific to the LLB watershed. This baseline incorporates information from the producer survey with respect to typical manure application practices. A significant proportion of cropland in the LLB watershed receives regular manure application for the purposes of crop fertilization. This is not surprising given the density of livestock and animal operations in the area (see Chapter 3). Given the significant impact that manure application has on nutrient availability and balance, it is important to account for this practice. However, the application of manure to meet crop nutrient requirements will result in a higher total level of nutrients added to the agro-environmental system than if chemical fertilizer were used instead. Several factors contribute to this outcome, including the ratio of N and P in manure, mineralization rate of organic nutrients, and the uncertainty of manure nutrient composition without laboratory analysis (compared to chemical fertilizer).[14] As such, nutrient inputs derived from manure will have an outsized impact on the calculation of nutrient balances when compared to chemical fertilizer inputs. Section 5.2.1.2 provides further details regarding the specification of this practice. This scenario, termed the 'BAU Baseline Scenario', is used as the baseline case in the evaluation of Scenario 3 (featuring the manure management BMP) and Scenario 5c (pastureland conversion with typical with manure application practices).

The primary baseline scenario, termed the 'Baseline Scenario', is used as the baseline case for comparing to all other BMP scenarios. Crop nutrient requirements are met only with chemical fertilizer applications and manure is not used as a nutrient source. This specification allows for a more precise evaluation of BMP impacts to total nutrient balance. Other than Scenarios 3 and 5c (to be discussed in more detail in the following sections), manure application is not present in any of the BMP scenarios. Therefore, comparison of nutrient balance outcomes between a BMP scenario and the BAU Baseline (which includes manure) would encompass both manure and non-manure related effects. In order to accurately isolate the impacts of specific BMPs, the level of manure must remain constant. As such, the Baseline Scenario, featuring chemical fertilizer as the sole means of meeting crop nutrient requirements, is used as the reference case for BMP scenarios without manure. Another option would be to keep the total amount of manure applied in each BMP scenario the same as the BAU Baseline, which is the strategy employed for scenario 5c.

5.2.1.1 Baseline Crop Rotations

Several sources of information were used in the development of the baseline crop rotations, which are used in both versions of the baseline scenario. A total of 4,040 acres

[14] A more complete discussion of these factors, and their implications, is available in Chapter 2 and in section 5.2.4 where the manure application BMP is presented.

are devoted to annual cropping in the baseline scenarios, 2,720 acres of which are irrigated and 1,320 acres dryland.

First, the producer survey was used to establish the typical mix of crops grown in the LLB watershed as well as provide an indication regarding typical fertilizer and manure application practices. The results and key findings were discussed and summarized in Chapter 3. Barley was the most common crop grown in the study area at the time of the survey, followed by canola and wheat.

The 2011 Census of Agriculture (Statistics Canada, 2011) was also used to determine the acreages of major crops in Lethbridge County. The baseline crop rotations were also designed to be representative of the major crops planted in the county based on this information. The most prevalent crops, in terms of both total acreage and number of farms, were wheat, barley, and canola.

Finally, past work on the economics of BMP adoption in Alberta was also used to inform the development of representative crop rotations. Trautman (2012) and Xie (2014) both modeled representative southern Alberta cropping operations to assess the private costs of BMP adoption. Koeckhoven (2008) modeled BMP adoption for a mixed dryland cropping and livestock farm in the LLB region. The crops chosen to be part of the representative rotations in all three studies were generally based upon i) expert opinion, ii) agronomic factors, and iii) the most recent Census of Agriculture data available at the time.

To represent the Dark Brown soil zone, Trautman (2012) modeled a typical dryland cropping operation based in Starland County. In addition to sharing the same soil zone, Starland county is geographically close to Lethbridge county. In 2006, spring wheat, barley, and canola were the most common (in terms of acreage) crops grown in Starland County, along with a significant portion allocated to summerfallow. Expert opinion consulted for the study (Bergstrom, 2009) indicated that typical crop rotation in this area was comprised of approximately one-third wheat, one-third canola, and the remaining one-third a combination of barley, field peas, specialty crops (e.g., potatoes, sugar beets, dry beans), and silage or forages. Farms also typically alternate cereals and broadleaf annuals from to year to year within individual fields for disease and pest control reasons.

The base cropping rotation used by Trautman (2012) for this farm was a four year rotation consisting of Spring Wheat – Canola – Barley – Summerfallow.

Trautman (2012) also modeled a typical irrigated operation for the Brown soil zone. The author chose Taber Municipal District (M.D.), which is immediately adjacent to Lethbridge County, as the representative region. While there is typically less moisture present in the Brown zone (AAF, 2005a) than for Dark Brown soils, this difference is minimized when modeling irrigated operations since the crop water needs are assumed to be satisfied. In 2006, spring wheat, barley, durum wheat, canola, and potatoes were the top five crops in Taber M.D. by acreage. The base cropping rotation used by Trautman (2012) for the Brown (irrigated) soil zone was Spring Wheat – Canola – Durum Wheat – Dry Beans.

Xie (2014) modeled a representative commercial irrigated crop farm in southern Alberta. To build base crop rotations, Xie (2014) used census data from the Taber and St. Mary Irrigation districts, combined with expert opinion. In these districts, the most common irrigated crops are barley, canola, dry beans, durum wheat, red spring wheat, sugar beets, and potatoes. Three different base rotations were developed in order to respect certain agronomic constraints. For example, sugar beets and canola are generally not grown in rotation with one another due to agronomic concerns (Dunn, 2011; Smith, 2011). The three representative rotations used in the analysis were four year rotations of Potato – Spring Wheat – Sugar Beet – Dry Bean, Spring Wheat – Canola – Durum Wheat – Dry Bean, and Potato – Spring Wheat – Canola – Cereal (one of Spring Wheat, Durum Wheat, or Barley). The cereal chosen for the fourth year in the latter canola rotation was based on the highest expected gross margin calculated in the simulation analysis.

Koeckhoven (2008) did not define an explicit crop rotation in terms of which annual crops are planted on specific parcels of land in any given year of the modeling timeframe. However, the model used did allocate certain acreages to crops within the representative farm. Implicitly, in terms of annual crop production, a crop rotation of durum wheat, spring wheat, barley, and canola was assumed.

The final source of information providing input into the baseline crop rotations to be used for this analysis comes from empirical studies done in the area. Little et al (2003), in their analysis of surface water quality of the LLB River, mapped land use in six sub-basins of the LLB watershed. Cereals made up the largest percentage of land use devoted to dryland annual cultivation, ranging from 9.70% to 73% of the sub-basin areas. The proportion of native grassland ranged from 3.6% to 73.9%. This particular land use was generally inversely related to the area of the sub-basin in irrigated crop production. Relatively small percentages of land in the watershed were devoted to canola, sugar beet, potato, and forage production. Area in fallow roughly equated to one-fifth of the area used for dryland cropping in each sub-basin, suggesting that the practice of summerfallow was used in rotation approximately every five years. Rodvang et al (2004) and Rodvang et al (1998) also used Lethbridge County as a study area, and found a similar mix of crops grown. Rahbeh et al (2011), studying certain hydrologic facets of the LLB watershed, found a disproportionate area of both dry and irrigated cropping was devoted to barley. A significant portion of barley grown was used for silage.

Taking into account the above sources of information, three representative crop rotations were constructed: one for dryland cropping, and two for irrigated. The base rotation for dryland cropping consists of wheat, canola, barley, and summerfallow. Both the census data (Statistics Canada, 2011) and the producer survey indicated that a significant portion of rainfed cropland was allocated to these uses. The rotation is identical to the one modeled by Trautman (2012, Dark Brown soil zone) and Koeckhoven (2008). The four year rotation is as follows:

Dryland Rotation:
Spring Wheat – Canola – Barley – Summerfallow

Two different rotations were modeled for fields used for irrigated cropping. Irrigated crop rotations are diverse due to the increased variety of options available to crop producers when the moisture constraint is lifted. Attention must therefore be paid to the sequencing and mix of crops in the rotation, particularly if “specialty” crops (e.g., sugar beets or potatoes) are grown, which are generally of higher value. For instance, canola and sugar beets should not be grown in a rotation sequence due to disease issues. For these reasons, it was determined that more than one representative rotation was required for purposes of modeling irrigated cropping in the LLB watershed.

The first base irrigated rotation features canola alongside two varieties of wheat and dry beans. Xie (2014) modeled two different canola rotations, one with potatoes included and the other not. Because potatoes are included in the second base rotation and are generally not common in the LLB watershed, the latter rotation is adopted. The rotation is a four year sequence as follows:

Irrigated 1 Rotation:
Spring Wheat – Canola – Durum Wheat – Dry Beans

This is similar to Xie (2014), except that durum wheat is chosen explicitly for the third year. According to Dunn (2009), in southern Alberta, dry beans are typically only grown under irrigated conditions. When modeling an irrigated farm in the Brown soil zone, Trautman (2012) also used dry beans as part of a four year base rotation.

The second baseline irrigated rotation modeled in this study features two specialty crops: sugar beets and potatoes. While the producer survey did not indicate that potatoes had previously been grown in the LLB watershed, and that sugar beets were allocated only a very small acreage, it was deemed appropriate that both specialty crops would be included in order to encompass the range of crop possibilities representative of irrigated cropping in southern Alberta. The rotation is a four year sequence as follows:

Irrigated 2 Rotation:
Potato – Spring Wheat – Sugar Beets – Barley

This is similar to the sugar beet rotation modeled by Xie (2014), except that barley replaces dry beans in the fourth year. Although Trautman (2012) found that barley was not as commonly grown using irrigation (Dunn, 2009), this decision was made because the survey of producers in the study area indicated that barley has historically been a popular crop for both rainfed and irrigated fields. Miller (2014) confirmed this finding, and also suggested that the use of barley as silage was popular due to the prevalence of feedlots in the area and the marketability of silage as a feedstock. Also, according to the 2011 Census of Agriculture, dry beans did not make up a large enough acreage in Lethbridge County to justify inclusion in both irrigated base rotations.

5.2.1.2 Manure Application

Manure application on certain fields was modeled in the BAU Baseline Scenario. Of the 65 agricultural fields in the study area, 30 have a history of regular manure application. Therefore, it is important to account for the environmental and economic impacts of typical manure application practices. Application practice varies by quantity, type of manure, frequency, and use of chemical fertilizer. A total of 25 of these 30 fields were used for cropping purposes; of these, 16 were irrigated fields and 9 were dryland.

For the BAU Baseline, a proportional number of cropped fields receive manure application in the model (25 of 39 fields). A total of 2,520 acres receive manure, of which 680 acres are dryland and 1,840 acres are irrigated. A standard practice is developed from the producer survey data. For irrigated fields, 25 tonnes of dairy manure are applied every four years, along with supplemental inorganic fertilizer at rates of 31.75 kg N per acre (70 lbs per acre) and 9 kg P per acre (20 lbs per acre) each of the other three years. Manure application takes place on the year that spring wheat is grown in each rotation. For dryland fields, the modeled practice is a 20 tonne application of dairy manure every four years, in addition to 22.7 kg N per acre (50 lbs per acre) and 2.7 kg P per acre (6 lbs per acre) of inorganic fertilizer on non-manure years. The manure application strategy is explained in further detail in section 5.2.4.1, within the discussion of the manure application BMP (Scenario 3).

In the Baseline Scenario, chemical fertilizer is used as a substitute for manure to meet the nutrient requirements of crops. Section 5.6.1.1 details the crop-specific baseline fertilizer application levels used in this version of the baseline.

5.2.1.3 Crop Allocation Strategy

In both baseline scenarios, 15 fields totaling 1,320 acres are used for dryland cropping. The four year rotation of spring wheat, canola, barley, and summerfallow is employed. However, in order to ensure that not all of the fields are devoted to one crop in any given year over the 20 year time frame of the analysis, the rotation sequence is staggered equally among the fields. This is realistic in the sense that producers would be unlikely to plant the same crop in each of their fields for a particular growing season – doing so would present too much risk. As such, the strategy surrounding the decision of what to plant in a given field for a given year is as follows. Each field allocated to dryland cropping (15) is randomly assigned to one of four rotation groups. Each of the rotation groups begins the modeling timeframe planted to a different crop (or fallow), but then follows the same base rotation sequence specified in the preceding section. The four different rotation groups, including the number of fields and acreage in each group and the corresponding crop each field in the group is planted to in a particular year, are summarized in Appendix B.

A similar procedure is undertaken for fields allocated to irrigated cropping in the baseline scenarios. However, an additional categorization must be made: the separation between fields used for the first rotation (includes potatoes and sugar beets) or the second rotation (includes canola). This was done according to the textural category of the field. Expert opinion suggests that potatoes do best when grown in relatively coarse-textured soils (Haarsma, 2015). Coarse textured soils, having a higher percentage of sand and silt as opposed to clay, tend to have better drainage than finer textured soil. These conditions are more favourable for growing potatoes, and as such these fields would be more likely to be selected for that particular use. Of the 29 fields (2,720 acres) allocated for irrigated cropping, 15 (1,440 acres) had coarse-textured soil and were assigned the first irrigated rotation. The other 14 fields (1280 acres), with medium-textured soil, were used for the second rotation in the baseline scenario. From there, each of the two categories were further divided into four rotation groups, using the same procedure as outlined for the dryland fields. Appendix B summarizes the four different rotation groups for the two different irrigated base rotations and the corresponding crops grown in each year.

5.2.1.4 Pastureland

The allocation of land as pasture in the baseline scenarios, and the corresponding categorization of those fields, is outlined in section 5.1.1 and summarized in Table 5.1. Of the 65 fields, 26 are used for pasture, which corresponds to 2,640 acres. The majority of that area is native range (76%).

5.2.2 Scenario 1: Introduction of Alfalfa

The addition of alfalfa, a perennial forage legume, to base crop rotations is considered a BMP for crop production in the LLB watershed. The benefits of including a perennial legume forage in annual crop rotations are discussed in previous chapters. These benefits are the potential for reduced chemical fertilizer application, yield increases to subsequent crops, and increases in soil organic matter due to the increase in accumulated root material (Entz et al., 2002). The reduced need for fertilizer application can help prevent the buildup of residual nutrients in the soil while additions to soil organic matter lead to increases in soil organic carbon (SOC) storage. The following section introduces the strategy employed with respect to this BMP, and details how the economic and biophysical impacts are accounted for.

5.2.2.1 Scenario 1a

Scenario 1a involves the addition of an alfalfa hay crop to irrigated rotations and an alfalfa/grass mix crop to dryland rotations across all cropped land in the watershed. The addition of this perennial crop to base rotations triggers additional management considerations. Because alfalfa is a perennial plant, a decision must be made regarding the length of the stand in the rotation. An alfalfa stand may be productive for three to five years; however, alfalfa stand yields tend to increase over the first few years of the life of the stand and then decrease as the stand ages further (Aasen and Bjorge, 2009; Leyshon et al., 1981). A stand must remain for a minimum of two or three years before certain benefits, such as soil N accumulation and weed suppression, are realized. According to Entz et al (1995), the economically optimum stand length is most likely around three to four years, and producers in the prairie regions of Canada are unlikely to break alfalfa stands prematurely due to termination costs and difficulty in re-establishing.

Because alfalfa is being introduced to crop rotations for environmental reasons, it is important to consider environmental risks as well. Producers who leave alfalfa stands longer than four years run the risk of accumulating excess NO_3^- in the subsoil system (Entz et al, 2001). An alfalfa plant will first 'mine' the soil for NO_3^- already present in the soil. However, when this resource is exhausted the plant will produce N via biological fixation. Excess NO_3^- from this process, if not used by the plant, is at risk of leaching into groundwater if it moves too far down through the soil profile. Additionally, Campbell et al (1994) found that if legume plowdown is followed by a fallow period, the risk of NO_3^- leaching is greater. This is because higher net N mineralization may occur due to decomposing legume residues in conjunction with increased soil moisture storage and without uptake from a subsequent annual crop. Therefore, the BMP strategy employed in this modeling scenario is the introduction of three years of alfalfa hay into crop rotations, following by three years of annual crops. An alfalfa stand of this length represents a balance between economic and environmental considerations, including both benefits and risks. A three year alfalfa stand was also modeled by Trautman (2012) and Xie (2014).

Another decision is where in the rotation to insert the alfalfa stand. Previous studies were used to guide this decision. Based on information from Roessel (2012), potatoes are not generally grown directly following an alfalfa stand. Instead, a cereal is usually inserted between the two crops, for two agronomic reasons. First, the soil tends to become compacted due to harvest traffic during the alfalfa stand life. This creates unfavourable soil conditions for growing potatoes (Xie, 2014). Second, perennial stands leave a significant amount of root mass after termination. Until this root mass begins to decompose, conditions are less favourable for growing new crops. Because potatoes are generally a high-value crop, producers would likely prefer to wait until conditions improve with the decomposition of root mass and lessening of soil compaction. Thus, the BMP rotations in Scenario 1a are as follows:

Dryland:
Spring Wheat – Canola – Barley – Summerfallow – AGM – AGM – AGM
Irrigated (1st rotation):
Spring Wheat – Canola – Durum Wheat – Dry Beans – AH – AH – AH
Irrigated (2nd rotation):
Potato – Spring Wheat – Sugar Beets – Barley – AH – AH – AH – Durum Wheat

Where AGM = Alfalfa / Grass Mixed Hay, and AH = Alfalfa Hay.

A similar strategy was used to stagger the rotations and allocate crops to each field in each time period as in the baseline scenarios. However, instead of four rotation groups, dryland and irrigated fields were split into seven or eight groups to reflect the length of the new BMP rotations. Each group begins the modeling timeframe planted to a different crop, but follows the same overall sequence. A summary of the various groups for each rotation can be found in Appendix B. Cropped fields retained their original designation as either dryland or irrigated; additionally, the same categorization of irrigated fields into the two different rotations based on soil texture category was used. Fields used for pastureland in the Baseline Scenario were kept that way in Scenario 1a under the same conditions and parameters.

5.2.2.2 Scenario 1b

In Scenario 1b, the alfalfa BMP is introduced only on dryland fields. Because of the availability of supplementary moisture, irrigated fields produce higher yields than dryland, and can grow a greater variety of crops. Therefore, adopting an alternative BMP rotation is likely less economically feasible for an irrigated crop producer due to the greater opportunity cost of shifting to a lower value crop (i.e., alfalfa). For this reason, when irrigation infrastructure is already in place for a particular field, no alternative BMP is adopted and the baseline rotations are maintained in Scenario 1b. Therefore, the alfalfa/grass mix is included only in the dryland rotation and only implemented on the 1,320 acres allocated to dry cropping.

5.2.2.3 Yield Benefits Following Alfalfa

An important benefit of using alfalfa in crop rotations is the increased yield to crops grown after the stand is terminated. A number of empirical studies have documented this effect on subsequent yields, including Hoyt (1990) and Hoyt and Henning (1971) for Alberta. Hoyt (1990), reporting on trials done in northern Alberta (McLennan, Alberta), indicated that wheat yields increased between 66 and 114% in the eight years following forage termination, relative to continuous wheat cropping. Similar, albeit less pronounced, yield increases have been reported by other studies from different geographic regions of the Canadian prairies as well (e.g., Campbell et al., 1990, in Melfort, SK; Ellert, 1995, in Lethbridge, AB). According to Entz et al (2002), this rotational yield benefit is most pronounced in the northern and eastern zones of the prairie region, where the climate is slightly wetter, and less so in the drier western and southern zones. Where water limits crop productivity, the inclusion of perennials such as alfalfa can deplete soil moisture content and sometimes depress subsequent crop yields, especially when the proportion of fallow periods is reduced to accommodate the perennial. In contrast to annual crops, perennials begin to dewater soil as early as April (when plant growth begins) as opposed to mid-June when ground cover has been achieved (Twerdorff et al., 1999). Entz et al (2002) also suggest that alfalfa grown with a grass, such as bromegrass, may not affect subsequent yields to the same degree as a pure alfalfa stand.

The modelling approach taken by Trautman (2012) was that yield increases could be observed for three years of cropping following a three-year alfalfa stand. Delineated by region of the province, the modeled yield increases for southern Alberta were 10-80% in year 1, and 4-74% in years 2 and 3 for both irrigated and dryland cropping. For the purposes of this analysis, modifications were made to the modelling of these yields benefits regarding dryland cropping. Due to the decrease in proportion of fallow periods, as well as the semi-arid climate of southern Alberta, yield increases in subsequent crops have been revised downward to range from 10-50% in year 1 (spring wheat) to 4-40% in years 2 and 3 (canola and barley) (Table 5.3). According to Dunn (2012) and Bennett and Harms (2011), the maximum percentage increase for an average crop yield ranges from 44% to 61% for irrigated barley, canola, and wheat. For dryland crops, which must contend with moisture limitations and the potential of alfalfa to deplete soil moisture in drier years, it is prudent to set maximum increases in line with these limitations. While some studies have shown yield benefits lasting for longer periods, three years is chosen as a conservative estimate.

As for increases to irrigated crops following alfalfa stands, Xie (2014) set the increase in cereal yield in the first year following alfalfa to be in the range of 10-50%. The range of yield increase in the second year was 5-55% for canola and 5-50% for potatoes. In the third year, seeded to wheat, the yield increases were between 5-40%. Several studies have documented yield increases to potatoes grown after alfalfa in line with this range (Wheeler, 1946; Emmond and Ledingham, 1972; Boring, 2005). Thus, because the same irrigated BMP rotations were employed, the same approach is used in this analysis (Table 5.3).

Table 5.3. Percentage Increase in Yield to Crops Following Alfalfa Hay on Irrigated Fields or Alfalfa/Grass Mix on Dryland Fields.

Year		Dryland	Irrigated 1 [a]	Irrigated 2 [b]
1	Crop	Spring Wheat	Spring Wheat	Durum Wheat
	Range of Increase	10-50%	10-50%	10-50%
2	Crop	Canola	Canola	Potato
	Range of Increase	4-40%	5-55%	5-50%
3	Crop	Barley	Durum Wheat	Spring Wheat
	Range of Increase	4-40%	5-40%	5-40%

[a] The first irrigated BMP rotation;
[b] The second irrigated BMP rotation.

Consistent with Trautman (2012) and Xie (2014), the annual crop yield increases are assumed to be stochastic, and vary from year to year. Since there is a lack of guidance in the literature regarding the distribution and potential trends of this effect, a uniform distribution is assumed and a random draw is taken using the minimum and maximum values above the ranges specified in the tables above. A random draw is done for each field in each year.

5.2.2.4 Estimation of Nitrogen Fixation From Alfalfa

The estimation of biological atmospheric nitrogen (N_2) fixation, in terms of the amount of nitrogen (N) added to the soil is a crucial element of this analysis. Firstly, biological fixation adds N to the soil system, which can be used by subsequent crops. The resulting potential for reduced fertilizer application in subsequent years is an important economic and environmental benefit of the alfalfa BMP. Secondly, biological fixation is a significant nutrient input into the soil system (and potentially the groundwater system). Thus, it is important to keep an accurate accounting of the magnitude of this natural process for the purposes of this analysis. This information is used to inform the calculation of field and watershed-level N balances, the results of which are reported in Chapter 6.

The amount of symbiotically fixed N depends on a number of factors, including legume species, available soil N, crop management, water availability, type of fixing bacteria, and the soil chemical environment (Meisinger and Randall, 1991). The relationships between these factors are difficult to account for, and thus estimates of fixation tend to be crude without direct measurement. Two major features of this process in alfalfa were identified in the literature and incorporated into this analysis. The first important feature is that an alfalfa plant will strategically use available N in the soil before generating N through fixation. Available mineral N in the soil, such as NO_3^-, is easy to obtain via the extensive root system of the perennial plant and requires less energy than the process of fixation. The second important feature of fixation in perennial legumes is that higher amounts of N_2 are fixed by the plant in each additional year of a stand. Generally, the amount of fixation that occurs in an establishment (first) year is expected to be lower, especially if high amounts of mineral N are available in the soil (Russelle, 2004). However, Kelner et al (1997), in a field study of three-year stands of alfalfa in two Manitoba locations, found that biological fixation increased in each successive year of the stand even in the presence of relatively high levels of available soil N. The total amount of N fixed ranged from 174 kg N per hectare for first year alfalfa to 466 kg N per hectare for third year alfalfa (Kelner et al., 1997). This finding corroborated an earlier study by Lamb et al (1995), who found that N_2 fixation can tolerate high levels of soil N. The interactions of these two patterns of the fixation process are illustrated in field experiments conducted by Entz et al (2001). Studying the effect of alfalfa stand length on subsoil N content on soil with initially elevated levels of available N, Entz et al (2001) found that in the first four years of a stand subsoil NO_3^- concentrations were effectively reduced when compared to an annual crop rotation as the perennial plant used up excess mineral N. However, soil NO_3^- levels increased 250% after the fourth year, as the rate of biological fixation increased and eventually exceeded the N needs of the plant. Due to this mineralization and the potential for leaching of legume N, Entz et al (2001) recommended an optimum stand length for alfalfa to be less than six years. When considering the high indigenous NO_3^-concentrations of the soil in the study area, a conservative approach might be to reduce alfalfa stand length to less than 4 years when rotated with annual crops.

Previous studies have estimated N fixation from alfalfa in absolute terms. For instance, Trautman (2012) considered the average contribution of nitrogen by alfalfa to be between 45 and 107 pounds per acre based on information from MAFRI (2010). Xie

(2014) assumed that the total accumulated soil N, after removals from harvesting of hay, from three years of alfalfa was 90 kg per acre. Yang et al (2010) estimated the rate of biological fixation of alfalfa and mixed alfalfa/grass hay on Canadian agricultural land to be between 141-300 and 27-141 kg N per hectare, respectively, although these estimates vary regionally. In Alberta, the estimated average N fixation rate was 212 kg per hectare (86 kg per acre).

In order to account for the fixation patterns described above, and thus reflect variation in the year-to-year quantity of N added to the system, an approach similar to the one suggested by Meisinger and Randall (1991) is employed. Absolute level of N added to the soil from fixation in alfalfa depends on the amount of mineral N left in the soil (residual N), the year of the stand, and the yield of the crop. The yield of alfalfa hay and the alfalfa/grass mix varies year-to-year based on the age of the stand, and is discussed further in section 5.3.2.

According to Meisinger and Randall (1991), the first step is to estimate the N content of the legume crop yield. This is an important step toward estimating N_2 fixation because it sets the upper limit on the amount of fixed N that is removed through harvest. Book values based on the amalgamation of data from various lab analyses can be used for this estimation. According to Alberta's Nutrient Management Planning Guide (AAF, 2007), the amount of N removed per tonne of dry matter alfalfa ranges from 26.1 to 31.9 kg. Taking the mean of this estimate, the amount removed per kg of dry matter is then 0.029 kg N. For a grass crop, the amount of N removed tonne of dry matter ranges from 15.38 to 18.89 kg, for an average of 0.017 kg N per kg dry matter. On dryland fields, a mixed alfalfa/grass hay stand is grown. Assuming the stand is 70% alfalfa, the weighted mean amount of N removed per kg of dry matter is calculated to be 0.025 kg. The book values reported by AAF (2007) and used to calculate the amount of nutrients (N and P) removed from various crops at harvest are reported and discussed in section 5.6.2.1. The values cited above can be used to calculate the total amount of N in the alfalfa crop harvest. For instance, using an average yield of 4451 kg of alfalfa hay per acre, the total amount of N removed in the harvested portion of the crop is:

$$\text{Average Yield (kg acre}^{-1}\text{) x N Removal Rate (kg N kg}^{-1}\text{) = N Removed (kg acre}^{-1}\text{)}$$

$$4451 \text{ kg acre}^{-1} \times (0.029 \text{ kg N kg}^{-1}) = 129.079 \text{ kg N acre}^{-1}$$

The next step in the process is to estimate the proportion of the total N harvested that can be attributed to N_2 fixation (Meisinger and Randall, 1991). This proportion is largely determined by the amount of residual soil N available and the type of legume species. Meisinger and Randall (1991) provide a range of values for the percent of total plant N derived from N_2 fixation for various legume species and available soil N conditions. Since N availability is related to organic matter mineralization, it was recommended that the former be estimated using measurements of soil organic matter; however, the present analysis lacks accurate measurements of this soil characteristic for each field. Bremer et al (2008) assumed 2% organic matter across the LLB watershed and estimated a base N availability of 60 kg N per hectare per year. Another important factor that can be used to estimate fixation rate is the amount of residual available N (e.g., NO_3-N) from the soil. The mineralization of organic matter from prior manure applications does not need to be considered since scenarios featuring the alfalfa BMP (Scenarios 1a and 1b) do not model manure application. Therefore, an estimate of available N can be calculated from a simple calculation of N inputs and removals from the previous annual crops grown in the rotation. The calculation is as follows:

$$\text{Residual N} = \text{N Requirement (kg acre}^{-1}) - (\text{Yield (kg acre}^{-1}) \times \text{N Removed (kg kg}^{-1}))$$

Where *N requirement* is the N input from chemical fertilizer to a certain field based on the annual crop grown that year, *yield* is the average yield of that crop, and *N removed* is the amount of N removed in the harvested portion. The difference is a rough estimate of the amount of residual N leftover on the field and available in subsequent years. The specific values of N requirements, removal, and crop yield for each crop grown in the various scenarios of the analysis are detailed later in this chapter. Note that annual crop yield will fluctuate in the alfalfa BMP scenarios due to the yield benefits discussed in Section 5.2.2.3.

Four years of annual crops are grown in between alfalfa stand in each of the rotations. For the first (establishment) year of alfalfa, the total available N in the soil is calculated to be sum of the residual N from the previous three years, minus any losses from leaching through the soil profile. Janzen et al (2003) estimated, for Canadian cropping systems, that leaching removed 10% of added N in immediately soluble forms (i.e., fertilizer N). However, for dryland cropping in the semi-arid climate of the prairies, leaching is likely to be less because potential evapotranspiration exceeds precipitation. This reduces the rate of downward movement of N through the soil profile and into groundwater (De Jong et al., 2009). Thus, the 10% removal rate is used for irrigated fields and a rate of 5% is used for dryland fields. Three years is seen as a reasonable estimate of the longevity of residual N in the soil. The dry climate also makes N less likely to be exported through runoff from high levels of precipitation. However, after three years it is likely that other biophysical processes, such as denitrification, will result in loss of remaining N (McCallum et al., 2008). The same process is used for the second and third years of the alfalfa stand; however, this three year period now includes the first and second years of the alfalfa stand, respectively.

Using this information, the amount of inorganic N available to the legume annually in the soil can be calculated. The following strategy was developed to estimate the percentage of total harvested plant N that is derived from N_2 fixation. The value for the sum of residual N in the previous three years was broken down into three different levels: <25 kg per acre, 25-50 kg per acre, and >50 kg per acre. Following the direction of Meisinger and Randall (1991) for perennial legumes, ranges for the percentage derived from fixation were developed for alfalfa. Depending on the level of residual N, the lower, middle, or upper end of the range was used. Additionally, the range used depends on the year of the alfalfa stand: first, second, or third. The range of values increases over the three years to reflect one of the major features of N fixation discussed earlier: that fixation rates tend to increase with stand age. The other major feature, that available N in the soil will be 'mined' before the fixation process fully kicks in, is reflected in the adjustment of the range according to the level of the calculated residual N. The overall strategy is summarized in Table 5.4.

Table 5.4. Ranges for Percentage of Harvested Plant Nitrogen Derived From Biological Atmospheric Fixation in Alfalfa, By Year of Stand and Sum of Residual Nitrogen in the Soil.

	Percent Derived from N_2 Fixation					
	1st Year		*2nd Year*		*3rd Year*	
Sum of Residual N [a]	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
< 25 kg $acre^{-1}$	75%	85%	90%	95%	90%	95%
25-50 kg $acre^{-1}$	70%	85%	85%	95%	85%	95%
>50 kg $acre^{-1}$	65%	80%	60%	90%	80%	95%

[a] Calculated as the sum of residual N left in the soil from the previous three years.

When residual soil N levels are high (>50 kg per acre), either a lower range is used (65-80% for first year alfalfa) or the range is expanded to include the lower end (60-90% for second year alfalfa). The opposite strategy is the case when soil N levels are low. Holding the soil N level constant, the range of values generally increases over the three years of the stand.

To calculate the final fixation rate in the harvested portion of the plant, a uniform distribution across the specified range of values is used and a random draw is taken for each field growing alfalfa in each of the 20 years. Without on-site measurements, the method of applying a range of values to estimate biological fixation is appropriate in the context of uncertainty regarding particular environmental variables. For instance, fixation rates depend on the effectiveness of the bacteria inoculum, amount of moisture, and vigor of the particular stand, all of which may vary year to year. Based on experiments done in southern Alberta, the percentage of plant N derived from atmospheric fixation for alfalfa crops is around 80% according to SAFRR (2005). This finding is well within the range of values used in this analysis.

The final component required to estimate the total input of N from biological fixation is the N contained in the non-harvested portion of the alfalfa crop. This includes leaves, stems, crowns, and the root system (Meisinger and Randall, 1991). Approximately one-third of the total plant N is non-harvested N (Meisinger and Randall, 1991; Yang et al., 2010), although some perennials may have up to 60% of their fixed N in the root system (Carlsson and Huss-Danell, 2003). For this study, it is assumed that in the first year of an alfalfa stand, the non-harvested N can be estimated as 50% of the harvested N (one third of the total). In each succeeding year of the stand, it is assumed that the non-harvested N amounts to 25% of the harvested N (Meisinger and Randall, 1991).

5.2.2.5 Chemical Fertilizer Reduction Benefits

A major benefit to the introduction of a perennial legume (alfalfa) in an annual crop rotation is the reduction in required N fertilizer application needed following the perennial stand. According to MAFRI (2010), a five year stand of alfalfa hay can produce an N benefit for up to seven subsequent crop years. Trautman (2012) took the following approach to model this effect. In the first year following an alfalfa stand, 25% of the normal amount of nitrogen fertilizer was applied to the annual crop (spring wheat). In the second, third, and fourth years, it was assumed that 50, 80, and 100% of the normal amount was applied, respectively. Xie (2014) took a similar approach. However, in rotations featuring potatoes grown in the second year following alfalfa, the N application (as a percentage of the normal) was increased to 65% instead of 50%. In those same rotations the third year application (for a cereal crop) was also increased to 85% of the normal. The reason for elevating the amount of N applied to potatoes was two-fold. First, because potatoes are both a high value crop and involve high input costs, producers will generally apply N fertilizer at relatively high rates which can satisfy potato N requirements in most years (Zebarth and Rosen, 2007). In other words, producers make the decision to apply higher rates of fertilizer in order to take advantage of good growing years. This finding is corroborated by Rajsic and Weersink (2008), who determined that the cost of over-application (in terms of wasted fertilizer) is generally low compared to the cost of under-application (in terms of opportunity cost). Second, it is unreasonable to assume that all of the alfalfa fixed N is available to subsequent crops at the right time. As previously discussed, a number of important processes can reduce the availability of N in the soil, including leaching, runoff, and variability in the decomposition rate of organic matter from alfalfa plant residues.

Since this analysis tracks the changes in soil N availability over time, it was possible to assess the N needs of subsequent crops more precisely. For each of the three years following an alfalfa stand, a potential reduction in the need for N fertilizer was modeled. In the first year following alfalfa, it is assumed that the sum of the residual N left by the whole (3 years) of the perennial stand is available, minus a certain percentage of losses (via leaching, runoff, etc.). As before, it is assumed that 10% of available N is lost on irrigated fields, and 5% on dryland fields (Janzen et al., 2003). Thus, to calculate the percentage of normal N application needed in the first year of annual cropping following an alfalfa stand:

N application (as a % of normal) =

[(N Requirement of Crop (kg acre^{-1}) – (Σ (Residual N from each of the previous 3 years) – 10%)] / N Requirement of Crop (kg acre^{-1})

The crop grown in the first year following alfalfa is either spring or durum wheat in each of the modelled rotations. It should be noted that in a given year of an alfalfa stand there is not necessarily a surplus of residual N; in some cases fixation rates may not be high enough and there will be an N deficit. This generally occurs only in the first year of the stand when the previously available N is used first. The modelled level of fertilizer N application is constrained to be no less than 0% of the normal application (nothing is applied) and no more than 100% of the required amount. Fertilizer application levels for each crop and soil type are discussed further in section 5.6.1.1.

In the second year following the perennial stand, it is assumed that only the last two years of alfalfa-fixed residual N are available; similarly, it is assumed that in the third year only the last year of alfalfa-fixed residual N is available. While some of the available N is absorbed by the preceding annual crop (in the first and second years following the stand), significant amounts of organic N can remain as decomposing alfalfa residues. By the third year after a stand is terminated, however, the final amounts of organic N (from the non-harvested portion of the plant) are assumed to mineralize and become available. Similar to Trautman (2012) and Xie (2014), decreasing N credits are available from the legume stand and the amount of fertilizer N that must be applied rises in each subsequent year. In the fourth year following the alfalfa stand, which is still seeded to an annual crop in each of the irrigated rotations (fallow on dryland fields), N fertilizer application is assumed to be 100% of normal.

5.2.2.6 Summary

Scenario 1 presents the introduction of a perennial forage legume, alfalfa, into both irrigated and dryland annual crop rotations across the Lower Little Bow watershed. The irrigated BMP crop rotations, which include a 3 year alfalfa hay stand, are 7 and 8 years long in total. The dryland BMP rotation, which includes a 3 year alfalfa/grass stand, is 7 years in length. In scenario 1b, the alfalfa BMP is only modelled on dryland cropping fields, acknowledging the higher opportunity cost of displacing high value crops on irrigated fields.

In this BMP strategy, several unique environmental and economic outcomes are explicitly accounted for, along with the relevant changes to management decisions. Firstly, following evidence from the literature and guidance from previous modelling approaches, a yield benefit to crops following the perennial stand in the rotation was modelled. Next, the contribution of the biological N_2 fixation process of legumes in adding N to the soil system was accounted for, and changes to soil nutrient levels are tracked. Finally, the resulting benefits of savings from reduced fertilizer application were calculated.

5.2.3 Scenario 2: Introduction of Legume Green Manures, Field Peas, and Elimination of Summerfallow

The following suite of scenarios were developed to model the addition of legume green manures, field peas, and the subsequent reduction or elimination of summerfallow to baseline rotations. Using a crop as a green manure, meaning that it is plowed down into the soil instead of harvested, provides certain benefits such as improved soil organic matter content and reduced erosion when compared to a period of summerfallow. These benefits have been observed to generate increases in yield to subsequent crops over time as soil quality improves (e.g., St. Luce et al, 2015). Additionally, in the case of a legume, nitrogen is fixed and available in the soil for use by subsequent crops, which reduces the need for chemical fertilizer application. Field peas are an alternative, economically viable crop that can be grown in both the Brown and Dark Brown soil regions of Alberta (Trautman, 2012). In addition to the revenue generated by harvesting this annual legume, field peas also have the ability to fix nitrogen in the soil and thus can be considered a

BMP. Yield benefits to following crops in the rotation have also been observed (Harapiak, 2007; St. Luce et al, 2015). Another significant drawback of frequent fallow periods is the increased risk of NO_3^- leaching, which occurs because a crop is not present to utilize mineralized N left in the soil from previous cropping (Campbell et al., 1994). However, the practice is still employed in regions of southern Alberta, including the current study area, because it allows soil moisture to build up. The substitution of a green manure crop for a fallow period may suppress subsequent yields if soil moisture levels are inadequate (Zentner et al, 2004). Both the benefits and drawbacks of reducing the proportion of fallow periods, or eliminating the practice entirely, are accounted for in this analysis when a legume green manure crop is added to dryland fields.

Three versions of Scenario 2 were modeled. In each, a legume green manure crop (fababean) is added to both irrigated rotations. Field peas are added to the dryland rotation in the first version, and a legume green manure (red clover) in the second. In the third version, a rotation featuring both field peas and red clover is employed on dryland fields.

5.2.3.1 Scenario 2a

Each version of this scenario features the introduction of a green manuring practice on irrigated fields. As discussed in Chapter 2, this practice can provide many benefits to the soil and subsequent crops. One approach to this practice is to grow a green manure crop following the harvest of an another annual crop in the same year. For example, Xie (2014) modeled the planting of chickling vetch following barley harvest. Another strategy modeled by Xie (2014) was the addition of one year of vetch under-seeded with barley. However, in this analysis the strategy chosen was to grow a legume crop for one year in the rotations solely for the purposes of green manuring. This strategy was modeled by both Trautman (2012) for dryland production and Xie (2014) for irrigated production.

Several species of legumes, both annual and perennial, have been used for green manuring in Alberta, including field peas, black lentil, chickling vetch, alfalfa, clovers, and fababeans. Although other types of crops can be used, legumes are most commonly used for green manuring because they supply their own N through fixation and fertilizer application is not necessary (reducing costs). There are several important considerations when choosing a green manure legume, including soil type and climate (especially precipitation levels). According to AAF (1993), the legume crop should provide enough ground cover to protect against soil erosion, have a high rate of N fixation and biomass production, as well as have high water-use efficiency when used in drier regions. The final consideration is important for dryland fields of the LLB watershed due to the semi-arid climate; however, on irrigated fields, the moisture requirements are assumed to be satisfied and legumes with a higher reliance on moisture can be considered.

The aforementioned legume species each have benefits, drawbacks, and unique management requirements. Field peas produce a substantial amount of biomass and have adequate N fixation capabilities in a range of environments. Additionally, the crop residues break down quickly following plowdown and can generally contribute to the N requirements of following crops (AAF, 2004). According to Rennie and Dubetz (1986), field peas can fix 178 lbs N per acre under irrigation (81 kg per acre). However, seeding costs of field peas can be significant, especially when revenue is not generated from harvest. Lentils also produce adequate levels of N fixation (134 lbs N per acre) and biomass, as well as have quickly decomposable residue, but seed is not always available and often expensive (AAF, 2004). Although lack of availability is also an issue with chickling vetch, Biederbeck et al (1995) found that average N mineralization was greatest after 3 months of incorporation into the soil, when compared to black lentil, Tangier flat pea, and field pea. Addition of N via fixation from the vetch crop was able to adequately balance removal from the following cereal grain harvest under dryland conditions. Sweet clover has good dry matter production and N fixation rates, but has relatively higher moisture requirements and may suppress subsequent crop yield (AAF, 2004). Lastly, fababeans, a member of the pulse family, are considered a good legume for green manuring because they produce a high amount of symbiotically fixed N as well as a high quantity of dry matter. Rates of N fixation have been observed as high as 267 lbs N per acre (Rennie and Dubetz, 1986; Bremer et al., 1988). Alipour et al (2013) found that fababean acquire approximately 80% of their total N through atmospheric fixation, although rates as high as 90% have been reported (SAFRR, 2005). While fababeans have enormous potential to contribute N to the soil system, a significant drawback is the high intensity of moisture use. According to AAF (2004), fababeans require as much as 8 inches of water in a season to reach their N fixation potential, as lower amounts of moisture would be detrimental to fixation rates. However, for fields under irrigation the water requirement can likely be satisfied and therefore fababean can be considered a viable green manure crop. The high rates of N fixation will have both economic and environmental impacts.

For the reasons outlined above, fababean is selected as the crop to be used for green manuring on irrigated fields in all three versions of Scenario 2 (a, b, and c). Fababean is grown as an annual crop in one period of the rotation. The corresponding change to each of the baseline irrigated rotations is thus as follows:

Baseline Irrigated 1 Rotation:
Spring Wheat – Canola – Durum Wheat – Dry Bean

Scenario 2a Irrigated 1 Rotation:
Spring Wheat – Fababean – Canola – Durum Wheat – Dry Bean

Baseline Irrigated 2 Rotation:
Potato – Spring Wheat – Sugar Beet – Barley

Scenario 2a Irrigated 2 Rotation:
Potato – Spring Wheat – Sugar Beet – Barley - Fababean

In both irrigated BMP rotations of Scenario 2 one additional year is added to include fababean. Similar to Xie (2014), fababean is grown before potatoes in the second BMP rotation. In the first BMP rotation, fababean is inserted between spring wheat and canola into the second year of the cycle. The BMP rotations are employed across all 2,720 irrigated cropping acres of the study area, and the same method as in the Baseline Scenario is used to delineate the fields as belonging to either the first or second irrigated rotation grouping.[15]

Regarding dryland cropping fields, field peas are added to the baseline rotation in Scenario 2a and grown as a revenue-generating crop. Similar to alfalfa, field peas are a legume species and have the ability to symbiotically fix N from the atmosphere and add it

[15] This was done based on soil texture category (i.e., coarse or medium textured). Fields with coarse-textured soil were assigned to the Irrigated 2 BMP rotation, which features potatoes. Appendix B details the field allocation strategy.

to the soil. Along with this potential for reducing chemical fertilizer application on subsequent non-legume crops, field peas have also been shown to generate yield increases to following crops in the rotation. A relatively small percentage (5%) of area under irrigated production was allocated to field peas in Lethbridge County in 2011 (Statistics Canada, 2011). As this may reflect the economic viability of the crop vis-à-vis other crops grown under irrigation, the inclusion of field peas is only modeled for dryland fields. Field peas are an annual crop and therefore are included in one year of the BMP rotation. Trautman (2012) modeled the inclusion of field peas in representative dryland crop rotations in the Dark Brown region of Alberta, which is replicated here:

Baseline Dryland Rotation:

Spring Wheat – Canola – Barley – Summerfallow

Scenario 2a Dryland Rotation:

Spring Wheat – Canola – Barley – Field Peas – Spring Wheat - Summerfallow

Field peas are included between barley and the second period of spring wheat. The proportion of summerfallow decreases as the BMP rotation is six years long. All 1,320 acres of the study area allocated to dryland cropping are devoted to this BMP rotation in Scenario 2a.

5.2.3.2 Scenario 2b

The same two BMP rotations featuring fababeans as a green manure crop are utilized on irrigated cropping fields across all three versions of Scenario 2, including Scenario 2b. For dryland cropping fields, however, the use of field peas as a revenue generating annual crop is replaced in Scenario 2b with a green manure crop. Unlike irrigated fields, which use fababean as the legume green manure crop, red clover is included in the dryland rotation. Red clover is often grown for hay and silage as part of grass mixtures. The species is also a good fixer of N, which makes it suitable for green manuring (Aasen and Bjorge, 2009). According to the Alberta Forage Manual (Aasen and Bjorge, 2009), red clover can fix between 60-115 lbs of N per acre. Red clover also has fair drought tolerance, and unlike fababean, does not require large amounts of moisture to maintain fixation levels. Thus, the dryland BMP rotation in this scenario is as follows:

Baseline Dryland Rotation:
Spring Wheat – Canola – Barley – Summerfallow

Scenario 2b Dryland Rotation:
Spring Wheat – Canola – Barley – Red Clover

Red clover is plowed down and used as a green manure crop, replacing the fallow period of the baseline rotation.

5.2.3.3 Scenario 2c

The final version of Scenario 2 features the introduction of both field peas and red clover (as a green manure) on dryland fields. Evidence from Trautman (2012) indicates that, in the Dark Brown region, the inclusion of both these crops can be an economically viable option on a representative farm. Therefore, an investigation of biophysical impacts is warranted.

The crop rotation on dryland fields is modified to include both field peas and red clover, resulting in an eight-year long BMP rotation. The practice of summerfallow is eliminated, and spring wheat is grown following both BMP crops. The resulting dryland BMP rotation for version 2c is as follows:

Baseline Dryland Rotation:
Spring Wheat – Canola – Barley – Summerfallow

Scenario 2c Dryland Rotation:
S.Wheat – Canola – Barley – Red Clover – S.Wheat – Field Peas – S.Wheat – Canola

The procedure outlined previously for allocating fields to various crops in each year is replicated here with eight different groups, each with a different starting point in the rotation. See Appendix B for details.

5.2.3.4 Yield Benefits Following Legume Green Manuring and Field Peas

Positive impacts to the yield of subsequent crops grown in the rotation are an important element in the potential viability of this suite of BMPs. According to the literature, both the inclusion of field peas and the practice of green manuring can result in increased yields of other crops grown on the same field in later years (e.g., Zentner et al., 2004; St. Luce et al., 2015). This benefit is quantified and modeled in the analysis for Scenarios 2a, 2b, and 2c.

Considering irrigated cropping fields, there are two crops grown after the fababean green manure: potato in the first irrigated BMP rotation and canola in the second. Sincik et al (2008) investigated the impacts of green manuring on tuber yield and quality of potato, and found that, averaging over the four experimental treatment rates of N application, potato yields following fababean increased approximately 15% in comparison to control groups (following wheat). The effectiveness of growing fababean as a green manure in terms of increasing subsequent crop yield has been further corroborated by studies such as Boydston and Hang (1995), Jensen et al (2010), and St. Luce (2015). Boydston and Huang (1995) found that potato yields respond well following green manuring (in this case rapeseed) due to suppression of weeds, whereas the other studies found that fababean green manuring positively impacted both wheat and canola yields. St. Luce et al (2015) reported that wheat grain yield following a fababean green manure crop increased by 28-39% across several western Canadian sites when compared to wheat as the preceding crop. While significant increases to canola yields following annual legumes and the fababean green manure crop were not observed, the authors attributed this finding to adverse weather conditions (drought period) experienced at the sites in those particular years. However, the fababean green manure did result in a 50% reduction in the economic optimal nitrogen rate (EONR), suggesting a positive impact nonetheless. Further evidence of positive impacts on canola yields is provided by O'Donovan et al (2014). In a series of field tests in western Canadian locations, the authors observed greater canola seed yield when grown following fababean green manure versus when wheat, canola, or a range of other legume crops harvested for seed. Across locations, the increase in yield observed was 27%. It is of note that the experiments in that study were conducted on dryland conditions.

As such, the increase in potato yields following fababean green manure is modeled assuming a uniform distribution with a minimum of 5% and a maximum of 20%. A random draw is taken for each field and each year. Since potatoes are grown on irrigated fields, there is no need to model a negative yield impact for a dry year. This same approach was utilized by Xie (2014).

The yield increase to canola following fababean was set to be between 10-30%, again assuming a uniform distribution where a random draw is taken for each field and each year. This approach represents a departure from Xie (2014), who modeled a more conservative 0-10% increase. The 10-30% benefit range is considered appropriate for this analysis based on the amount of evidence suggesting significant improvement to canola yield outcomes following fababean green manuring (O'Donovan et al, 2015; Walley et al, 2007; Khakbazan et al, 2014).

How a legume green manure practice impacts subsequent crop yields on dryland fields remains unclear. Certain studies, such as Zentner et al (1996), have reported a negative impact of the green manure practice on yields in semi-arid conditions due to the decreased availability of soil moisture. Similarly, others have reported a decrease in water use efficiency as a result of green manuring (Krobel et al, 2014). As such, Trautman (2012) opted not to model a yield benefit on dryland conditions and instead modeled a potential yield penalty on crops following the green manure in the event of a dry year. In the absence of a fallow period, the yield decrease was set between 0-16% when the simulated yield of the green manure crop was less than one standard deviation below the minimum yield from municipal level data (indicating a dry year). However, other studies have suggested that alterations in management can minimize and potentially mitigate this issue. For instance, Zentner et al (2004) found that by turning down the green manure crop before full bloom (in early-July vs. late-July or early-August) soil water depletion by the legume plant (lentil in this case) could be minimized. Subsequent wheat yields were equal or almost equal that of a control fallow-wheat-wheat rotation. Improved snow management is also a recommended practice of a green-manure cereal system in the semi-arid prairies (Brandt, 1990; Brandt, 1999). Lastly, results from St. Luce et al (2015) demonstrate the ability of a legume green manure crop to have a positive impact on yields on a dryland site.

Therefore, this study assumes that the appropriate management methods (i.e., early plowdown) are put into place that mitigate potential moisture shortages and suppressed crop yields. Bearing in mind the existing balance of evidence, neither a yield benefit nor a yield penalty is modeled for wheat yields following the red clover green manure on dryland fields in Scenarios 2b and 2c. However, when the legume plant is plowed down earlier in the growing season, biological N fixation is not maximized and the benefit to subsequent crops in terms of an N credit may be diminished. This impact is further discussed in the following sections.

Lastly, annual legumes such as field pea, when grown in cereal-based rotations have also been shown to improve yields to subsequent crops. O'Donovan (2015) found that wheat yields preceded by field pea had increases of approximately 10% across several western Canadian sites. Harapiak (2007) indicated that differences in yield benefits following field peas are related to rainfall levels. Therefore, Trautman (2012) modeled an increase between 20-30% for regions of northern Alberta (Black and Dark Grey soil zones) and a benefit between 0-10% for southern, more arid Alberta regions (Dark Brown and Brown soil zones). A similar approach was undertaken in this analysis for Scenarios 2a and 2c. Yield increases to spring wheat following field pea were assumed to follow a uniform distribution with a minimum of 0 and a maximum of 10% with a draw taken for each field and each year.

Table 5.5 summarizes the modeled yield increases following fababean green manure, red clover green manure, and field peas.

Table 5.5. Percentage Increase in Yield to Crops Following Fababean Green Manure, Red Clover Green Manure, and Field Pea, By Crop Rotation. [a]

	Dryland [b]		**Irrigated 1**		**Irrigated 2**	
BMP	Crop	Range	Crop	Range	Crop	Range
Fababean	-	-	Canola	10-30%	Potato	5-20%
Field Pea	Spring Wheat	0-10%	-	-	-	-
Red Clover [c]	Spring Wheat	0	-	-	-	-

[a] Unlike the alfalfa BMP, yield increases are only modeled for crops in the first year following the BMP practice.
[b] Spring wheat follows both field pea and red clover in the combined dryland rotation of Scenario 2c.
[c] Due to the possibility of moisture depletion, an increase to spring wheat yield is not modeled in this analysis.

5.2.3.6 Estimation of Nitrogen Fixation from Legume Green Manure Crops and Field Peas

Like alfalfa, the BMP crops introduced in Scenario 2 are legumes and have the ability to biologically fix N_2 from the atmosphere. As a significant input of N into the soil system, it is therefore an important aspect of the analysis to estimate the rate of fixation for each of red clover, field pea, and fababeans. As discussed in section 5.2.2.4, rates of N fixation vary widely across different legume species. The amount of N added to soil will factor into the calculated watershed nutrient balance as well as fertilizer reduction benefits to subsequent crops.

Pulse crops such as fababeans have been shown to positively contribute to the N supply capacity of soil over time (Walley et al., 2007). Irrigated fababeans in particular have been singled out for high rates of biological N fixation, which often exceed the net export of crop when harvested (Kopke and Nemecek, 2009). Rennie and Dubetz (1986) investigated rates of N fixation in inoculated (with *Rhizobium* bacteria) legumes grown under irrigated conditions in southern Alberta and found that fababean crops can contribute 216 kg of N per hectare (87.4 kg N per acre). AAF (2004) and SAFRR (2005) reported similar rates of N inputs from fababean crops, and estimated the percentage of plant-N derived from fixation to be 90%. This percentage was highest among a range of legume species reported, but in line with Meisinger and Randall (1991) who state that annual grain legumes can derive between 70-95% of N from fixation when soil N availability is low. Thus, it is assumed that the fababean green manure crops fixes 87.4 kg of N per acre on irrigated fields in Scenario 2.

Red clover is grown on dryland fields as a green manure in Scenarios 2b and 2c. Meisinger and Randall (1991) estimate that a red clover plant fixes between 80-95% and 60-90% of plant-N in low and moderate soil N conditions, respectively. Due to a lack of data in the study area regarding perennial red clover yields, absolute N fixation levels could not be estimated from these ranges. However, the Alberta Forage Manual (Aasen and Bjorge, 2009) provides estimates of nitrogen fixation rates in several legume species directly, including red clover. The range of N fixation cited for red clover is 67-129 kg of N per hectare (27-52 kg N per acre). Within this range, actual N fixation is dependent on soil type, soil pH, moisture, and effective nodulation (Aasen and Bjorge, 2009). An additional consideration is the practice of early plowdown applied to mitigate future moisture shortages. Applying this practice results in a lower rate of N fixation because the plant has not had an entire growing season to develop (Zentner et al, 2004). Therefore, without guidance from the literature regarding the further distribution of fixation rates, it is assumed that red clover fixation adds 33.25 kg of N per acre to the soil system annually, the value of the first quartile of the aforementioned range.

Lastly, field peas also have the ability to fix N in the soil. Rennie and Dubetz (1986) estimated that properly inoculated field peas can contribute 74 kg of N per acre, or 79% of plant N derived from the atmosphere. However, this estimate is based on experiments conducted in irrigated conditions. SAFRR (2005) estimated a slightly higher figure, 80 kg N per acre, but also under irrigated conditions. In a meta-analysis of previous experimental studies done in the northern great plains region (including the Canadian prairie provinces), Walley et al (2007) suggested that the percent of atmosphere derived N for field pea was in the range of 38-75% of plant N with a median value around 57%. Ultimately, the authors concluded that fixation rates are highly variable and dependent on local climate and growing conditions. Yang et al (2010) estimated total legume fixation using crop yields and published estimates for below ground biomass (roots) for each province in Canada. For Alberta, they estimated a N fixation rate of 104 kg per hectare for field peas, or 42.1 kg per acre.

Taking the average yield over the past 10 years of field peas in Lethbridge county, 1,327 kg per acre, and multiplying it by the N removal rate of 0.039 kg N per kg of yield (AAF, 2007), results in 51.75 kg N per acre. Using the value of 79% atmosphere derived N from Rennie and Dubetz (1986), 40.89 kg of N is assumed to be fixed, a value which compares well to the estimated rate by Yang et al (2010). Therefore, for the purposes of this analysis, the estimated rate of N fixation for field peas is 80% of crop yield N.

5.2.3.7 Chemical Fertilizer Reduction Benefits

Similar to the alfalfa BMP modeled in Scenarios 1a and 1b, the addition of biologically fixed N to the soil results in N fertilizer reduction benefits that can be realized for successive crops in the rotation. The magnitude of these N credits varies greatly depending on the legume crop and the practice (i.e., green manuring or harvesting).

The plowdown of fababean and red clover results in the addition of crop residues to the soil system. These residues are a source of organic N that mineralizes over time to provide available inorganic N to subsequent crops. However, Zentner et al (2004) suggest that the mineralization of organic N does not generate a noticeable benefit to subsequent crops until the second time in rotation. According to AAF (1993), between 10-20% of the total annual N fixed by legumes is available to the next crop when the legumes are used as a green manure source, but that an additional 64% of legume N becomes available as plant residues decompose over time. Therefore, Trautman (2012) and Xie (2014) adjusted their modelling of N fertilizer reduction benefits in the following ways. Trautman (2012), in the case of red clover in dryland conditions, assumed an increasing N benefit over the length of the modelling period, where fertilizer N application is 97%, 90%, and 81% of normal levels in the first, second, and third occurrence of legume green manuring in a rotation, respectively. Xie (2014) assumed that 15% of N fixed by fababeans is available to the following crop, regardless of number of occurrences in the rotation timeframe.

A variation on these approaches is taken in this analysis with respect to calculating N benefits following green manuring. Several studies indicate that the benefits from mineralization of organic N extend beyond the first year following a green manure crop (e.g., Jensen et al, 2010). In the semi-arid climate of the LLB watershed study area, it is reasonable to assume that both organic and inorganic N sources remain in the soil for longer than one year. Taking the range of 10-20% provided by AAF (1993), and applying the findings of Zentner et al (2004) regarding the timing of noticeable N benefits, it is assumed that in the case of fababean green manure 10, 15, 20, and 20% of total N fixed is available in the first year following plowdown for the first, second, third, and fourth time in the rotation, respectively. In the second year following fababean plowdown, 0, 5, 10, and 10% of the total N is available to the annual crop. As discussed in the previous section, fababean contributes 87.4 kg of N per acre to the soil through fixation. Table 5.6 breaks down the percentage available and the absolute level of N available in each period following fababean green manure.

Table 5.6. Nitrogen Fertilizer Reduction Benefits to Crops Following Fababean Green Manure, By Year Following Plowdown and Number of Times in Rotation.

	1st Year [a]		**2nd Year** [b]	
Time in Rotation	*% of Fixed N Available* [c]	*N Available (kg acre^{-1})*	*% of Fixed N Available* [c]	*N Available (kg acre^{-1})*
1st	10%	8.74	0%	0
2nd	15%	13.11	5%	4.37
3rd	20%	17.48	10%	8.74
4th	20%	17.48	10%	8.74

[a] The first year following plowdown of fababean; canola is grown in the Irrigated 1 BMP rotation, and potato is grown in the Irrigated 2 BMP rotation.
[b] The second year following plowdown of fababean; durum wheat is grown in the Irrigated 2 BMP rotation, and spring wheat is grown in the Irrigated 2 BMP rotation.
[c] The amount of N fixed by fababean is 87.4 kg per acre.

A slight modification is made in the case of red clover green manuring. Two factors distinguish this process from the fababean green manure: dryland conditions and an early plowdown to mitigate potential future moisture deficiency. The lack of irrigation makes it likely that a lower percentage of total N in the soil will be lost via processes such as leaching and denitrification (De Jong et al, 2009). An earlier plowdown, say in early July, has the consequence of reducing fixation levels and reducing potential N benefits to subsequent crops. This factor was taken into account when estimating total N fixation of red clover (33.25 kg N acre^{-1} year^{-1}). It is assumed that the tendency of soil N to remain longer in the soil under dryland conditions results in a small amount of mineral N available in the third year following plowdown by the third and fourth time that red clover occurs in the rotation. Therefore, the percent of fixed N available to the first year of subsequent crops following red clover in dryland rotations is 10, 15, 20, and 20% for the first, second, third, and fourth time in rotation, respectively. The percentage of N available decreases to 0, 5, 10, and 10% in the second year, and 0, 0, 5, and 5% in the third year. The fertilizer reduction benefits of the red clover green manure are summarized in Table 5.7.

Table 5.7. Nitrogen Fertilizer Reduction Benefits to Crops Following Red Clover Green Manure, By Year Following Plowdown and Number of Times in Rotation.

	1st Year [a]		**2nd Year** [b]		**3rd Year** [c]	
Time in Rotation	*Percent* [d]	*(kg acre^{-1})*	*Percent*	*(kg acre^{-1})*	*Percent*	*(kg acre^{-1})*
1st	10%	3.33	0%	0.00	0%	0.00
2nd	15%	4.99	5%	1.66	0%	0.00
3rd	20%	6.65	10%	3.33	5%	1.66
4th	20%	6.65	10%	3.33	5%	1.66

[a] The first year following plowdown of red clover; spring wheat is grown in the dryland BMP rotation.
[b] The second year following plowdown of red clover; canola is grown in the dryland BMP rotation.
[c] The third year following plowdown of red clover; barley is grown in the dryland BMP rotation.
[d] Percent of fixed N available, which in the case of red clover is 33.25 kg N per acre.

Based on information from Harapiak (2007), Trautman (2012) assumed that N fertilizer application following field peas was 33% of normal application. The same approach is used in this analysis.

5.2.3.8 Summary

Section 5.2.3 detailed the implementation strategy involved in Scenario 2, which includes the legume green manure and field pea rotational BMPs. The first version of the scenario, Scenario 2a, features the inclusion of field peas on dryland cropping fields. Scenario 2b incorporates red clover as a green manure crop into dryland crop rotations. Lastly, Scenario 2c includes both field peas and red clover in a combined BMP rotation on dryland fields. Each version features the use of fababean as a legume green manure on irrigated cropping fields. Crop yield benefits, estimation of N fixation from each of the legume crops, and benefits of reduced fertilizer requirements to crops grown afterwards are discussed.

5.2.4 Scenario 3: Manure Management BMP

As noted in previous sections, the application of manure to satisfy crop nutrients is a common practice in the LLB watershed. Feedlots and other livestock operations are common in the area, producing large quantities of manure that are readily available for application on cropland (Rodvang et al., 2004). In addition to being a substitute for inputs of chemical fertilizer, there are other benefits to manure application. Manure contributes to several desirable soil property benefits, including increased water filtration, improved soil conditions for plant growth, and increased organic matter (Sommerfeldt and Chang, 1985). Improved soil conditions due to manure application can in turn lead to positive crop yield effects (e.g., Lupwayi et al, 2005). However, the over application of manure can negatively impact environmental quality, and in turn pose risks to human health (AAF, 2007). The quantity of manure, the method and timing of application, and the long-term monitoring of soil nutrient status are important factors in mitigating these risks. A commonly recommended practice to address the first factor is P-based manure application, where manure is applied in quantities designed to meet the P requirements of annual crops. The following sections describe this strategy, as well as the modifications made to it for the purposes of Scenario 3 in this analysis. The biophysical outcomes (in terms of N and P balance) are evaluated primarily using the BAU Baseline Scenario as a reference case. However, the impacts to producer returns are evaluated using the Baseline Scenario instead. These decisions are explained in more detail in Chapter 6, section 6.2.3.

5.2.4.1 BMP Strategy

Currently, manure management in Alberta is governed by the *Agricultural Operation Practices Act* (AOPA), which guides and regulates the practice of manure application (AAF, 2008). The current standard for manure application in Alberta is based on the N requirements of crops, where the amount of manure applied cannot exceed the amount required by a crop to meet its N needs. This standard is in place to avoid excessive levels of N building up in soil and the associated environmental risks related to large applications.

However, because the ratio of N to P taken up by major grain and hay crops (4:1) outstrips the typical level of N to P in manure (2:1), N-based manure application has often led to increased levels of P in the soil (Sharpley et al., 1998). Therefore, alternative manure management practices involving P-based application rates have been proposed in some jurisdictions (Miller et al., 2011b). These practices would result in lower overall manure application levels. A variation of this practice that has also been considered is to apply manure at three times the annual P requirement every three years (triennial), which would reduce the application costs to producers compared to yearly application.[16] When comparing the two methods, Miller et al (2011b) found that triennial P-based application did not result in elevated P and N levels in surface water runoff on southern Alberta sites compared to annual applications. However, when compared with the standard annual N-based practice, significantly lower concentrations of mainly dissolved P fractions in runoff were found. These results imply that both triennial and annual P-based application practices retain the targeted environmental benefits.

Xie (2014) modeled a cattle manure application strategy based on meeting the annual N requirement of crops, done once every four years, for a representative southern Alberta irrigated farm. Using this strategy, both N and P requirements of the crop are satisfied the year of application (due to the 2:1 N to P ratio found in cattle manure), and additional chemical fertilizer is not required. As the organic factions of N and P in manure mineralize[17] over time, a further proportion of the total nutrients become available to crops in subsequent years. This availability can be estimated using standard rates of mineralization (see Table 5.9). Based on these estimates, supplemental chemical fertilizer can be applied to close the gap between the available nutrients in the soil and the crop-specific requirements. This strategy closely resembles the triennial P-based application investigated by Miller et al (2011b), as the P requirements of crops grown in the second and third year following manure application are either fully and nearly satisfied by the available P in the soil (see section 5.2.4.2).

An important effect of manure application taken into consideration in this analysis is the non-nutrient yield benefit. This benefit is a result of non-nutrient impacts to soil quality, as discussed in Chapter 3. In order to incorporate this effect, the availability of nutrients in the soil to a crop must be the same across the reference (Baseline Scenario) and alternative (Scenario 3) case. Specifically, in accordance to Liebig's Law of the Minimum where the growth of the plant is governed by its scarcest resource, the availability of at least one nutrient (e.g., N or P) must be equal between the Baseline and

[16] Smith and Miller (2008) found that, compared to an N-based application rate, a triennial P-based rate reduced net returns to feedlot producers by \$0.95 tonne^{-1} versus \$1.55 tonne^{-1} when done annually.
[17] Mineralization is the chemical process in which organic factions of a nutrient are transformed into mineral form and become available for plant uptake.

Scenario 3 for a valid assessment of non-nutrient benefits. This issue was addressed by Xie (2014) in the above strategy, as N availability remained consistent due to supplemental chemical fertilizer inputs. The non-nutrient yield benefit can be incorporated due to the absence of confounding nutrient availability effects.

Therefore, the strategy of manure application in Scenario 3 is to apply manure to both irrigated and dryland fields every four years based on the one-year N requirements of the crop. In order to incorporate the yield benefit effect outlined above, this strategy was chosen over the more common practice of applying 3-4 times the P-requirements once every three years (Smith, 2011). Annual P-based application was not considered for this analysis as the increased cost generally prohibits the widespread adoption of this practice and evidence from Miller et al (2011b) suggests no significant difference in environmental benefit (e.g., P in runoff).

Several important considerations arise when planning a manure application schedule for a particular field. First, certain crops generally do not receive manure for agronomic reasons. For instance, potatoes, sugar beets, and dry beans are susceptible to various negative effects resulting from exposure to manure. Elevated soil N levels can increase root yield in sugar beets, which can impact sucrose content and increase concentration impurities (Carlson and Bauder, 2005). High-value crops, such as potatoes, rarely receive manure due to disease issues (Smith, 2011). Therefore, the application of manure in this scenario is done only on cereal crops. Due to the nature of the baseline rotations, which are used in this scenario, a simplifying assumption is made in that manure is only applied to fields seeded to spring wheat. Since spring wheat is grown every four years in the baseline dryland rotation, as well as both baseline irrigated rotations, this assumption suits the overall BMP strategy well. However, like other preceding scenarios, spring wheat is grown on some fields in each year of the modelling timeframe, reflecting the staggered starting points in crop sequences. The grouping of fields in Scenario 3 is identical to that of the baseline scenario, with the only changes being to manure and fertilizer application rates. As such, the following manure application schedule and rotations are used in Scenario 3:

Dryland Rotation:
Spring Wheat (Manure) – Canola – Barley – Summerfallow

Irrigated 1 Rotation:
Spring Wheat (Manure) – Canola – Durum Wheat – Dry Beans

Irrigated 2 Rotation:
Potatoes – Spring Wheat (Manure) – Sugar Beets – Barley

5.2.4.2 Manure Application Levels and Chemical Fertilizer Reductions

The appropriate application rates of both manure and chemical fertilizer depends on several factors, including crop nutrient requirements, the nutrient content of manure, and the rate of organic nutrient mineralization. Soil testing and manure sampling are commonly done to fine tune application rates over time to avoid an excessive or deficient level of available nutrients (Olson, 2015). However, such information is unavailable for a study of this nature; as such, literature estimates and "book values" are used to estimate the following factors.

First, the nutrient content of manure is estimated according to book values published in Alberta Agriculture's Nutrient Management Planning Guide (AAF, 2007). The typical manure nutrient content of a variety of different livestock species (beef, swine, chicken, etc.) are reported and summarized in Table 5.8. For this analysis cattle manure is used on all cropping fields in this scenario. Although some fields have received swine manure in the past (according to the producer survey), cattle manure is used as a simplifying assumption. Second, nutrient requirements for crop growth, discussed in more detail in section 5.6.1.1, are assumed constant in each year and are developed based on recommended fertilizer rates provided by Kryzanowski et al (1988), AAF (2015), and other materials.

Table 5.8. Typical Nutrient Content of Cattle and Swine Manure, Wet Basis.

Species	Class	Total N [a] (%)	Total N [a] (kg $tonne^{-1}$)	NH4-N [b] (kg $tonne^{-1}$)	Total P [c] (kg $tonne^{-1}$)
Beef	Feeders Finishers Feeder calves Cow/calf pair Cows/bulls	0.65-1.25	10	2.6	2.2
	Paved Feedlot	0.45-0.80	7	2.7	0.9
Swine	Liquid	0.20-0.55	3.5	1.6	1.1
	Solid	0.60-0.90	8	3.2	1.5

[a] Total N is the average amount of N contained in one tonne of manure, including both organic and inorganic factions. The standard observed range of the percentage of total N content in manure is also listed.
[b] Ammonium (NH_4—N) is an inorganic N compound and is plant available. NH_4—N is included within the Total N value. However, NH_4-N is at risk of being converted to NH_3 and lost via volatilization and availability to crops must be estimated using a retention factor.
[c] Total P is the amount of P contained in one tonne of manure.
Source: AAF (2007)

Lastly, the availability of nutrients in manure for crop growth depends on both the mineralization rate of organic factions as well as the availability of NH_4-N. Since it is the mineral form that is available to and can be absorbed by plants, the rate of mineralization plays a large role in the determination of nutrient availability and necessity of supplemental chemical fertilization. The mineralizing rate of organic N and P nutrients is provided in Table 5.9 (AAF 2007). This decay series has previously been found to be characteristic of cattle manure under irrigated conditions in southern Alberta (Olson et al, 2009), although a higher total N mineralization percentage (56%) has also been observed (Chang and Janzen, 1996).

Manure NH_4-N (ammonium) is available in the first year after application, and is listed in Table 5.8. However, NH_4-N is at risk of being converted to NH_3 (ammonia) and lost to the atmosphere via the process of volatilization during and immediately following manure application. The rate of loss depends on a variety of factors, including manure placement, weather conditions at the time, and time elapsed until the manure is incorporated into the soil (AAF, 2007). Therefore, the proportion of NH_4-N available to the crop is estimated using retention factors provided by MAFRD (2008), which depend on application method and weather conditions (Table 5.10). Without further site-specific information, an average retention factor of 0.75 and 0.65 is used for dryland and irrigated fields, respectively, where incorporation into the soil is assumed to take place within one day.

After the third year, it is assumed that the N mineralized from residual manure is negligible, and that a combination of biological processes (e.g., leaching, denitrification, volatilization) are responsible for the fate of the remaining percentage of total N initially applied (Olson, 2015). Environmental conditions, such as intensity and frequency of precipitation, play an important role in the relative significance of each pathway. This issue, and its implications to this analysis, are discussed further in Chapters 6 and 7.

Any residual P is also subject to various biological processes, although a higher percentage of organic P is mineralized and available to crops in this first three years and therefore a smaller amount is likely to be lost to the surrounding environment. Due to high P adsorptive capacity of soils in the area, it is assumed that residual available P is not lost from one growing season to the next over the rotation period (Olson et al., 2010).

Table 5.9. Mineralizing Rate of Organic N and P in Cattle Manure.

	Nitrogen	**Phosphorus**
1st Year	25%	70%
2nd Year	12%	20%
3rd Year	6%	6%
Total	43%	96%

Source: AAF (2007).

Table 5.10. Average Manure Ammonium Nitrogen (NH_4-N) Retention Factors Based on Expected Volatilization Losses.

	Method [a]	**Average Retention Factor**
Dryland	Incorporated within 1 day	0.75
	Incorporated within 5 days	0.55
	Not incorporated	0.34
Irrigated	Incorporated within 1 day	0.65
	Incorporated within 5 days	0.45
	Not incorporated	0.24

Source: MAFRD (2008).
[a] Assuming a surface application strategy instead of injected directly into the soil.

Manure is applied to both irrigated and dryland fields (4,040 acres) of the study area. In each rotation, manure is applied to spring wheat based on its annual N requirement, which works out to a manure application on each field every four years. The following series of calculations are used to determine the amount of manure applied and the subsequent use of N and P chemical fertilizer in following years. This method was adapted from AAF (2007).

- Total organic N (kg $tonne^{-1}$ of manure) after manure application:
  = Total N – NH_4-N
  = 10 kg $tonne^{-1}$ – 2.6 kg $tonne^{-1}$
  = 7.4 kg $tonne^{-1}$
- Available (mineral) N from total organic N in the first year of manure application:
  = Mineralizing rate of N (first year) x Total organic N after manure application
  = 25% x 7.4 kg $tonne^{-1}$
  = 1.85 kg $tonne^{-1}$
- Available N from NH_4-N (kg $tonne^{-1}$ of manure)
  Dryland Fields (retention factor of 0.75)
  = Total NH_4-N – Loss of NH_4-N
  = 2.6 kg $tonne^{-1}$ – (0.25 x 2.6 kg $tonne^{-1}$)
  = 1.95 kg $tonne^{-1}$

Irrigated Fields (retention factor of 0.65)

= 2.6 kg $tonne^{-1}$ – (0.35 x 2.6 kg $tonne^{-1}$)

= 1.69 kg $tonne^{-1}$

- Total available N for crop growth in the first year of manure application:

  = Available mineralized N + Available N from NH_4-N

  Dryland Fields

  = 1.85 kg $tonne^{-1}$ + 1.95 kg $tonne^{-1}$

  = 3.8 kg $tonne^{-1}$

  Irrigated Fields

  = 1.85 kg $tonne^{-1}$ + 1.69 kg $tonne^{-1}$

  = 3.54 kg $tonne^{-1}$

- Amount of manure applied to spring wheat (kg $acre^{-1}$):

  = Annual crop nutrient requirement[18] / Total available N in the first year

  Coarse Texture Dryland Fields

  = (31.75 kg $acre^{-1}$) / (3.8 kg $tonne^{-1}$)

  = 8.36 tonnes $acre^{-1}$

  Medium Texture Dryland Fields

  = (34.02 kg $acre^{-1}$) / (3.8 kg $tonne^{-1}$)

  = 8.95 tonnes $acre^{-1}$

  Irrigated Fields

  = (68.04 kg $acre^{-1}$) / (3.54 kg $tonne^{-1}$)

  = 19.22 tonnes $acre^{-1}$

- Based upon the above calculated application rates, the available P in the first year:

  = Manure application (tonnes $acre^{-1}$) x Total P content of manure x Mineralizing rate of P in first year

  Coarse Texture Dryland Fields

  = (8.36 tonnes $acre^{-1}$) x (2.2 kg P $tonne^{-1}$) x 70%

  = 12.87 kg $acre^{-1}$

  Medium Texture Dryland Fields

[18] Annual crop nutrient requirements are listed in Table 5.21.

= (8.95 tonnes acre^{-1}) x (2.2 kg P tonne^{-1}) x 70%

= 13.78 kg acre^{-1}

Irrigated Fields

= (19.22 tonnes/acre) x (2.2 kg P/tonne) x 70%

= 29.60 kg/acre

- Since the calculated quantities of manure application result in an excess of available P in the first year, residual amounts of P are available in the second year. Residual P available in the second year is calculated as:

  = Available P in the first year – Crop P requirement

  Coarse Texture Dryland Fields

  = 12.87 kg acre^{-1} – 6.93 kg acre^{-1}

  = 5.94 kg acre^{-1}

  Medium Texture Dryland Fields

  = 13.78 kg acre^{-1} – 6.93 kg acre^{-1}

  = 6.85 kg acre^{-1}

  Irrigated Fields

  = 29.60 kg acre^{-1} – 9.90 kg^{-1}

  = 19.70 kg acre^{-1}

- The total available P available in the second year is the sum of residual P and P from mineralizing organic matter:

  = Residual P + (Manure applied x Total P x Mineralization Rate)

  Coarse Texture Dryland Fields

  = 5.94 kg acre^{-1} + (8.36 tonnes acre^{-1} x 2.2 kg^{-1} tonne x 20%)

  = 9.62 kg acre^{-1}

  Medium Texture Dryland Fields

  = 6.85 kg acre^{-1} + (8.95 tonnes acre^{-1} x 2.2 kg tonne^{-1} x 20%)

  = 10.79 kg acre^{-1}

  Irrigated Fields

  = 19.20 kg acre^{-1} + (19.22 tonnes acre^{-1} x 2.2 kg tonne^{-1} x 20%)

  = 27.66 kg acre^{-1}

- The total available N in the second year is the N from mineralizing organic matter:
  = Manure applied x Mineralized N from organic N
  Coarse Texture Dryland Fields
  = 8.36 tonnes $acre^{-1}$ x (7.4 kg $tonne^{-1}$ x 12%)
  = 7.42 kg $acre^{-1}$
  Medium Texture Dryland Fields
  = 8.95 tonnes $acre^{-1}$ x (7.4 kg $tonne^{-1}$ x 12%)
  = 7.95 kg $acre^{-1}$
  Irrigated Fields
  = 19.22 tonnes $acre^{-1}$ x (7.4 kg $tonne^{-1}$ x 12%)
  = 17.07 kg $acre^{-1}$

From the above process the amount of manure applied to each type of field every four years and the availability of nutrients (both N and P) in the second year following application are calculated. The same set of calculations involved in the latter is repeated for the third and fourth years following manure, applying the corresponding mineralization rate and nutrient requirements of the various crops in the different rotations. Tables 5.11 and 5.12 summarize the calculated results. The resultant need for supplemental chemical fertilizer application is calculated as the difference between the crop nutrient requirement and the nutrient availability in the soil. Note that on irrigated fields (both irrigated rotations), supplemental P fertilizer is not required until the fourth year following manure application. This is because the sum of mineralized and carryover (residual) P covers the crop requirements in the first three years. On dryland fields supplemental P fertilization is required in the third year, but not in the fourth as fields are left fallow. Regarding the availability of N, the mineralization of organic factions of manure over time requires that various levels of supplemental N fertilizer be applied in each subsequent year following manure application. By the fourth year following application, the full N crop requirements must be met with chemical fertilizer on irrigated fields. Fertilizer N is not required in the fourth year on dryland fields because of the summerfallow practice.

5.2.4.3 Non-Nutrient Yield Benefits

As discussed earlier in this section, manure application provides non-nutrient benefits to crop yields. This may happen for several reasons, including increased soil organic matter, better moisture-holding capacity, and improved soil structure (Black and White, 1973). Several studies have documented this benefit in a range of different conditions and regions (e.g., Lupwayi et al., 2005; Nitschelm and Reginig., 2005). Following the direction of Xie (2014), non-nutrient benefits to crop yields are assumed to fall within the range of a 1-5% increase, determined by drawing from a uniform distribution for each field and each year.

Table 5.11. Phosphorus Availability from Manure Application on Dryland and Irrigated Fields, Scenario 3.[a]

	Field Category	Crop Grown	P Requirement [c] (kg acre^{-1})	Available P (kg acre^{-1})	Supplemental P Required ($ acre^{-1})	Carryover P (kg acre^{-1})
1st Year [b]	Course Dryland	Spring Wheat	6.93	12.87	$0.00	5.94
	Medium Dryland	Spring Wheat	7.92	13.79	$0.00	5.86
	Irrigated 1	Spring Wheat	9.90	29.60	$0.00	19.70
	Irrigated 2	Spring Wheat	9.90	29.60	$0.00	19.70
2nd Year	Course Dryland	Canola	6.93	9.61	$0.00	2.68
	Medium Dryland	Canola	7.92	9.80	$0.00	1.88
	Irrigated 1	Canola	9.90	28.15	$0.00	8.35
	Irrigated 2	Sugar Beets	19.81	28.15	$0.00	18.25
3rd Year	Course Dryland	Barley	6.93	3.78	$3.99	0.00
	Medium Dryland	Barley	7.92	3.06	$6.15	0.00
	Irrigated 1	Durum Wheat	9.90	10.89	$0.00	0.99
	Irrigated 2	Barley	9.90	20.79	$0.00	10.89
4th Year	Course Dryland	Summerfallow	0.00	0.00	$0.00	0.00
	Medium Dryland	Summerfallow	0.00	0.00	$0.00	0.00
	Irrigated 1	Dry Beans	11.90	0.99	$13.81	0.00
	Irrigated 2	Potato	15.85	10.89	$6.29	0.00

[a] Based on an application rate of 8.36, 8.95, and 19.22 tonnes per acre for coarse dryland, medium dryland, and irrigated fields, respectively.
[b] 1st year refers to the year that manure is applied, 2nd is the first subsequent year, etc.
[c] Crop P requirements are discussed further in section 5.6.1.

Table 5.12. Nitrogen Availability from Manure Application on Dryland and Irrigated Fields, Scenario 3.[a]

	Field Category	Crop Grown	N Requirement [c] (kg acre^{-1})	Available N (kg acre^{-1})	Supplemental N Required ($ acre^{-1})
1st Year [b]	Course Dryland	Spring Wheat	31.75	31.75	$0.00
	Medium Dryland	Spring Wheat	34.02	34.02	$0.00
	Irrigated 1	Spring Wheat	68.04	68.04	$0.00
	Irrigated 2	Spring Wheat	68.04	68.04	$0.00
2nd Year	Course Dryland	Canola	38.56	7.42	$44.00
	Medium Dryland	Canola	40.82	7.95	$46.45
	Irrigated 1	Canola	90.70	17.07	$84.84
	Irrigated 2	Sugar Beets	68.00	17.07	$72.02
3rd Year	Course Dryland	Barley	31.75	3.71	$39.62
	Medium Dryland	Barley	34.02	3.97	$42.45
	Irrigated 1	Durum Wheat	68.04	8.53	$84.08
	Irrigated 2	Barley	68.04	8.53	$84.08
4th Year	Course Dryland	Summerfallow	0.00	0.00	$0.00
	Medium Dryland	Summerfallow	0.00	0.00	$0.00
	Irrigated 1	Dry Beans	49.90	0.00	$70.50
	Irrigated 2	Potato	79.38	0.00	$112.17

[a] Based on an application rate of 8.36, 8.95, and 19.22 tonnes per acre for coarse dryland, medium dryland, and irrigated fields, respectively.
[b] 1st year refers to the year that manure is applied, 2nd is the first subsequent year, etc.
[c] Crop N requirements are discussed further in section 5.6.1.

5.2.4.4 Summary

A manure management BMP was modeled in Scenario 3. Specifically, cattle manure was applied every four years at a rate calculated to satisfy the one-year N requirement of spring wheat. This practice can be considered a BMP because the total amount of manure applied to cropland is reduced, and the availability of P each year from manure closely aligns to the P requirements of crops, which reduces the potential for buildup of residual P in the soil system. The BMP implemented in this analysis closely resembles that of a triennial P-based application practice, where manure is applied every three years at three times the P requirement of the cop. A triennial P-based application was investigated by Miller et al (2011b), who found that certain environmental benefits (such as reduced P loads in runoff) were preserved using this practice compared to an annual P-based application. Also included in this section is a rundown of the specific application levels of both manure and supplemental chemical fertilizer, as well as a discussion of the non-nutrient benefits of manure use.

5.2.5 Scenarios 4 and 5: Land Use Conversions

The following two sets of land use scenarios considered for the LLB watershed involve conversions of land use. Regarding Scenario 4, a less intensive form of agricultural production is implemented, namely the conversion away from annual cropping to permanent forages. The first version, Scenario 4a, represents the most extreme deviation from current land use in the study area, in that all fields initially assigned to annual cropping for the Baseline Scenario (4,040 acres, including both irrigated and dryland) are instead devoted to perennial vegetation. This dramatic decrease in cultivation intensity can induce benefits regarding increased soil carbon and organic matter retention as well as reduced nutrient inputs and risk of leaching in groundwater. A more tempered approach is modeled in the second version (Scenario 4b), where only dryland fields are used for perennial crops and the more productive irrigated fields remain in the baseline annual crop rotations. Lastly, the third version of this scenario, Scenario 4c, involves the application of the land use conversion BMP only on dryland fields deemed marginally productive for annual cropping. The tradeoff between the aforementioned benefits and the cost to producers in terms of foregone annual crop revenue will be evaluated across the three alternative versions and the Baseline Scenario.

The opposite approach was considered for Scenario 5. A significant portion of land in the study area is allotted to less intensive activities, such as tame and native pasture for livestock grazing (see Table 5.1). Changes in ownership, or in relative values between different activities (e.g., changes in annual and perennial crop prices relative to livestock prices) may result in conversions between various agricultural uses. Therefore, an assessment of the tradeoffs in a scenario where less intensive uses are interchanged for more intensive uses is required. In Scenario 5a, all fields used for pasture in the Baseline Scenario (2,640 acres) are converted to annual cropping. The 360 irrigated acres are assigned to the irrigated baseline rotations, and the remaining acreage is assigned to the dryland baseline rotation. The degree of conversion is reduced in Scenario 5b, as the alfalfa BMP employed in Scenario 1a is implemented across the watershed, including the recently converted pastureland. The scenario strategies, including consideration of biophysical impacts, are discussed in the following sections.

5.2.5.1 Scenario 4a

The 4,040 acres allocated to annual cropland in the Baseline Scenario are converted to permanent forage in this BMP scenario. There are several alternatives to annual cropping that are less intensive (e.g., require less inputs) and involve the establishment of forage. As was discussed previously, some fields in the LLB watershed are used solely for livestock grazing purposes and are maintained as permanent native grass cover. These fields require minimal inputs and can generate a multitude of environmental benefits (Dollevoet, 2010). However, the retirement of fields previously used for annual cropping to this use would result in a dramatic loss of income to a producer. Therefore, to maintain a significant revenue stream, this acreage is used instead for the establishment and harvest of hay-producing perennial crops. Hay is a important off-season food for livestock, and a particularly significant source of feed for cattle. A variety of plant species can be used for hay, including many grass and legume species (Aasen and Bjorge, 2009). The prominence of livestock operations in Lethbridge County makes it likely that there will be demand for the increased hay production modeled in this scenario. While the degree of land conversion depicted in Scenario 4a is unlikely to occur (at least over a short period of time), it is nonetheless informative to have a preliminary assessment of the economic and environmental impacts of a widespread change of this nature.

Similar to Scenarios 1a and 1b, a mixed alfalfa/grass crop is seeded on the 1,320 acres of dryland fields. Following the strategy employed by Koeckhoven (2008), a stand length of 7 years is modeled. A range of prospective stand lengths from as short as three years to as long as seven have been shown to produce certain benefits; in Saskatchewan, the average forage stand length is about 6-7 years (Entz et al., 1995). When rotating forage based crops in with annual crops (as in Scenario 1b), important benefits such as nitrogen accumulation and disease suppression can be realized within a short timeframe and more profitable annual cropping can resume. However, establishment and termination of a perennial stand is often difficult and costly, making it more likely that a longer stand length will be employed when a field is exclusively used for permanent forage production (Baron, 2015). Table 5.13 summarizes the dryland alfalfa/grass mix rotation for this scenario. In year 1, the stand is established with the use of a barley cover crop. This is a common practice that provides physical protection to the undeveloped forage as it establishes a root system. Years 2-6 are the alfalfa/grass hay production years, and in year 7 a first cut is taken before the stand is terminated. The hay yield fluctuates over the length of the stand, which is discussed further in section 5.3.2.

Table 5.13. Forage Stand Progression on Dryland Fields, Scenario 4.

Year	**Forage**
1	Greenfeed (Barley Cover Crop)
2	Alfalfa-Grass Mix
3	Alfalfa-Grass Mix
4	Alfalfa-Grass Mix
5	Alfalfa-Grass Mix
6	Alfalfa-Grass Mix
7	Fallow (First cut taken)

On irrigated fields, two perennial forage crops are grown: alfalfa hay and timothy hay. Growing pure alfalfa stands in succession is not recommended in more intensive growing conditions (i.e., irrigation) due to concerns regarding autotoxicity. Autotoxicity occurs when the growth of new alfalfa plants in a stand is hindered by the production of toxins such as medicarpin or ethylene (AARD, 2013). These toxins are allelopathic chemicals that are produced and used by older plants to defend themselves against diseases and pests; however, as the stand matures, toxins build up in the soil and inhibit new alfalfa growth (Miller, 1996). A recommended practice is to therefore seed an alternate crop before the establishment of a new stand, lest future alfalfa yields be suppressed by any remaining toxins. Consequently, timothy hay is seeded following a five year alfalfa stand. The production of timothy hay for export markets has increased dramatically in Alberta in the past several decades, and is a viable forage alternative on irrigated fields in southern Alberta (McKenzie et al., 2009). McKenzie et al (2009) report results from field experiments conducted in Lethbridge county, where four year irrigated timothy stands were grown under various nutrient input regimes. This information was used to inform nutrient input decisions and typical yield assumptions in the present analysis. A stand length of five years is modeled for timothy hay on irrigated fields following the termination of alfalfa.

The conversion to perennial forages on dryland and irrigated fields takes place over the first 3 and 4 years, respectively, of the modeling timeframe, and is completed in year 4 and 5. In both cases, fields are categorized into 1 of 4 or 5 groups. The perennial forage stand (alfalfa/grass mix in the case of dryland fields, pure alfalfa on irrigated) is established in a later year in each of the groups. In the cases of groups 2-5, the forage stand is preceded by one or more years of annual crops. This strategy was employed to more evenly distribute the costs and revenues associated with hay harvesting across years. The rotational strategy employed in Scenario 4a is reported in Appendix B.

5.2.5.2 Scenario 4b

The complete conversion from annual cropping to perennial forages is only implemented on dryland fields in Scenario 4b. The exclusion of high-value annual crops on irrigated fields involves a high opportunity cost to producers in terms of foregone revenue. Therefore, a more realistic scenario would be the conversion of less productive rainfed fields to perennial forage in order to obtain certain environmental benefits. The modeling strategy utilized in Scenario 4b is identical to that of Scenario 4a, except that the land use change BMP only takes place on the 1,320 acres of dryland fields. Annual cropping activities are maintained on irrigated fields.

5.2.5.3 Scenario 4c

The final version of this scenario involves the conversion of marginal dryland fields to perennial forages. For the purposes of this analysis, marginal fields are those classified as containing soil polygons with a 'T' (slope) limitation, as indicated by the AGRASID (Soil Information Centre, 2016). As discussed in section 5.1.2 of this chapter, a slope limitation may reduce the productivity potential of a particular parcel of land. This classification indicates that there are slopes sufficiently steep to incur a risk of water erosion or to limit cultivation (AIWG, 1995).

Even though land may be considered marginal for crop production, a private producer may still have an economic incentive to use it for that purpose. In such cases, the private benefits still outweigh the private costs; however, because yields are lower and economic returns are reduced, the private costs may not outweigh public costs when external impacts are considered. In this way it may be socially beneficial if the land is retired to a less intensive use (Pannell, 2008). A lack of data makes quantifying the extent of decreased returns from marginal land difficult for purposes of this analysis. Since the yields used for modeling are averages across Lethbridge County, which crudely incorporates variation in land productivity potential (and other factors), the estimated decrease in returns may overstate the private impacts of taking marginal land out of annual crop production. Therefore, in a sense, the results from this scenario represent a 'liberal' assessment of the economic-biophysical tradeoffs and private costs may not necessarily be as high.

A total of 560 acres previously designated for dryland cropping activities are identified as 'marginal'. These fields are converted to the same alfalfa/grass hay mix grown in the first two versions of Scenario 4, whereas the remaining 760 dryland acres remain in annual crop production using the baseline rotation. Table 5.14 displays the acreage allotted to various land uses in the three versions of scenario 4.

Table 5.14. Land Use Categories and Corresponding Acreage, Lower Little Bow Watershed, Scenario 4.

		Scenario 4a			**Scenario 4b**			**Scenario 4c**		
	Land Use	Acres	Number of Fields	Percent of Total Area	Acres	Number of Fields	Percent of Total Area	Acres	Number of Fields	Percent of Total Area
Irrigated	Annual Crop	0	0	0%	2720	24	41%	2720	24	41%
	Perennial Crop	2720	24	41%	0	0	0%	0	0	0%
	Pasture (All)	360	5	5%	360	5	5%	360	5	5%
Dryland	Annual Crop	0	0	0%	0	0	0%	760	8	11%
	Perennial Crop	1320	15	20%	1320	15	20%	560	7	8%
	Pasture (All)	2280	21	34%	2280	21	34%	2280	21	34%
	Total	6680	65	100%	6680	65	100%	6680	65	100%

5.2.5.4 Scenario 5a

Further intensification of land use in the LLB watershed is a plausible future scenario. Researching the transitions between various land uses in Alberta between 2000-2012, Haarsma (2014) found that increasing crop commodity prices during this time period resulted in the significant conversion of land used for forage production and pasture to annual cropping. Similarly, the continued expansion of intensive confined animal feeding operations (CAFOs) near the study area may drive demand for feed such as hay and silage and result in the conversion of native areas. As such, an examination of the tradeoffs between increased agricultural intensity and the impact on various biophysical outcomes is warranted.

In Scenario 5a all pastureland is brought into cropping production. Fields originally designated for pasture activity are categorized by soil texture and whether they are rainfed or irrigated. Of the 2,640 acres to be newly cultivated, 1,000 acres have a medium soil texture and 1,640 acres have a coarse soil texture. Only 360 acres of former pasture were irrigated. These categories are used to assign the newly cultivated fields to baseline crop rotations and appropriate management practices (e.g., nutrient input levels).

5.2.5.5 Scenario 5b

Environmental impacts associated with converting pastureland to annual cropping may be mitigated through the implementation of BMPs. In Scenario 5b, the same conversion as Scenario 5a is modeled. However, instead of baseline crop rotations, the alfalfa BMP rotations utilized in Scenario 1a are implemented to mitigate the projected increase in nutrient inputs and decrease in SOC storage. As such, the seven year dryland alfalfa/grass mix BMP rotation is employed on all dryland fields in the watershed, and the two irrigated alfalfa hay BMP rotations are used for cropping on irrigated fields.

The allocation of 26 pasture fields to the irrigated and dryland crop rotations (both baseline and BMP) is displayed in Appendix B. Because neither Scenario 5a nor Scenario 5b involve manure application, the Baseline Scenario is used as a point of reference for the evaluation of private and public impacts.

5.2.5.6 Scenario 5c

The final version, Scenario 5c, features the identical watershed land use as Scenario 5a. All pastureland in the study area is converted for annual crop production purposes. However, the typical manure application practices modeled in the BAU Baseline Scenario are maintained, instead of substituting to chemical fertilizer like the Baseline and Scenario 5a. As such, the total amount of manure in the watershed is held constant between the BAU Baseline and Scenario 5c, permitting the comparison of nutrient balance outcomes. The same fields that receive manure in the BAU Baseline also receive manure in Scenario 5c. The allocation strategy is presented in Appendix B.

5.2.5.7 Nitrogen Fixation and Fertilizer Reduction

Much like Scenario 1, biological N_2 fixation by alfalfa stands represents a significant source of N input into the soil system and generates fertilizer reduction benefits to following crops. The primary difference between Scenario 1 and Scenario 4 is the length of time that the alfalfa stand is grown. In the case of Scenario 1, when the perennial forage is inserted between annual crops, a three-year stand is considered. However, due to practical concerns regarding cost and effort of establishment and termination, a longer five or seven year stand for irrigated and dryland fields, respectively, is modeled in Scenario 4.

The same general trends and characteristics of N_2 fixation in alfalfa identified and discussed in section 5.2.2.4 were accounted for in these scenarios. First, the rate of fixation depends on residual levels of N available in the soil (Meisinger and Randall, 1991). Second, the relative level of N added to the soil system tends to increase over the length of the stand (e.g., Entz et al., 2001; Kelner et al., 1997). Over time, the former trend will outweigh the latter in importance, especially as the stand ages beyond 5 years. The high levels of N added to the system in earlier years of the stand will eventually create a negative feedback effect on fixation rates. In this way perennial legume plants can self-regulate the N levels of their soil environment in the long-run (Penney, 2015). More significantly, however, is the impact that decreased yields have on absolute levels of N additions. While the percent of harvested N attributed to atmospheric fixation may remain relatively constant, the diminished level of total harvested N will result in lower calculated N inputs. Thus, by the latter years of the alfalfa stand, absolute levels of N addition to the soil will begin to decrease compared to earlier levels.

The following table (5.15) summarizes the estimated range of plant N derived from biological fixation by year in the alfalfa stand and sum of residual soil available N levels from the previous three years.

Table 5.15. Ranges for Percentage of Harvested Plant Nitrogen Derived From Biological Atmospheric Fixation in Alfalfa, By Year of Stand and Sum of Residual Nitrogen in the Soil, Scenario 4.

	Sum of Residual N in Soil System [a]		
Year of Stand	*< 25 kg/acre*	*25-50 kg/acre*	*>50 kg/acre*
1	75-85%	70-85%	65-80%
2	90-95%	85-95%	60-90%
3	90-95%	85-95%	80-95%
4	90-99%	80-95%	80-90%
5	90-99%	80-95%	80-90%
6[b]	90-95%	80-95%	60-90%
7[b]	90-95%	80-95%	60-90%

[a] The sum of residual N left in the previous three years is taken for irrigated fields; for dryland fields, the sum of the previous 4 years is taken.
[b] Years 6 and 7 are for dryland alfalfa/grass mix stands only.

Non-harvested N (e.g., in root mass) attributed to atmospheric fixation is accounted for in the same manner as in Scenario 1. In the first year of the stand, 50% of the amount of fixed N in the harvested portion of the plant is assumed to be the value of fixed N in the non-harvested portion; in each subsequent year, 25% is assumed (Meisinger and Randall, 1991).

In the case of irrigated fields, the alfalfa stand is followed by a timothy hay crop. The benefit of reduced fertilizer application is calculated in the same way it was previously for annual crops in scenario 1. That is, in the first year following the termination of alfalfa, the sum of the residual N from the previous three years is considered to be an N credit and deducted from the typical fertilizer requirement. The second and third years of the timothy stand receive an N credit equal to the sum of the residual N from the last two years of the alfalfa stand and the last year of the alfalfa stand, respectively. Again, it is assumed that 10% of available N on irrigated fields and 5% on dryland fields is lost to leaching. By the fourth year of the timothy stand, 100% of the normal nutrient requirement is applied.

5.2.6 Summary of Land Use Scenarios

A total of fourteen land use scenarios for the LLB watershed were developed for the evaluation of certain economic and biophysical outcomes. Two baseline scenarios were described in section 5.2.1, a 'business-as-usual' case featuring the inclusion of typical manure application practices (the 'BAU Baseline') and a non-manure baseline scenario ('the Baseline Scenario'). Each of the two baselines were comprised of the same land uses, including crop rotations. The remaining scenarios featured alternative land uses or management practices, including BMPs designed to improve various environmental outcomes. Scenarios 1a and 1b introduced alfalfa, a perennial forage legume, into annual crop rotations. Scenarios 2a, 2b, and 2c presented the practice of legume green manuring on both irrigated and dryland cropping fields, as well as the inclusion of field peas in crop rotations. A different manure management strategy was modeled in Scenario 3. Lastly, wholesale or partial watershed land use conversions were modeled in Scenario 4 and Scenario 5, the former featuring the retirement of fields historically used for annual cropping to permanent forage, and the latter showcasing a hypothetical set of land uses involving the cultivation of native range originally used as pasture for livestock.

5.3 Crop Yields

Annual crop yield data by county in Alberta were obtained from the Agriculture Financial Services Corporation (AFSC). For the purposes of this analysis, historical yield data between 1978-2013 from Lethbridge County was used. Average annual yield data for dryland production was obtained for the following crops: barley, canola, field peas, durum wheat, and red spring wheat. Irrigated production yields were obtained for the same crops, as well as for potatoes, sugar beets, and dry beans.

5.3.1 Detrending Annual Crop Yield Data

Before deciding which values to use as representative of crop yields in Lethbridge County, the yield data must first be tested for a time trend. Specifically, adding to variability in yields over time due to external production risk factors (e.g., weather) may be effects of changes in technology or 'technical change'. If this technology bias is not removed then the year-to-year variability may be overstated (Swinton and King, 1991). The detrending of yield data can be done by first testing for a time trend using a relatively simple regression of yield (Y) on time (t), which is shown in equation 5.1.

$$Y_t = \alpha + \beta t + \varepsilon_t \tag{5.1}$$

A t-test is used to test the null hypothesis that β=0. A statistically significant slope would be an indication of a significant trend in crop yields over time. If the slope is positive, then progressive technical change over time may be having a positive impact on yield levels. In that case, the yield data can be detrended using the residuals from the regression. Residuals (observed minus predicted values) are added to the predicted value for a base year to create a new, detrended yield series for a particular crop. The most recent year of available data is chosen as the base year, which is 2013 in this case. This procedure was carried out for each of the crops identified above. Regression results, including coefficient estimates, t-statistics, and p-values, are reported for each crop (both dryland and irrigated) in Appendix C.

A significant positive time trend (rejection of the null hypothesis using a 5% level of significance) was found for each of the crops in both dryland and irrigated production, except for irrigated dry beans. Therefore, detrended yield data were generated for all the crops, the exception being dry beans which was left unaltered. A summary of detrended historical crop yields is provided in Table 5.16.

Other studies have used historical data to generate yield distributions as a stochastic parameter for simulation models (e.g., Trautman, 2012; Xie, 2014). However, a simpler approach is utilized here. Using the detrended yield data, the 20-year, 10-year, 5-year, and overall yield averages were calculated for each crop. For the purposes of this analysis, the 10-year (2004-2013) average yield was chosen. It is assumed that the 10-year average represents a reasonable proxy for expected yield into the future. The sensitivity of the final results to this assumption is explored in Chapter 6, section 6.4.3.

Table 5.16. Summary of Detrended Historical Crop Yields in Lethbridge County, Alberta, 1978-2013 (kg $acre^{-1}$).

	Crop	Avg.	10-Year Avg. [a]	Max	Min	Std. Dev.
Dryland	Barley	1,380	1,511	2,202	663	328
	Canola	723	812	1,038	234	190
	Field Peas	1,311	1,327	1,780	782	289
	Durum Wheat	1,236	1,299	1,735	642	252
	Spring Wheat	1,147	1,235	1,584	613	235
Irrigated	Barley	2,208	2,118	2,637	1,511	262
	Canola	1,172	1,219	1,390	801	127
	Durum Wheat	2,265	2,293	2,696	1,589	253
	Spring Wheat	2,024	2,028	2,339	1,315	230
	Dry Beans	968	950	1,273	335	229
	Potatoes	16,281	16,772	18,591	10,661	2,058
	Sugar Beets	20,951	22,570	28,086	13,886	3,509

[a] Average of the years 2004-2013.
Source: AFSC (2014).

5.3.2 Perennial Crop Yields

Historical yield data for the perennial crops considered in this study (e.g., alfalfa, timothy) were not available from AFSC. Variation in perennial crop yields have previously been estimated using published correlation coefficient values between the perennial yield and a reference crop. For instance, Xie (2014) used barley as a reference crop to predict changes in irrigated alfalfa hay and dryland alfalfa-grass hay yields. Since barley yield was modeled as a stochastic variable, alfalfa yield would similarly be stochastic.

Based on expert consultation, the estimated average yield of irrigated alfalfa hay and dryland alfalfa/grass mix in the southern Alberta region was set at 4,451 and 1,600 kg per acre, respectively (Dunn, 2011). The latter figure was also used by Koeckhoven (2008). Since the modeling for both studies was done for representative farms in southern Alberta, these figures are considered appropriate to use for the current study area.

McKenzie et al (2009) investigated the yield and quality response of irrigated timothy hay to various fertilizer application regimes in southern Alberta. Average annual yields of 4 tonnes per acre were obtained based on two field experiment locations.

However, with adequate levels of fertilization, yields closer to 4.5 tonnes per acre were consistently attained. AAF (2009) indicated that under irrigated conditions and when supplied with a high level of nutrients, timothy is capable of producing 5 tonnes per acre. This value is used as the average timothy hay yield in this analysis. Appropriate levels of fertilization are discussed in section 5.6.1.1.

However, as was alluded to in previous sections, it is important to account for patterns of yield change over perennial stand length. Following establishment, perennial forage stand yields tend to first increase over time and then decrease (Entz et al., 2002). At a certain point, termination of the stand becomes necessary. Accurately accounting for this pattern has several important implications in this analysis, both economic and environmental. For instance, the amount of revenue generated is impacted by the dry matter yield each year. Similarly, calculations of N input via biological fixation (in the case of the legume forage) and both N and P outputs also depend on dry matter yield.

Several perennial stand lengths are used in this analysis. In Scenarios 1a and 1b, as well as Scenario 5b, three-year alfalfa stands are modeled on both irrigated and dryland fields, whereas longer stands of five and seven years are used in Scenario 4. Five-year timothy hay stands are also modeled in rotation with alfalfa on irrigated fields in Scenario 4. The percent variation in yield differential relative to the average (discussed above) in each year of the stand is based upon information reported in Leyshon et al (1981) and used in previous studies (Koeckhoven, 2008; Xie, 2014). Based on field experiments conducted in southwestern Saskatchewan, Leyshon et al (1981) collected data on dry matter yield for a five-year alfalfa/grass stand to determine variation in yield between years. They found that alfalfa/grass dry matter yield ranges from as high as 34.20% greater than the five-year mean in year two, to 53.88% lower than the mean in year five. These results are used to construct the trends in perennial alfalfa yields. Table 5.17 displays the percent differential and revised yield for each year of alfalfa hay and alfalfa/grass mix hay stand. Several modifications were made to adapt the values reported by Leyshon et al (1981) to a seven-year alfalfa/grass stand. First, a 0% yield differential is assumed for the first year of the alfalfa/grass mix due to the use of a barley cover crop. Second, yield is 35% lower in the sixth year and 53.88% lower in the seventh for dryland alfalfa, whereas a reduction of 53.88% is found in the fifth year for irrigated alfalfa hay.

Lastly, yield is multiplied by 50% (1/2) and 33% (1/3) in the first year of the dryland and irrigated stands, respectively, because only one cut is taken in the establishment year.[19] In other years, it is assumed that irrigated alfalfa hay produces three cuts and dryland alfalfa/grass mix produces two.

No guidance regarding trends in timothy hay yield over time was found in the literature. Because timothy stands do not share the same concerns of autotoxicity, it is assumed that the degree of decrease in dry matter yield occurring in later years is lower than for alfalfa. Table 5.17 also displays the pattern of yield differential relative to the five-year mean (4,989 kg per acre) assumed for irrigated timothy stands. Similar to irrigated alfalfa, yield is multiplied by 33% (1/3) in the establishment year.

Table 5.17. Yield Variation Over Time for Dryland Alfalfa/Grass Mix, Irrigated Alfalfa Hay, and Irrigated Timothy Hay Stands (kg acre^{-1}).

	Alfalfa/Grass Mix		**Alfalfa Hay**		**Timothy Hay**	
Year	% Diff [a]	Yield	% Diff [a]	Yield	% Diff [a]	Yield
1[b]	0.00%	800	10.00%	1,632	10.00%	1,829
2	10.00%	1,760	34.20%	5,973	34.20%	6,695
3	34.20%	2,147	20.38%	5,358	20.38%	6,006
4	20.38%	1,926	-14.98%	3,784	-7.49%	4,615
5	-14.98%	1,360	-53.88%	2,053	-26.94%	3,645
6	-35.00%	1,040	-	-	-	-
7	-53.88%	738	-	-	-	-

[a] Relative to the five-year mean, which is 1,600 kg acre^{-1}, 4,451 kg acre^{-1}, and 4,989 kg acre^{-1} for alfalfa/grass mix, alfalfa hay, and timothy hay, respectively.
[b] In the first year, only one cut is taken in each of the different perennial stands. Therefore, yield is multiplied by 1/2 in the case of alfalfa/grass, and 1/3 in the case of irrigated alfalfa and timothy hay.

5.4 Crop Prices

Crop price data were obtained from several different sources. Regarding annual crops, prices for barley, canola, field peas, and dry beans were obtained from AAF. The

[19] Perennial crops grown for use as feedstock (e.g., hay) can be harvested multiple times in a growing season, which is referred to as a 'cut'. The average yield values cited above are annual figures, and equal the sum total of all cuts in a season.

2013 edition of the Agricultural Statistics Yearbook (AAF, 2014) was used to collect price data for sugar beets and potatoes. For perennial crops, the 2015 commodity price lists published by AFSC were used to obtain prices for alfalfa/grass hay, alfalfa hay, and timothy hay. Prices of both grass hay and alfalfa hay are reported for each region in Alberta (Peace, North, Northeast, Central, and South) and for each quarter of the year. Since hay is primarily used as a winter season food source, most hay transactions take place in the fall, i.e. the last quarter of the year. Therefore, the prices for alfalfa hay in the last quarter and in the southern region were used. To determine the appropriate price for a mix of alfalfa/grass hay, the following procedure was employed. First, using reported prices for the second cut, the proportional difference between the price of grass hay and alfalfa hay was calculated for the last six years of available data (2009-2015). The average of this proportional difference over the six years was calculated (0.85), and applied to the 10-year (2004-2013) average alfalfa hay price ($0.085 per kg). The resulting 10-year grass hay price is therefore $0.072 per kg, which is used for timothy hay. Regarding the alfalfa/grass mix, a 70% legume (alfalfa) content is assumed. Therefore, the price of alfalfa/grass hay is calculated as a weighted average of the two: (70% x $0.085) + (30% x $0.072) = $0.081 per kg.

Using the Consumer Price Index (CPI) provided by Statistics Canada, the price data for each of the crops were adjusted for inflation. All prices were set to 2014 Canadian Dollars. Similarly, prices were set to a standardized unit: dollars per kilogram (or tonne). As with crop yields, previous studies have used simulated prices as a parameter in simulation models (e.g., Trautman, 2012; Koeckhoven, 2008). In this analysis, a more simplistic approach is applied. Based on a visual inspection of inflation-adjusted crop prices plotted over time, it is determined that the average price of the previous 10 year period (2004-2013) represents a stable value to use for the analysis. This impact of this assumption on final results is explored in via sensitivity analysis, and can be found in Chapter 6, section 6.4.2. A summary of the price data, adjusted for inflation, is displayed in Table 5.18.

Table 5.18. Summary of Historical Crop Price Data (2014 Canadian Dollars) ($ tonne^{-1}).

Crop	**Avg.**	**10 Year Avg.**[a]	**Max**	**Min**	**Std. Dev.**
Barley	247	216	609	112	112
Canola	618	436	1,488	304	244
Field Peas	265	247	440	148	74
Durum Wheat	393	260	1,251	146	217
Spring Wheat	358	253	858	176	157
Dry Beans	803	733	1,313	506	202
Potatoes	244	243	293	146	32
Sugar Beets	84	52	250	42	56
Alfalfa	124	85	233	67	44

[a] Average of 2004-2013.
Source: AAF (2014).

5.5 Economic Relationships

This section details how economic returns to producers are calculated based on the sale of annual and perennial crops, the private returns generated by pasture, and costs of agricultural production in the LLB watershed, including assorted variable and nutrient costs.

5.5.1 Cropping Revenue

Revenue generated by cropping activity in the study area is calculated as the product of crop yield, price, and corresponding acreage in any given year of the modeling timeframe. The acreage allocated to each crop in any given year for each of the scenarios depends on the crop rotation and the assignment given to each individual field. A summary of crop acreage by crop, year, and land use scenario is provided in Appendix B. The specific crop grown on each of the 65 fields in each year can also be found in Appendix B.

5.5.2 Returns from Pasture

The AgriProfit$ Pasture Cost and Returns Profiles provided by AAF (2015a) were used to quantify economic returns from pasture activities. Profiles are available for three regions in Alberta: southern, central, and northern. The southern Alberta region, which includes fescue grassland, mixed grassland, and moist mixed grassland, is chosen for this analysis. Costs and returns are reported for five categories of dryland pasture crops: legume/grass, tame grass, swath grazing, tame/native mixed grass, and native grassland. As previously detailed in section 5.1.1, fields assigned to pasture activities in the LLB study area were categorized as either tame grass or native range. The corresponding crop profiles for the southern Alberta region were retrieved from AAF (2015a). Based on this information, the economic value generated by native and tame grass pasture was set to \$6.07 and \$18.64 per acre, respectively. The higher productivity of tame pasture is reflected in these values (AAF, 2007). Whereas native pasture is land that has not been cultivated, fertilized, or irrigated, tame pasture involves the establishment and maintenance of introduced forage species that generate a greater economic return. Pasture productivity can also be quantified in terms of the rate of animal unit months (AUMs) for grazing it can support. Koeckhoven (2008) cited AUM per acre values of 1.54 and 0.26 for tame and native pasture in the LLB region, respectively.

Another distinction that was made when categorizing fields was whether the field was in an upland or riparian area. Riparian areas are generally more productive than upland areas due to the relatively lower water table which leads to a greater abundance of forage availability (Bork et al., 2001). Studies on riparian ecosystem productivity have shown that double the vegetation production is possible in these area when compared to drier, more upland areas (Unterschultz et al., 2004; Bork et al., 2001). Based on this, Soulodre (2007) indicated that riparian pastures can be considered to have twice the productive capacity of upland pastures. Koeckhoven (2008) adjusted the productivity of tame pasture to 3.08 AUMs per acre and native range to 0.52 AUMs per acre for riparian areas. Based on this, the economic returns generated by native and tame riparian pastures are revised to \$12.14 and \$37.28 per acre, respectively. Lastly, the returns from irrigated tame pasture, of which there are 360 acres in the LLB watershed (no irrigated native range), is set to double the returns of dryland tame pasture. Bremer et al (2008) indicated that recommended stocking rates for irrigated tame pastures in ‘good’ to ‘excellent’ quality can range between 5 to 7.5 AUMs, which is roughly twice the productivity of dryland riparian tame pasture. Therefore, economic returns for upland and riparian irrigated pastures are conservatively calculated as \$37.28 and \$74.56 per acre, respectively.

As was discussed in previous chapters, another suite of BMPs considered for application in the Lower Little Bow watershed involve various livestock management strategies designed to protect riparian areas. Livestock grazing in these areas can have significant impacts on surface water quality, stream bank stability, and erosion (Miller et al, 2011a). For instance, Miller et al (2010) studied how the exclusion of cattle from riparian pasture areas might impact rangeland health, vegetative and soil properties, and water quality. Among other improvements in environmental variables, the finding of reduced mass loads of total N fractions in adjacent surface water suggested that the cattle-excluded pasture was able to act as a buffer for certain runoff variables. This suggests that environmental quality can be improved when riparian pasture is set aside. Although none of the scenarios included in this analysis address the potential for BMPs such as these to improve environmental outcomes in the LLB watershed, it is worth noting that the range of effective practices is not limited to the BMPs investigated here.

5.5.3 Input Costs

The costs to producers of inputs used in the crop production process are referred to as input costs. The following input costs are considered in this analysis: seed (including cleaning and treatment), chemical, trucking and marketing, fuel, oil, and lube, machinery repairs, building repairs, custom work, utilities and miscellaneous, and irrigation (pumping costs) for irrigated crops. Estimates of these costs were obtained from the 2015 AgriProfit$ Cropping Alternatives (AAF, 2015b). Production cost and returns profiles are posted for each of Alberta's soil zones, and values for the Dark Brown region, as well as those for irrigated soils, were obtained for this analysis. It is assumed that the custom costs associated with baling, cutting, and hauling the perennial hay crops is included within various reported input cost values, including 'custom work' and 'trucking and marketing' (Thangaraj, 2015). The values in 'mixed hay' category were used for the dryland alfalfa/grass mix crop. Seed costs for both alfalfa hay and alfalfa/grass mixed hay were included only in the first (establishment) year of the stand. A termination cost of $14.48 per acre was included in the final year of all perennial stands, representing the additional chemical costs of terminating a perennial stand before planting a new crop the following year. Labour is not included as an input cost in this analysis. The costs related to the maintenance of summerfallow fields is also estimated based on the information provided by AAF (2015b).

A Forage Enterprise Analysis, produced by AAF, was used to estimate average production costs for timothy hay (AAF, 2015b; Thangaraj, 2015). Supplemental information from MAFRD (2015) was used in the development of cost estimates. Establishment costs were evenly spread over the five years of stand length, however, seed costs were only included in the first year. Because the values provided by AAF (2015b) and MAFRD (2015) were for dryland production, a $20 per acre irrigation pumping cost was added (same as for alfalfa hay). The various input costs for each crop on a dollar per acre basis are reported in Table 5.20 (dryland, Dark Brown soil zone) and Table 5.21 (irrigated).

Additional production costs for green manure crops were obtained from Xie (2014) and Trautman (2012). An additional cost in the production of fababeans is the inoculant cost. Inoculation of legume crops with symbiotic bacteria species is recommended when a period of at least three years has elapsed since the last legume was planted (PES, 2015). Since fababeans are grown every four or five years (depending on the rotation), the addition of inoculant is appropriate and likely necessary. According to Denton et al (2013), the proportion of plant N derived from fixation significantly increases when inoculant is applied at high rates. Thus, adequate nodulation by Rhizobia bacteria on the fababean is necessary to attain the levels of N fixation detailed in section 5.2.3.7. Three forms of fababean inoculants are available, including peat powder, liquid, and granular soil (Douglas et al., 2013). Assuming a seeding rate of 69 kg per acre (SAFRR, 2005), a seed cost of $1.76 per kg, and an inoculant cost of $1.17 per 25 kg of seed, Xie (2014) set the per acre inoculant cost of fababean to $3.20. That value is used in this analysis. In addition to the cost of the seed itself, seeding cost was set to be the same as field peas ($18 per acre) due to similarity in seeding rates (AAF, 1993). Lastly, termination is done via chemical means and is assumed to carry the same cost as alfalfa termination ($14.48 per acre). Chemical termination was chosen over physical termination (e.g., tilling) because more crop residues are left on the field. This results in a greater pool of organic N material, which over time mineralizes into N available to subsequent crops. Following Trautman (2012) the seeding costs of the red clover green manure crop are assumed to be equal to the average of field pea and mixed hay seeding costs. The remaining input costs are assumed to be the same as that of summerfallow.

The costs associated with the maintenance of pasture fields are reported in Table 5.19. Estimates of these costs were obtained from the southern Alberta profile of AgriProfit$ Pasture Costs and Returns (AAF, 2015a). The tame grass and native grassland figures were used. Compared to annual crops, a less intensive irrigation regime was assumed for the 360 acres of irrigated pastures; the assumed annual cost of pumping for irrigation purposes per acre was $7.50, about half of the typical cost for annual crops such as barley, wheat, and canola. The remaining total direct input expenses are $0.78 per acre for native grassland and $2.52 for tame grass. Fertilization regime is considered separately and discussed in section 5.6.1.

Table 5.19. Input Costs By Pasture Type, Southern Alberta Region ($ acre $^{-1}$).

	Native Range	**Tame Grass**
Seed	0	0
Chemicals	0.01	0.42
Trucking & Marketing	0	0
Fuel, Oil, & Lube	0.26	0.53
Machinery Repairs	0.15	0.41
Building Repairs	0.23	0.39
Custom Work	0	0.19
Utilities & Miscellaneous	0.13	0.58
Irrigation: Pumping Costs	0	0
Total Costs	0.78	2.52

Source: AAF (2015a)

Table 5.20. Input Costs by Crop, Dark Brown Soil Region of Alberta, Dryland Production ($ acre^{-1}).

	Spring Wheat	**Durum Wheat**	**Barley**	**Canola**	**Field Peas**	**Alfalfa/Grass Mix**	**Summerfallow**
Seed	21.00	22.75	16.40	33.01	37.13	3.24	0.00
Chemical	36.97	36.97	14.00	32.36	37.65	1.50	18.00
Trucking and Marketing	25.28	25.28	31.47	14.05	25.28	44.70	0.00
Fuel, Oil, and Lube	10.91	13.05	12.24	13.35	13.48	8.61	8.32
Machinery Repairs	12.48	11.21	12.23	15.28	13.75	9.68	8.15
Building Repairs	1.00	1.00	2.00	1.50	1.75	2.50	1.50
Custom Work	2.10	3.16	6.31	5.26	3.16	9.99	0.00
Utilities and Miscellaneous	11.95	10.35	10.35	10.35	10.35	7.25	0.00
Total Direct Expenses	121.69	123.77	105.00	125.16	142.55	87.47	35.97

Source: AAF (2015b)

Table 5.21. Input Costs by Crop, Irrigated Production in Alberta ($ acre^{-1}).

	Spring Wheat	**Durum Wheat**	**Barley**	**Canola**	**Dry Beans**	**Alfalfa Hay**	**Sugar Beets**	**Potatoes**	**Timothy**
Seed	27.00	26.00	20.50	33.01	54.01	14.88	146.60	367.17	20.12
Chemical	46.25	46.25	23.50	38.30	44.38	2.25	46.38	572.06	9.25
Trucking and Marketing	50.57	56.19	49.45	28.09	23.69	102.17	111.46	167.96	102.17
Fuel, Oil, and Lube	22.84	25.32	27.90	26.69	42.09	41.60	54.33	132.15	24.63
Machinery Repairs	29.04	29.55	28.27	29.55	63.68	34.64	53.49	132.44	10.03
Building Repairs	2.00	2.00	2.00	2.00	4.26	1.00	3.76	20.00	5.87
Custom Work	8.42	9.47	12.62	7.36	31.03	8.42	46.81	138.86	29.26
Utilities and Miscellaneous	15.55	15.55	15.55	15.55	20.70	20.70	27.45	124.30	18.08
Irrigation: Pumping Costs	15.55	14.75	15.00	16.00	20.00	20.00	33.00	52.80	20.00
Total Direct Expenses	217.22	225.08	194.79	196.55	303.84	245.66	523.28	1707.74	239.41

Source: AAF (2015b)

5.5.4 Chemical Fertilizer Costs

Fertilizer costs are also reported in the production costs and returns profiles posted as part of AgriProfit$ Cropping Alternatives (AAF, 2015b). However, the quantified costs for each crop are reported as a generic application, which is unsuitable for this study. Because the scenarios investigated in this analysis involve changes to crop nutrient regimes, it is necessary to break down the composition of nutrient addition for each individual macronutrient. Four macronutrients are considered in the calculation of crop fertilization cost: nitrogen (N), phosphorous (P), potassium (K), and sulphur (S). The source of both N and P is adjusted across different scenarios (i.e., from chemical fertilizer, manure, biological fixation), whereas the application of both K and S is held constant throughout the analysis.

Bulk prices for common fertilizers were obtained from AAF (2015c), and included urea, monoammonium phosphate, muriate of potash, and ammonium sulphate. Prices in April 2015 were used. These prices were broken down into individual nutrient costs using the procedure outlined by AAF (2002). The resulting prices per kg of N, P, K, and S were \$1.41, \$1.27, \$1.18, and \$1.03, respectively. Total fertilizer cost for each crop can be calculated as the sum of each of the four nutrient costs multiplied by the specific nutrient requirements of the crop. The nutrient requirements for each crop grown are discussed in section 5.6.1.1 and are based on the soil texture of the field, year of rotation/stand, and whether the field is dryland or irrigated.

The expected price of chemical fertilizer, in particular to meet N and P requirements, will have a substantial bearing on the results of this analysis. For instance, the attractiveness of the economic tradeoff between obtaining N from chemical fertilizer application or from biological fixation of legume plants will depend to some extent on the price of chemical fertilizer. In monetary terms, the benefits of reduced chemical application will be greater if fertilizer is more expensive. In the last five years, the price of urea (used for N requirements) has ranged from a high of approximately \$792 per tonne in April-May of 2012 to a low of approximately \$530 per tonne in October 2013 (AAF, 2015c). Part of the variation is due to relative demand at different points in the season, as fertilizer prices tend to be higher when producers are seeding crops and fertilizer application is required. However, the price of urea in the most recent seeding season (April, 2015) was \$650 per tonne, a difference of 21% from April 2012 levels. Xie (2014) used a price of \$830 per tonne in his analysis, whereas the April 2015 is used here. A sensitivity of the results to the expected price of fertilizer is explored in Chapter 6, section 6.4.1.

5.5.5 Manure Application Costs

Manure is applied to select fields in the BAU Baseline Scenario, as well as all cropping fields in Scenario 3. The cost of manure application is highly variable and depends on a number of factors, including regional availability, hauling distance, livestock type, and nutrient content. Assumptions used in a previous study (Xie, 2014) that are based on expert opinion (Smith, 2012) are used in this analysis. While a market for manure does not exist per se, crop producers are generally able to purchase manure from nearby livestock producers at a cost that would be approximately equivalent to the cost of N and P obtained through chemical fertilizer (Smith, 2012). Based on a two-mile hauling distance, Xie (2014) set the total cost of manure application at \$8 per tonne. Because Lethbridge County, and around the Lower Little Bow region in particular, is host to a high number of livestock operations, it is assumed that the cost of manure application is relatively constant at the \$8 per tonne rate. Livestock producers also require cropland to spread the manure produced by their animals, and must do so in accordance with Alberta government regulations (based on the N requirements of crops). As such, it is often mutually beneficial for both crop and livestock producers to use manure as a nutrient source on adjacent crop land. Smith and Miller (2008) developed a manure transport model for a typical beef feedlot in the LLB region to evaluate the economics of hauling distance and application rate. The tradeoff between the value of the manure nutrients and the cost of transporting and applying manure was investigated, and they concluded that hauling distance significantly impacts costs to livestock producers. Therefore, it is assumed that manure supply is both abundant and relatively inexpensive to crop producers in the LLB watershed. Even if the two-mile hauling distance was exceeded, it would be unlikely that crop producers would have to pay more than the \$8 per tonne cost since livestock producers also benefit from applying manure on nearby cropland.

Based on the procedure outlined in section 5.2.4.1, the total costs of manure application on a dryland field are between \$57-65 per acre every four years depending on soil texture, and \$172 per acre every four year on irrigated fields.[20] In each case, manure is applied exclusively to spring wheat in order to account for certain agronomic concerns, as discussed earlier.

5.6 Biophysical Relationships

The methods regarding the specification of biophysical variables are discussed in this final section of Chapter 5. The total watershed nutrient balance of both N and P is monitored over the modeling timeframe of each scenario using detailed estimates of nutrient imports and exports. Section 5.6.1 describes the procedures used to determine the level of imports from various sources, including chemical fertilizer, manure, and atmospheric deposition. Imports from the biological N_2 fixation process by leguminous crops are discussed in previous sections (5.2.2.4 and 5.2.3.6). The procedure used to determine nutrient exports is outlined in Section 5.6.2, which consists primarily of harvested crop material and weight gain in livestock on pastureland. Field and watershed-level nutrient balances are calculated as the difference between the sum of nutrient imports and the sum of nutrient exports.

In addition, an estimate of the net change in soil organic carbon (SOC) content is calculated for each scenario. Certain changes in agricultural practices were recognized for the benefits they produce in terms of enhanced SOC storage. In this analysis, these practices were the reduction of summerfallow, increases in the use of perennial crops in crop rotations, and conversion of land to permanent cover. The negative impacts on SOC content due to the reverse of these practices, including bringing formerly native land into cultivation, are also quantified. Section 5.6.3 details the methodology used to quantify these impacts.

[20] Based on an application rate of 8.36, 8.95, and 19.22 tonnes per acre for coarse dryland, medium dryland, and irrigated fields, respectively.

5.6.1 Nutrient Inputs

The addition of nutrients to agricultural systems is a fundamental tenet of agricultural productivity and profitability. However, the misalignment of plant needs and quantity of nutrients supplied can create environmental problems (see Chapter 2). The additions of N and P modeled in each scenario are described in the following sections.

5.6.1.1 Chemical Fertilizer Inputs

Chemical fertilizer N and P inputs are a major source of nutrients in this analysis. Annual applications are modeled for each field in a majority of scenarios. However, the rate of application is highly dependent on a number of factors, including crop type, presence of irrigation, and soil nutrient status. For example, non-legume crops, which are unable to fix nitrogen from the atmosphere, can obtain nitrogen from three main sources: i) available nitrogen stored in the soil, ii) organic nitrogen mineralized over the course of the growing season (e.g., from decomposing crop residues or manure sources), and iii) fertilizer additions (AAF, 2004b).

A soil test is generally recommended as the most accurate way to determine the soil nutrient status. In Chapter 3, a snapshot of soil N and P status in the study area was discussed, based on soil testing done in the summer of 2006. A large degree of variation was found between fields with different histories of nutrient application and cropping activities.

Fertilizer N and P applications modeled in this analysis are based primarily upon extension materials produced by AAF, including the Alberta Fertilizer Guide (AAF, 2004b), the Soil Test Recommendation for Alberta technical manual (Kryzanowski et al., 1988), and AgriProfit$ Cropping Alternatives (AAF, 2015b). Additional information was also drawn from relevant literature sources (e.g., McKenzie et al., 2009; Rajsic et al., 2009), as well as the LLB watershed producer survey conducted between 2006-2011. The objective was to present a realistic assessment of likely application rates based on current trends, recommendations, and changes in management practices (in the case of BMP scenarios).

Kryzanowski et al (1988) detail a procedure to determine basic fertilizer recommendations for both N and P. The recommendation is based first on results of a soil test (calibrated for sample depth and time of collection), and then adjusted for various environmental and management factors, including the specific crop being grown, soil texture, soil region (e.g., Brown, Dark Brown), and whether the field is irrigated or rainfed. Each adjustment is made to reflect the interconnected impacts that various biophysical factors have on crop growth. For instance, fertilizer application rates are adjusted by soil region to reflect the climatic variation within the province (e.g., differences in annual growing season precipitation and rates of evapotranspiration), which impact crop growth potential. Similarly, coarse textured soils have a lower water retention capacity than finer textured soils. This limitation results in a lower amount of available moisture for a crop, which, depending on the season and relative amount of moisture, may impact crop growth. Additionally, coarse textured soils present an increased environmental risk as the potential for loss of nutrients (primarily soluble NO_3-N) through leaching into groundwater is greater (Olson et al., 2009). As such, N and P recommendations are reduced for coarse-textured fields. Another adjustment documented in Kryzanowski et al (1988) pertains to whether a field is irrigated or rainfed (dryland). The removal of the moisture constraint positively impacts the growth potential of a crop, and nutrient application must be adjusted upwards to fully realize this potential.

The baseline N and P fertilizer inputs modeled in this analysis are presented in Tables 5.22 and 5.23, respectively. Drawing from recommendations in Kryzanowski et al (1988), adjustments for soil test area, soil texture, and irrigated conditions were made. In the case of dryland fields, a spring soil moisture status of 'medium' was assumed (mid-point value between dry and wet conditions). Kryzanowski et al (1988) report specific recommendations for N and P fertilization aimed at various annual crops for a range of soil test N and P levels. The greater the level of nutrient availability already present in the soil, the lower the recommended level of fertilizer addition. Because up-to-date soil test results are unavailable for this analysis, this information is used assuming a moderate level of available nutrients in the soil. Specifically, levels of 30 and 10 lbs per acre per two feet of depth (13.61 and 4.54 kg per acre) for N and P, respectively, on irrigated fields, and 10 and 5 lbs (4.5 and 2.3 kg per acre) for N and P on dryland fields are assumed. The justification for this decision is two-fold. First, an examination of soil test N results for 2006 revealed that the average levels on irrigated and dryland cropping fields was 34 and 42 lbs per acre, respectively. Of the 17 dryland cropping fields, 16 had a soil test N level greater than 10 lbs per acre. Of the 22 irrigated cropping fields, a soil test N level greater than 30 lbs per acre was present in 15. Although these findings represent only a snapshot in time, and many factors can influence soil nutrient levels over the long-term, the assumed levels of 30 and 10 lbs per acre can be seen as an appropriate lower bound. Soil test P levels were generally found to be high in the LLB watershed, reflecting a history of high manure and fertilizer application. A discussion of soil test N and P levels in the LLB watershed can be found in Chapter 3, and the finding for each specific field is reported in Appendix A.

Second, based on evidence from Rajsic and Weersink (2008) and Rajsic et al (2009), crop producers will generally err on the side of over-application of fertilizer. This is because the cost of wasted fertilizer is less than the opportunity cost of not applying enough during a favourable growing season. This finding is corroborated by the typical fertilization regimes reported by producers in the study region, which tend to be higher than recommended. The high soil test N and P levels are evidence of that. For instance, N fertilization rates between 120-200 lbs per acre are routinely reported for a variety of annual crops on irrigated fields (see Chapter 3). Based on the above evidence, the stated soil test nutrient levels are seen as reasonable baseline estimates on which a producer might base the fertilization decision.

Based on an assumed 70% legume composition of the alfalfa/grass mix, nitrogen fertilization is unnecessary (AAF, 2004). Fertilization rates for irrigated timothy hay in Scenario 4 are based upon McKenzie et al (2009), who found that the optimum rate of annual N application for timothy yield and quality response in southern Alberta was 240 kg per hectare (97 kg per acre). McKenzie et al (2009) recommended that roughly two-thirds (62.5%) of the annual application occur early in the growing season (mid-April) and the remainder applied after the first cut (assuming two cuts are taken each year). Thus, in the fifth and final year of the timothy stand when only the first cut is taken before termination occurs, only 62.5% of the regular annual application rate is used (68 kg N per acre). Based on McKenzie et al (2009), the annual P fertilization rate was set at approximately 5.26 kg per acre (13 kg per hectare).

Regarding nutrient availability and cycling, pastureland is unique among agricultural systems. Whereas cropping activities involve the addition and removal of large quantities of nutrients (via fertilizer additions and harvest of crop materials), only a small portion of nutrients in pasture systems are removed (AAF, 2009b). Up to 90% of the nutrients removed from the system by grazing livestock are returned to the soil through various forms of excreta, and the remainder is primarily utilized for animal growth and maintenance (see section 5.6.2.2). Denitrification, volatilization, leaching, and runoff processes also play a role in nutrient losses from pastureland (AAF, 2009b). As such, the nutrient inputs required on pasture fields modeled for this analysis are minimal compared to cropping fields. For fields seeded with tame grass, the annual addition of N is set to 10, 12, and 15 lbs per acre (4.5, 5.4, and 6.8 kg per acre) for coarse textured dryland, medium textured dryland, and irrigated pastures, respectively. The annual addition of P is set at 3-4 kg per acre, depending on the type of field. Pasture fertilizer requirements are also reported in Tables 5.22 and 5.23. No additional fertilizer inputs are modeled for the 2,000 acres of native range.

Lastly, it is important to note that the nutrient inputs reported in this section represent a baseline level for activities in the study area. The level of application specified here is used in the Baseline Scenario. Across the various BMP scenarios, however, nutrient inputs are altered in response to changing management practices and soil status. Refer to section 5.2 for details regarding how these changes are quantified and modeled.

Table 5.22. Baseline Nitrogen Chemical Fertilizer Inputs By Crop, Dryland and Irrigated Production (kg N $acre^{-1}$).

Crop	**Dryland** *Coarse Texture*	**Dryland** *Medium Texture*	**Irrigated**
Barley	31.75	34.02	68.04
Canola	38.55	40.82	77.11
Field Peas	6.80	9.07	-
Wheat	31.75	34.02	68.04
Dry Beans	-	-	49.90
Potatoes	-	-	79.38
Sugar Beets	-	-	68.04
Timothy	-	-	97.10
Tame Pasture	4.54	5.44	6.80

Sources: Kryzanowski et al (1988), AAF (2004b), AAF (2015b).

Table 5.23. Baseline Phosphorus Chemical Fertilizer Inputs By Crop, Dryland and Irrigated Production (kg P acre $^{-1}$).[a]

Crop	**Dryland** *Coarse Texture*	**Dryland** *Medium Texture*	**Irrigated**
Barley	6.93	7.92	9.90
Canola	6.93	7.92	9.90
Field Peas	8.91	9.90	-
Wheat	6.93	7.92	9.90
Dry Beans	-	-	11.88
Potatoes	-	-	15.85
Sugar Beets	-	-	19.81
Timothy	-	-	7.92
Alfalfa Hay	-	-	11.88
Alfalfa/Grass Mix	6.93	7.92	-
Fababean	-	-	11.88
Red Clover	8.91	9.90	-
Tame Pasture	2.97	3.47	3.96

[a] Phosphate (P_2O) fertilizer recommendations were converted to P inputs by dividing by 2.29.
Sources: Kryzanowski et al (1988), AARD (2004), AARD (2015b).

5.6.1.2 Manure Inputs

The other significant import of N and P onto agricultural land in this region is the application of manure to cropped fields. Manure is applied to a fixed proportion of fields in the BAU Baseline Scenario, all fields used for cropping in Scenario 3, and the same fixed proportion of initially cropped fields in Scenario 5c. Quantification of nutrient additions via manure application was done using the typical nutrient content of livestock manure reported by AAF (2007). Although the nutrient content of manure can vary widely due to climatic factors, animal feeding regimen, storage conditions, and application method, a full laboratory analysis was not available for the purposes of this study. Therefore, 'book' values are used. For the sake of simplicity, it is assumed that only cattle manure is applied to fields. Slight variation in the manure N:P ratio between different types of livestock (e.g., swine, chicken) may impact the balance and availability of P (since manure is applied on a N requirement basis in both the baseline and BMP scenarios). However, the high density of cattle operations in the area, coupled with the common occurrence of past cattle manure use on fields in the LLB (according to the producer survey), make it likely that a majority of manure applied will be from cattle. The typical nutrient content per tonne of manure (on a wet basis) can be found in Table 5.8, which is located in section 5.2.4.2. The total N and total P content of cattle manure is estimated to be 10 and 2.2 kg per tonne, respectively.

5.6.1.3 Atmospheric Deposition

A final source of N inputs considered in this analysis pertains to atmospheric deposition. Because the LLB watershed is located adjacent to intensive livestock operations, wet and dry deposition of ammonia (NH_3) is likely to be a significant source of N (Hao et al., 2006). However, a more detailed assessment of the spatial distribution of ammonia deposition in the study area is unavailable. Rock and Mayer (2006), in an investigation of nutrient balance in the Oldman watershed (parent to the LLB watershed), assumed an annual deposition rate of 3 kg N per hectare (1.2 kg per acre). This value was used for both baseline scenarios and each BMP scenario.

5.6.2 Nutrient Outputs

The dominant modes of nutrient export in the agricultural regions of the Lower Little Bow considered in this analysis are the harvest of crop material and weight gain of grazing animals on pastureland. When a crop is harvested, significant portions of biotic material containing the nutrients of interest (N and P) are removed from the field. Similarly, the accumulation of nutrients and subsequent weight gain via grazing of pastureland is an important process of livestock production. However, other pathways can also be a significant source of N or P loss from an agricultural system. These pathways include denitrification and ammonia volatilization in the case of N, and leaching or runoff in the case of both N and P. A more detailed discussion of these pathways and their implications for analysis is available in Chapter 2. An accurate assessment of these processes is difficult to undertake, as it is often very site-specific and generally involves a complicated array of climatic and biophysical factors (Janzen et al., 2003). For instance, the leaching of NO_3^- into shallow groundwater may be attenuated by the process of denitrification under certain conditions (McCallum et al., 2008). Olson et al (2009) state that the timing, intensity, and frequency of precipitation and irrigation events has a significant impact on the relative accumulation of nutrients in the soil. Losses of P due to leaching or run-off in an arid climate like the LLB watershed are generally very small (Bremer et al, 2008). Leaching of P in this area is minimized due to the considerable capacity of calcareous soils to adsorb P (Olson et al., 2010); Lutwick and Graveland (1978) found that the P adsorption capacity of soils in southern Alberta ranged from 236 to 950 mg P per kg of soil. The occurrence of extreme precipitation events that would increase the risk of P (or N) loss via runoff is rare in the climate of the this region (Rahbeh et al., 2011). However, it is beyond the scope of this study to incorporate these factors and estimate the precise percentage of nutrient loss for which each pathway is responsible.

In earlier sections, an assumption was made when formulating certain cropping BMP strategies that 10% of immediately available N is lost due to leaching on irrigated fields and 5% is lost on dryland fields. This assumption was made based on Janzen et al (2003), and was corroborated with evidence from field observations in the southern Alberta region, including Rodvang et al (2004), Chang and Entz (1996), and McKenzie (2012). However, an analysis done by De Jong et al (2009) suggests that these estimates may be too high for agricultural land in Alberta, particularly for fields that are rainfed. The loss of N from this pathway was only used when accounting for N availability in soil in the calculation of biological N fixation from legume crops. The final calculations of watershed or field-level N balance in each scenario do not take this loss into consideration, and instead remain an upper-bound proxy of risk of nutrient loss to the environment. The implications of this assumption, and suggested avenues of future research, are discussed in Chapter 7.

5.6.2.1 Harvested Crop Materials

The production of harvestable products like cereal grain and dried forage (hay) depends upon the management and channeling of N into the reproductive, root, and photosynthetic tissues of a crop (Meisinger and Randall, 1991). Harvest and removal of these products is the dominant export of nutrients from the agricultural system.

The removal of nutrients via harvest of crops was estimated using data provided by AAF's Nutrient Management Planning Guide (2007). Book values of nutrient removal on a kg per kg (or tonne) basis are provided for a variety of commonly grown crops. An upper and lower estimate of removal rates are reported, and the mean of these was calculated and used for nutrient removal calculations in this analysis. AAF (2007) reports nutrient removal rates for grass crops on a dry matter (DM) basis, which was used to calculate removals in timothy hay yields. A weighted average was taken between the reported alfalfa and grass DM removal rates (70% alfalfa, 30% grass) for the alfalfa/grass mix crop grown on dryland fields. Note that the composition of this crop was assumed to be 70% legume (alfalfa). The estimated rates of nutrient removal in crop matter used in this analysis are reported in Table 5.24.

Nutrient removal is calculated as the product of the annual crop yield and the crop-specific rate of removal. For instance, the amount of N removed in the average barley harvest on an irrigated field is equal to:

$$= \text{Barley Yield (kg acre}^{-1}\text{) x Removal Rate (kg N kg}^{-1}\text{)}$$
$$= 2118 \text{ kg acre}^{-1} \text{ x } 0.02015 \text{ kg N kg}^{-1}$$
$$= 42.68 \text{ kg N acre}^{-1}$$

An identical procedure is used to calculate P removal. The total export of N and P is calculated for each field in each year of the modeling timeframe depending on the crop grown and the average yield. Note that the yield of perennial crops (alfalfa hay, timothy hay) varies from year to year over the course of a stand, and this impacts absolute level of nutrient removal.

Table 5.24. Average Nutrient Removal Rates in Harvested Crop Material (kg kg^{-1}).

Crop	Nitrogen	Phosphorus
	kg N kg^{-1}	*kg P kg^{-1}*
Barley	0.0202	0.0038
Canola	0.0386	0.0091
Field Peas	0.0390	0.0050
Wheat	0.0250	0.0043
Dry Beans	0.0417	0.0060
Potatoes	0.0032	0.0004
Sugar Beets	0.0020	0.0004
Timothy	0.0175	0.0022
Alfalfa Hay	0.0290	0.0030
Alfalfa/Grass Mix	0.0256	0.0026

Source: AAF (2007)

5.6.2.2 Animal Weight Gain

Weight gain in grazing livestock is another pathway by which nutrients are removed from the local agricultural system. The rate of nutrient export via this pathway was estimated using the livestock stocking rates applied by Koeckhoven (2008), which were detailed in section 5.5.2. Following the work of Bremer et al (2008), it was assumed that a cow-calf pair (AU) gains an average of 36 kg per animal unit month (AUM). Cattle are assumed to contain 24 g of N and 8 g of P per kg of live weight (Jarvis et al., 2002). Therefore, using the example of a 280 acre upland dry tame grass pasture, removal of nutrients (in this example N) via weight gain can be calculated in the following way:

= Total acres of field x Stocking rate (AUM) x (Weight Gain x g N kg Gained^{-1})

= 280 acres x 1.54 AUM acre^{-1} x [(24g x 36kg)/1000]

= 373 kg N

The same procedure is employed for each category of pasture (native, riparian, irrigated, etc.) in each year of the analysis. The total N and P removed each year are reported in Appendix D. These values are used in the watershed nutrient balances calculated for each land use scenario.

5.6.3 Changes to Soil Organic Carbon Storage

Certain land management practices have significant and well-documented impacts on the stock of carbon in agricultural soils in Canada (Vandenbygaart et al., 2003). Soil is an important carbon sink, and has the potential to contribute to either the mitigation or intensification of climate change through the sequestration or release of CO_2. This topic has been studied extensively in the Canadian context (e.g., Smith et al, 2001; Vandenbygaart et al, 2008), and is included in the Canadian government's submission to the UNFCCC[21] on greenhouse gas sources and sinks (National Inventory Report 1990-2009) (EC, 2009). The benefits of the BMP scenarios, in terms of impact on net soil organic carbon (SOC) storage, can be quantified and are reported in this analysis.

A review of the literature and discussion of alternative methods to measure the impacts of agricultural land management on soil carbon storage was carried out in Chapter 4, section 4.1.5. Based on the evaluation of available information, it is determined that the approach adopted in Canada's National Inventory Report (EC, 2009) is appropriate for the estimation of SOC change in soils in the LLB watershed. Included in this report are four land management practices (or changes) that have been shown through previous research to produce a consistent and verifiable impact on SOC storage in Canada: i) adoption of reduced or no tillage practices, ii) reduction or elimination of summerfallow, iii) conversion of annual cropland to either native grassland or permanent forage, and iv) inclusion of perennial crops in annual crop rotations. Because conventional tillage is already rare in the study region (Smith, 2011), only the impacts of the latter three changes in land management practices are quantified in this analysis.

McConkey et al (2014) and Vandenbygaart (2008) outline the methods used in the Canadian National Inventory Report (EC, 2009) to estimate changes in SOC levels. The CENTURY Model, a computer simulation designed to model the dynamics of soil-plant-climate interactions first developed by Parton et al (1987), was employed to estimate soil carbon change factors. These published factors are specific to soil type, agricultural zone, and land practice change and can be used to used to estimate SOC change under a range of different management scenarios (McConkey et al, 2014).

[21] United Nations Framework Convention on Climate Change.

Using carbon change factors produced by the CENTURY Model, McConkey et al (2014) investigated several approaches to predict carbon change in soil over a specified period time for a particular land management change. Based on this analysis, the following equation (5.2) was developed to assess the total change in SOC stock at a specified period of time after a land management change was undertaken.

$$\Delta\, SOC\ Stock_{LMC}(y) = LMC_{MAX} \times\ (1 - exp^{(-k*y)}) \tag{5.2}$$

where LMC is the land management change under consideration, y represents the years since the LMC was undertaken, LMC_{MAX} is the maximum change in SOC produced by the LMC, and k is the rate constant (yr^{-1}).

For each LMC, the rate constant (k) and maximum change in soil carbon (LMC_{max}) parameters are reported in McConkey et al (2014). These parameters are based on results of the CENTURY model, and have been developed for each soil texture category (coarse, medium, fine) and agricultural region in Canada. For the purposes of this analysis, the 'semi-arid prairie' reporting zone was selected. Vandenbygaart et al (2008) validated the results of the CENTURY model for the western prairie regions of Canada by comparing the parameters derived with empirical estimates from previous studies. The CENTURY model proved to be an accurate, albeit conservative, estimator of changes to SOC.

For the BMP scenarios, the total change in SOC stock for each field can be evaluated over the 20-year modeling timeframe using equation 5.2. Relevant land management changes are assumed to occur in year 1. The value of the change in SOC depends on how many years since the onset of the specific activity, *y*, which is set to 20. From McConkey et al (2014), the annual change (from one specified year to the next) can be estimated using the following equation (5.3):

$$\Delta\, SOC\ Stock_{LMC}(y) = LMC_{MAX} \times\ [(exp^{(-k*(y-1)} - exp^{(-k*y)}]/[y - (y-1)] \tag{5.3}$$

where the parameters are defined as before. Table 5.25 summarizes the parameters used for the above equations.

Equation 5.3 is used to calculate and plot the annual changes in soil carbon stock in each field across the various BMP scenarios. However, the parameters listed in Table 5.25 are for complete land management changes, which in some cases is not appropriate for the way the BMP scenarios are constructed. For instance, McConkey et al (2014) report a k-value of 0.0336 and a LMC_{max} value of 1639 mg C per m^2 for a change from annual cropping to perennial forage in a coarse textured field located in the semi-arid prairie reporting zone. These parameters imply a wholesale land conversion. However, in the case of Scenario 1a, a perennial crop (alfalfa) is rotated with annual crops instead of replacing them completely. The proportion of land under the new land management system (perennial cropping) has changed compared to base situation, going from 0 to 38-43% (three years of alfalfa in the seven and eight year rotations). To address this limitation, McConkey (2015) indicated that multiplying the calculated change in SOC (from equations 5.2 and 5.3) by the proportion of land in the new management system will provide the correct adjustment to SOC change. In the above example, the calculated change in SOC is multiplied by 3/7 or 3/8 to reflect the proportion of alfalfa in the rotation. In the case of Scenario 4, which involves the complete transition from annual to perennial crops (permanent cover), the above procedure is unnecessary.

A similar procedure must be utilized for scenarios where the proportion of summerfallow in the rotation is reduced but not eliminated. In Scenario 1a, for example, the proportion of summerfallow decreases from 1/4 (once every four years in dryland rotations) to 1/7 (once every seven years). In these cases, the difference between the calculated change in SOC at the base level (1/4) and the BMP level (1/7) is evaluated as the total change in SOC.

The three land management changes considered in the analysis of soil carbon change (reduction of summerfallow, inclusion of perennials in annual rotations, and conversion to perennial forages) are present in different combinations and intensities throughout the BMP scenarios. The number of fields (and thus acreage) that the changes are implemented on also varies across scenarios, which must be reflected in the calculation of total SOC change. When calculating the total change, an assumption of additivity is made following McConkey et al (2014). This refers to the assumption that changes in SOC from different land management changes do not interact and total change can be calculated as the sum of the individual changes.

Scenario 3, in which cattle manure is applied in place of chemical fertilizer, is the only scenario in which no changes to SOC are calculated. Manure application may locally increase soil organic matter but the impact on regional or national level carbon stocks in soil is negligible considering the C input in feed from which manure C is derived (Vandenbygaart et al., 2003; Schlesinger, 1999). As such, manure application is excluded from the GHG inventory of the National Inventory Report (McConkey et al., 2014; EC, 2009). Although the local increase in SOC due to manure application can provide private benefits, the specific benefit of C sequestration from the atmosphere by the soil system is the public benefit under investigation in this analysis.

In Scenario 5a, the calculated net change in SOC storage will be negative. This is because this scenario involves the intensification of land use, where land previously used for permanent forage (either as native range or tame pasture) is converted to annual cropping. Following McConkey et al (2014), an assumption of reversibility is made, which refers to the assumption that a change in soil carbon resulting from a particular land management change has the opposite sign but same magnitude when the reverse change is considered. As such, McConkey et al (2014) report k-value and LMC_{MAX} parameters for conversion from permanent forage to annual cropping as the negative values of the opposite change (annual to perennial). These parameters were used to calculate SOC storage change in Scenario 5.

Table 5.25. Parameter Values Used to Calculate Change in Soil Organic Carbon Storage, Semiarid Prairie Reporting Zone.

Land Use Change	Texture Category	K-Value	LMC Max
Annual to Perennial	Coarse	0.0336	1639
	Medium	0.0289	2519
Perennial to Annual	Coarse	-0.0336	-1639
	Medium	-0.0289	-2519
Decrease in Fallow	Coarse	0.0305	1314
	Medium	0.0305	1314
Increase in Fallow	Coarse	-0.0305	-1314
	Medium	-0.0305	-1314

Source: McConkey et al (2014).

5.7 Simulation of Scenario Outcomes

Simulation of certain environmental outcomes (N_2 fixation, yield benefits from alfalfa, legume green manures, and manure) was performed using a Monte Carlo simulation add-in for Excel 2013 called SimVoi. The range of each of the random parameters was specified in previous sections, and a uniform distribution was used in each case. A total of 1,000 simulations were used for each scenario. The mean and standard deviations of the results are reported in Chapter 6.

5.8 Chapter Summary

The purpose of this chapter was to classify the fields in the LLB watershed by land use and biophysical characteristics, introduce the alternative land use scenarios, detail the strategies used in the implementation of various BMPs, and describe how the economic and biophysical parameters are accounted for in the analysis. Section 5.1 breaks down the past land use of each field in the LLB watershed as well as the soil and topographic characteristics present. Section 5.2 introduces each of the scenarios, both baseline and alternative, along with the land use and BMP strategies involved in each. Sections 5.2 and 5.3 deal with crop yields and prices, respectively, and section 5.6 describes the various processes involved in the calculation of environmental outcomes.

Chapter 6: Results and Discussion

The results and a discussion of each watershed-level scenario introduced and developed in Chapter 5 are provided in this chapter. The total economic returns generated by agricultural activity in the watershed over the course of the 20-year modeling timeframe are reported through the NPV calculation done for each scenario. Similarly, watershed and field-level outcomes for nutrient balances (N and P) as well as the net change in total soil organic carbon (SOC) storage are reported. The baseline scenarios are used as a reference to determine how changes in land use (i.e., the adoption of a BMP) will impact each of the above parameters of interest. Tradeoff curves between producer returns and each of the biophysical metrics are presented and discussed. Following that, an estimation of public benefits is conducted and incorporated into the calculations of watershed NPV. A sensitivity analysis of a subset of model variables is included at the end of the chapter. While this chapter presents summary results, a comprehensive set of results can be found in Appendix D.

6.1 Baseline Scenario Results

Two scenarios of land use in the LLB watershed were developed to be a baseline point of reference for purposes of comparison with BMP land use scenarios. Both were constructed based on current and predicted land uses and management practices, including the base crop rotations and land use allocation among the 65 fields of the watershed specified in section 5.2.1. The primary baseline scenario ('Baseline'), however, does not include manure application but instead exclusively features chemical fertilizer as a nutrient source for crops. The purpose of the second baseline scenario was to present a 'business-as-usual' condition in the LLB watershed ('BAU Baseline'), and thus includes typical manure application practices (see section 5.2.1). The BAU Baseline is used as a reference case in the evaluation of Scenario 3 and Scenario 5a, the only two BMP scenarios to involve manure application. The rate of application is altered according to the manure management BMP strategy in Scenario 3, whereas it remains consistent with BAU Baseline conditions in Scenario 5a. The economic and biophysical impacts of the remaining BMP scenarios are assessed with the Baseline scenario as the reference condition.

Regarding changes in SOC storage, the baseline scenarios are assumed to have a net change of zero since no changes in land use are implemented to existing conditions. Although small changes are likely to occur throughout the watershed area over the course of the modeling timeframe via short-term changes in land use or management practices[22], the net effect is presumed to be zero in the absence of the three primary practices (long-term reduction of summerfallow, inclusion of perennial crops in rotations, conversion to permanent cover with perennial vegetation) known to have long-term impacts on SOC storage levels. The watershed and field-level N and P balance results are calculated for both versions of the baseline using the procedure described in section 5.6.

Table 6.1 summarizes the findings from each baseline scenario. The NPV of the economic returns generated from agricultural activity in the watershed over the 20-year modeling timeframe is $14,675,017 in the Baseline and $14,929,842 in the BAU Baseline, a difference of 1.71%. The annualized NPV on a per acre basis is $258 and $263, respectively. The slight increase in net returns found for the BAU Baseline is indicative of the tradeoff between the chemical fertilizer and manure as nutrient sources. Every other factor, including acreage allocated to each crop and crop yield, is held constant between the two baselines. At the input prices specified in Chapter 5, manure is a less expensive alternative to the purchase of chemical fertilizer. This finding helps to explain the prevalence of manure as a nutrient source in the study area.

It is important to note that the non-nutrient crop yield benefit of manure application discussed in section 5.2.4.3 is not accounted for in the BAU Baseline. This is because the availability of at least one nutrient (N or P) is not held constant between the baseline manure application rates and baseline chemical fertilizer rates used for non-manured fields and in the Baseline Scenario. Without this condition in place, increases in crop yields may be attributable to increased nutrient availability (or vice versa) instead of changes to various soil properties. However, the realistic possibility of increased crop

---

[22] Changes to fertilization regime, manure application, or irrigation scheduling can result in changes to SOC levels (McConkey et al, 2014). Similarly, the use of summerfallow on dryland fields will cause short-term fluctuations to SOC levels in the baseline scenarios.

yields due to manure application make it likely that the values reported in the table below for the BAU Baseline are conservative estimates, and that the difference in economic returns between the two versions of the baseline is greater. In section 6.10 a sensitivity analysis is performed using revised nutrient input prices, which will test the stability of the tradeoff between the two nutrient sources.

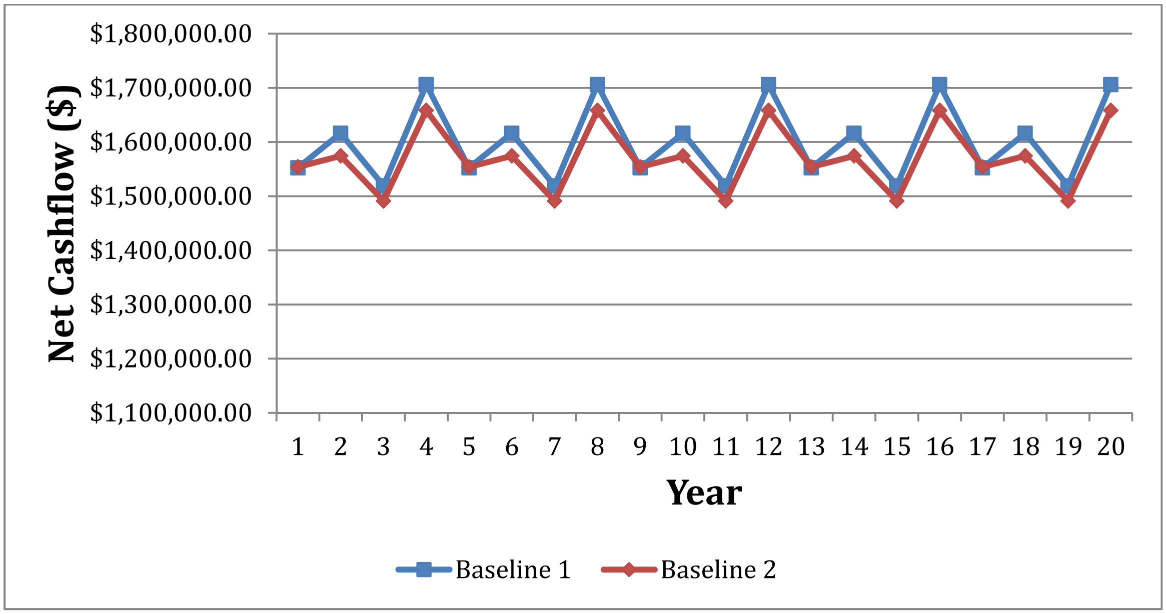
Table 6.1. Summary Economic and Biophysical Results, Baseline and BAU Baseline Scenarios.

	Baseline [a]	**BAU Baseline** [b]
NPV ($) (*Total Watershed*)	$14,675,018	$14,929,842
NPV ($ acre^{-1})	$2,197	$2,235
Annualized NPV ($ acre^{-1})	$258	$263
Mean Annual N Balance (kg acre^{-1})	10.11	18.70
Cumulative N Balance (Tonnes) (*Total Watershed*) [c]	1,350	2,498
Mean Annual P Balance (kg acre^{-1})	2.09	3.93
Cumulative P Balance (Tonnes) (*Total Watershed*) [c]	279	525
Net SOC Change (Tonnes) (*Total Watershed*)		0

[a] Excludes manure application on cropped fields;
[b] Includes typical manure application practices on a proportion of cropped fields;
[c] Cumulative over the 20-year modeling timeframe.

The modified net cashflow generated by agricultural activity in the LLB watershed for the baseline scenarios in each of the 20 years is displayed in Figure 6.1. Positive cashflow is generated in each of the 20 years of both versions, ranging from $1,491,392 to $1,705,873. The year to year variation reflects the change in acreage allocated to each crop in any given year and corresponding changes to input costs.

Figure 6.1. Modified Net Cashflow, Total Watershed, Baseline and BAU Baseline Scenarios.



Economic outcomes of the baseline scenarios can be compared to previous studies conducted on the private impacts of BMP adoption. Trautman (2012) found that dryland crop production in the Dark Brown soil zone generated an annualized mean NPV of $34.21 per acre, which is substantially less than the average found across the LLB watershed in this analysis. However, when breaking down the study area by land use, fields used for dryland production were found to generate an annualized NPV of $110.62 per acre (using Baseline scenario results). This difference can be attributed to certain additional costs included by Trautman (2012), such as the costs of machinery replacement and crop insurance. Xie (2014), who explicitly investigated irrigated crop production in southern Alberta, found similar baseline results to this study: annualized mean NPV ranged between $215 and $589 per acre, depending on the crop rotation. Fields devoted to higher value crops, such as potatoes and sugar beets, generated the highest return. A similar value was found for irrigated production in this study, $535.81 per acre. For the Baseline Scenario, the average annualized NPV of all cropland (4040 acres) was $410 per acre.

In the BAU Baseline Scenario the overall average annual N and P balances were 18.70 and 3.93 kg per acre, respectively. These findings indicate a net surplus of nutrients being added to the watershed each year. Over the 20-year timeframe this will equate to a cumulative surplus of approximately 2,500 tonnes of N and 525 tonnes of P if baseline practices continue. These findings correspond well to those of Bremer et al (2008), who report an annual N balance of 15.4 kg per acre and P balance of 2.02 kg per acre across the LLB watershed strictly based on the responses of the WEBs producer survey. Rock and Mayer (2006) also estimated an N budget for the LLB sub-basin, and found an annual surplus of 21 kg per acre. However, several important export pathways were not included (e.g., the export of certain agricultural products), which may conceivably inflate their estimates. Other literature estimates (e.g., Janzen et al., 2003; Chambers et al., 2001; Drury et al., 2007) of nutrient status on Canadian agricultural land have found positive net balances, although usually of lower magnitude. Bremer et al (2008) suggest that the high N and P surplus' found in the LLB watershed reflect the intensity of agricultural use in the area.

Unsurprisingly, the overall N and P balances calculated in the second version of the baseline scenario (Baseline Scenario) are lower. The average annual N balance across the watershed is 10.11 kg N per acre, and the corresponding P balance is 2.09 kg P per acre. Both equate to a nearly 50% decrease. Lack of manure application is the sole cause of the difference in nutrient balance outcomes, highlighting the disproportionate role that manure plays in nutrient import to an agro-environmental system.

A significant degree of heterogeneity exists across the study area with respect to nutrient balance, especially in the BAU Baseline Scenario. Table 6.2 displays the N and P balance found on various fields assigned to different uses and management practices (i.e., manure vs. chemical fertilizer) in the BAU version. Regular manure application significantly contributes to the incidence of elevated residual (surplus) nutrient levels. As discussed in Chapter 3, certain fields in the LLB watershed have a history of regular manure application and as a consequence can have extremely high soil test nutrient levels. Additionally, higher baseline fertilizer application rates on irrigated fields relative to export from crop harvest result in higher residual nutrient levels compared to dryland fields. The risk of nutrient loss to surrounding environment increases as residual nutrients accumulate in the soil (De Jong et al., 2009). Field-level nutrient balances in the Baseline

Scenario correspond to the values cited for fields with 'Chemical' nutrient source in Table 6.2, the only difference being the number of acres.

Table 6.2. Field Level Average Annual Nitrogen and Phosphorus Balances in the BAU Baseline Scenario, By Crop Rotation, Soil Texture, and Nutrient Source (kg $acre^{-1}$).

Activity	Rotation	Texture	Nutrient Source	Acres	N Balance	P Balance
Cropping	Dryland	Coarse	Chemical	480	2.35	0.85
		Medium	Chemical	160	4.05	1.59
		-	Manure	680	38.18	7.24
Cropping	Irrigated 1	-	Chemical	280	17.10	2.08
		-	Manured	1,000	37.64	7.91
Cropping	Irrigated 2	-	Chemical	600	22.64	5.74
		-	Manured	840	37.68	8.49
			Category			
Pasture	Dryland	Coarse	Tame, Upland	120	4.41	2.52
		Medium	Tame, Upland	160	5.32	3.02
		-	Native, Riparian	840	0.76	-0.15
		-	Native, Upland	1,160	0.99	-0.07
Pasture	Irrigated	-	Tame, Riparian	80	2.69	2.19
		-	Tame, Upland	280	5.35	3.07
Total Watershed				6,680	18.70	3.93

6.2 BMP Scenario Results

Section 6.2 details the results for each of the BMP scenarios included in this analysis. For each scenario, both the economic (producer returns) and biophysical impacts are reported. A discussion of the causes and implications of each result is provided.

6.2.1 Results for Scenario 1: Alfalfa

The first set of alternative land use scenarios feature the inclusion of alfalfa in base annual crop rotations. Growing alfalfa, a perennial forage legume, is considered a BMP due to the potential for reduced chemical fertilizer application and the ability of perennial vegetation to promote soil organic matter accumulation. In Scenario 1a, a three year stand of alfalfa is inserted between annual crops in both irrigated rotations and the dryland rotation on cropped fields of the LLB watershed. This rotational BMP is therefore implemented on 4,040 acres of the 6,680 total acres of the study area. In the second version of the scenario, 1b, the extent of BMP implementation is limited to only dryland fields, which comprises 1320 acres of the watershed. Details of Scenario 1 strategies are provided in section 5.2.2.

Table 6.3 reports the findings of each version of Scenario 1 alongside that of the Baseline Scenario. As discussed previously, the Baseline Scenario is used for comparison due to the lack of manure application. The mean NPV of agricultural activity in the watershed in Scenario 1a is $12,466,043, a reduction of 15.05% from returns in the Baseline. The mean NPV in Scenario 1b is $15,771,110, an increase of 7.47%.

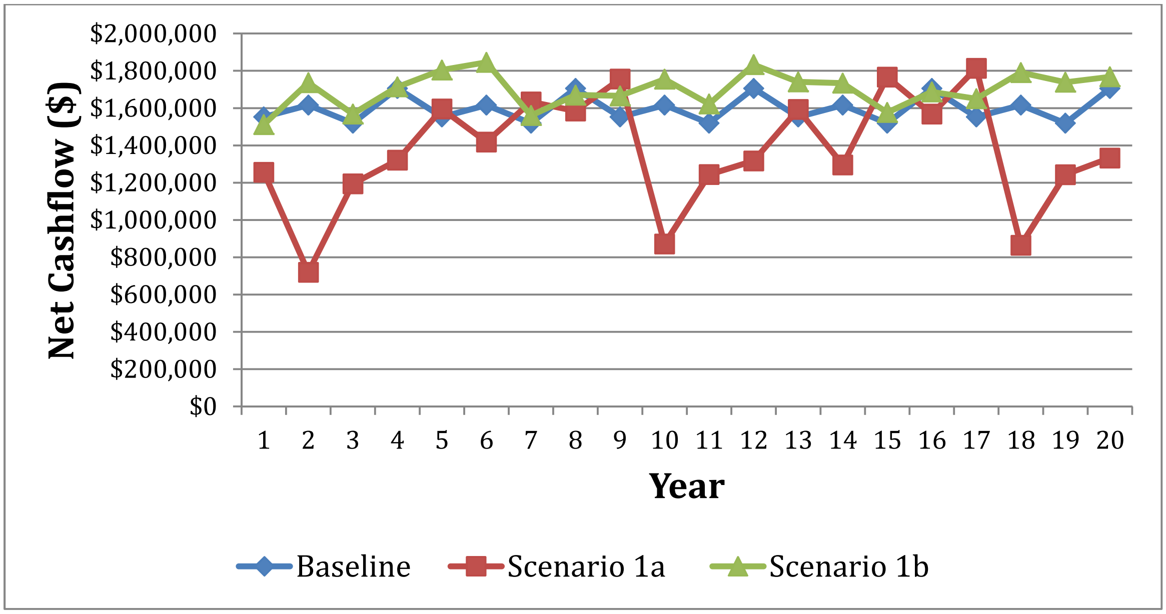
Table 6.3. Summary Economic and Biophysical Results, Baseline and Scenarios 1a and 1b.

	Baseline	**Scenario 1a**	**Scenario 1b**
Mean NPV ($) (*Total Watershed*) [a]	$14,675,018	$12,466,043	$15,771,110
Standard Deviation [b]	-	$204,675	$24,406
Mean NPV ($ acre^{-1})	$2,197	$1,866	$2,361
Mean Annualized NPV ($ acre^{-1})	$258	$219	$277
Percent Difference (Relative to Baseline)	-	-15.05%	7.47%
Mean Annual N Balance (kg acre^{-1})	10.11	2.87	9.34
Cumulative N Balance (Tonnes) (*Total Watershed*)	1,350	384	1,247
Standard Deviation (Tonnes)	-	22	4
Percent Difference (Relative to Baseline)	-	-71.61%	-7.60%
Mean Annual P Balance (kg acre^{-1})	2.09	0.94	2.22
Cumulative P Balance (Tonnes) (*Total Watershed*)	279	125	297
Standard Deviation (Tonnes)	-	2	0.8
Percent Difference (Relative to Baseline)	-	-55.02%	6.22%
Net SOC Change (Tonnes) (*Total Watershed*)	0	7,791	2,738

[a] NPV results are reported as the mean value of 1,000 simulations;
[b] Standard deviation is reported as a measure of variance of the simulation results.

The mean annualized NPV per acre is $219 and $277 for Scenarios 1a and 1b, respectively. Compared to the Baseline Scenario ($258), the decrease in returns generated by agricultural activity in Scenario 1a can be attributed to the opportunity costs of reducing the acreage devoted each year to high-value irrigated crops such as potatoes and sugar beets. However, when this rotational BMP is implemented only on dryland cropping fields, as in Scenario 1b, an increase in producer returns is observed. Because dryland cropping is not as productive as irrigated cropping, the opportunity costs of substituting in a lower value perennial forage such as alfalfa are lower. Instead, the benefits in terms of higher yields to subsequent crops following an alfalfa stand as well as the reduced need for fertilizer outweigh the opportunity costs and can increase returns. This finding is supported by Xie (2014) and Trautman (2012). Xie (2014) reported decreases in NPV for two of the three irrigated alfalfa BMP rotations modeled, ranging between 4 and 16%. The one rotation in which NPV increased, which featured potatoes, wheat, and canola, only increased by 4%. Trautman (2012), conversely, reported increases to mean NPV results due to alfalfa BMPs on each of the four representative farms modeled in four different soil zones. On the three dryland farms, mean NPV increased between 10 and 75%. This positive impact of including alfalfa in annual crop rotations is supported by the results of this study, since mean NPV also increases in the LLB watershed when the rotational BMP is applied only to dryland fields. Although the magnitude of the impact appears to be diminished in this analysis (7.47% increase), it is important to note that the effect is 'diluted' by the significant acreage that has not been subject to a land use change in Scenario 1b. Figure 6.2 tracks the modified net cashflow of both versions of Scenario 1 along with the Baseline Scenario. The periodic dip in net cashflow found in Scenario 1a (in years 2, 10, and 18) occurred because no potatoes were grown on irrigated fields throughout the watershed in those years and a majority of fields seeded to alfalfa are in the establishment (i.e., least productive) year.

Figure 6.2. Modified Net Cashflow, Total Watershed, Scenarios 1a, 1b, and Baseline.



The average annual N balance across the watershed in Scenario 1a was 2.87 kg N per acre, a 72% decrease from the baseline 2 scenario. The reduction in fertilizer application, coupled with increases in annual crop yield and therefore N exports, dramatically reduced the amount of residual N buildup in the soil. Although additional inputs came from the N_2 fixation capacity of alfalfa, stand length was short enough to prevent the buildup of excess N in the soil. In Scenario 1b, the annual N balance was also reduced with a average surplus of 9.34 kg per acre. Because the rotational BMP was implemented only on dryland fields, the benefit of reduced N surplus was reduced. The cumulative N balance over the 20 year timeframe was 384 and 1,247 tonnes for Scenarios 1a and 1b, respectively. Regarding annual P balance, only in Scenario 1a was the overall balance reduced. In this version the average annual P balance across the watershed is 0.94 kg per acre, leading to a cumulative total of 125 tonnes over 20 years. This represented a decrease of 55%. Interestingly, the field-specific P balances of dryland and irrigated cropland were impacted in the opposite manner. Whereas the average P surplus increased slightly on both coarse and medium dryland fields with respect to the Baseline Scenario, it decreased on irrigated fields. This result can partly be attributed to the difference in average nutrient removal rates between alfalfa hay and the alfalfa/grass mix. In alfalfa hay, 0.003 kg of P and removed for every kg of harvested crop material compared 0.0026 kg P is removed in the alfalfa/grass mix. Additionally, the amount of crop material removed in the harvest of alfalfa hay is relatively high compared to that of the alfalfa/grass mix and other annual crops, leading to a higher absolute amount of nutrient removed. For instance, the average alfalfa hay yield is 4,451 kg per acre. In contrast, the average yield of alfalfa/grass on dryland fields is 1,600 kg per acre, and the of yield of irrigated annual crops such as wheat and barley average between 2,000-2,300 kg per acre. Although the modeled P fertilizer rate is higher for alfalfa hay than several other common annual crops grown in the irrigated rotations (such as barley, canola, and wheat), the low (and negative) average P balances found suggest that in the long-term a higher P fertilizer rate may be necessary to maintain productivity.

The average annual P balance in Scenario 1b is 2.22 kg per acre, leading to a cumulative total of 297 tonnes over 20 years. This is a comparable amount to the P balance of the Baseline Scenario (2.09 kg per acre). A breakdown of both N and P average balances across different field types is displayed in Table 6.4. Figures 6.3 to 6.5 track the cumulative net surplus of N over the course of the modeling timeframe in both versions of Scenario 1 along with the Baseline Scenario.

Table 6.4. Field Level Average Annual Nitrogen and Phosphorus Balances on Cropped Fields in Scenarios 1a and 1b, By Crop Rotation and Soil Texture (kg acre^{-1}).

		Scenario 1a		**Scenario 1b**	
Rotation	**Texture**	N Balance	P Balance	N Balance	P Balance
Dryland	Coarse	0.12	1.50	0.12	1.50
	Medium	1.30	2.30	1.30	2.30
Irrigated 1[a]		2.69	-0.13	17.1	2.08
Irrigated 2[b]		4.96	1.76	22.24	5.74

[a] The first irrigated BMP rotation: SW-C-DW-DB-A-A-A
[b] The second irrigated BMP rotation: P-SW-SB-B-A-A-A-DW
Note: Nutrient balances on pasture fields remain the same as in the baseline scenarios, as outlined in Table 6.2.

Figure 6.3. Cumulative Nitrogen Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 1a (Tonnes N).

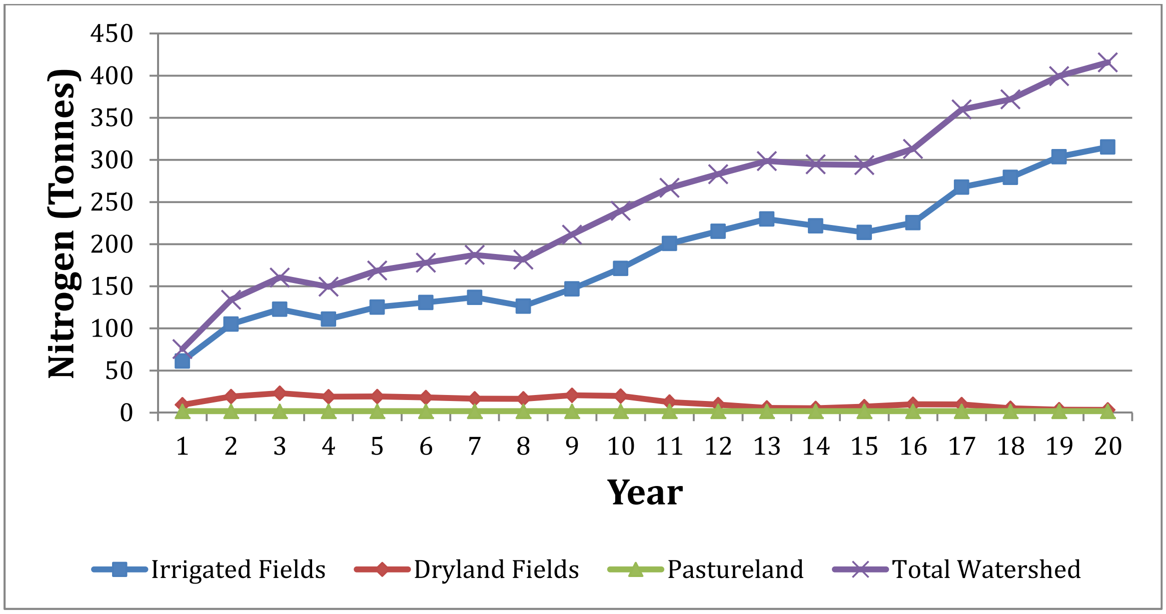


Figure 6.4. Cumulative Nitrogen Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 1b (Tonnes N).

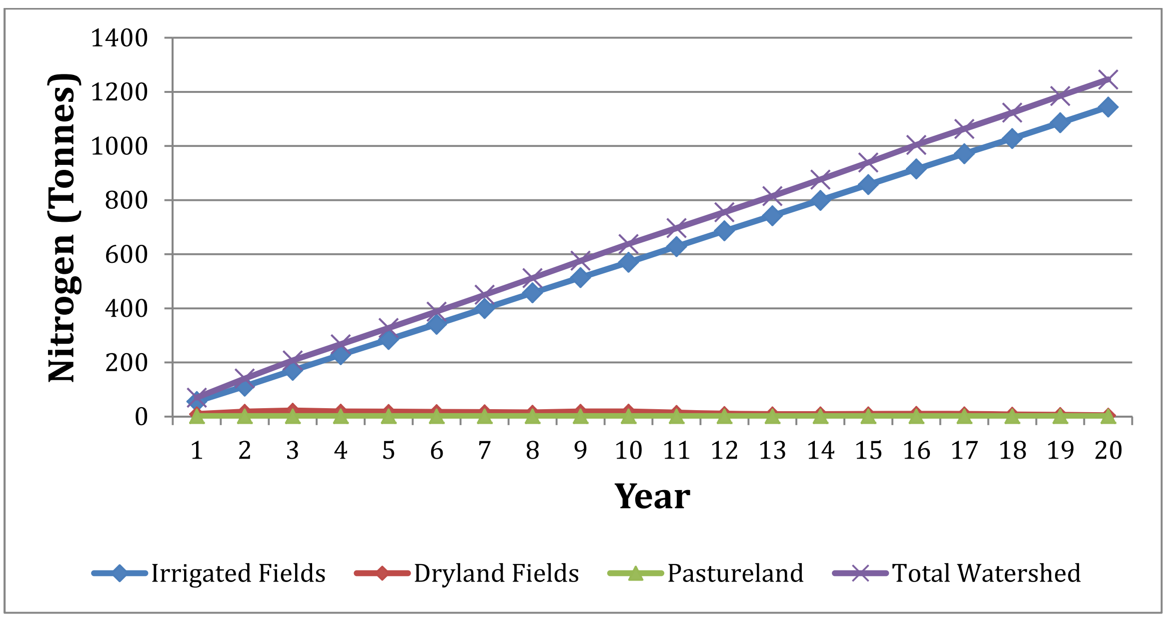
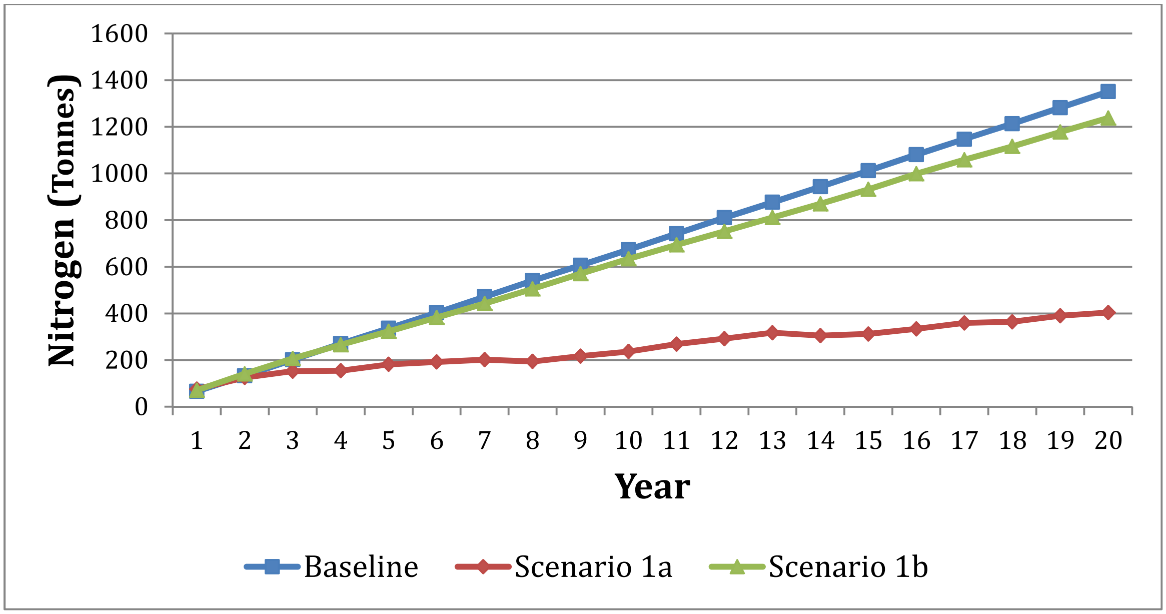


Figure 6.5. Comparison of Cumulative Nitrogen Balances in the Lower Little Bow Watershed Over the Modeling Timeframe Between Scenarios 1a, 1b, and the Baseline (Tonnes N).



The implementation of the alfalfa BMP in Scenario 1 had significant benefits in terms of an increase in net SOC as well. The contribution of two different changes in practices were accounted for: first, the inclusion of a perennial crop in three of seven (or eight) years in the newly implemented BMP rotations, and second, the proportional reduction in the number of summerfallow periods on dryland fields from once every four years to once every seven. The net gains in SOC storage over the course of the 20 year modeling timeframe are 7,791 tonnes in Scenario 1a and 2,738 tonnes in Scenario 1b. Figures 6.6 and 6.7 display the cumulative increase in SOC storage over 20 years in both scenarios. The effect of each different land management change is shown as well as the cumulative net change in SOC for the watershed as a whole. The use of perennials in rotation, both on coarse and medium textured fields, account for a majority of the gains in SOC, whereas the decrease in summerfallow comprises a much smaller portion of the total gain. SOC changes on medium and coarse textured fields[23] are reported separately as different parameter values are used in the calculations (see Table 5.24). Due to the

[23] Unlike other aspects of the analysis, the texture of irrigated cropping fields is accounted for in the calculation of SOC. Of the 2,720 irrigated acres, medium-textured soils were present in 1,440 acres and coarse-textured soils were present in 1,280 acres.

relatively short length of the modeling timeframe, the year over year increase in total SOC remains fairly constant and the accumulation of SOC is close to linear. Over a long period, however, it is expected that the marginal gains will decrease and that the relationship will approximate the logarithmic function described by McConkey et al (2014).

Figure 6.6. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 1a (Tonnes C).

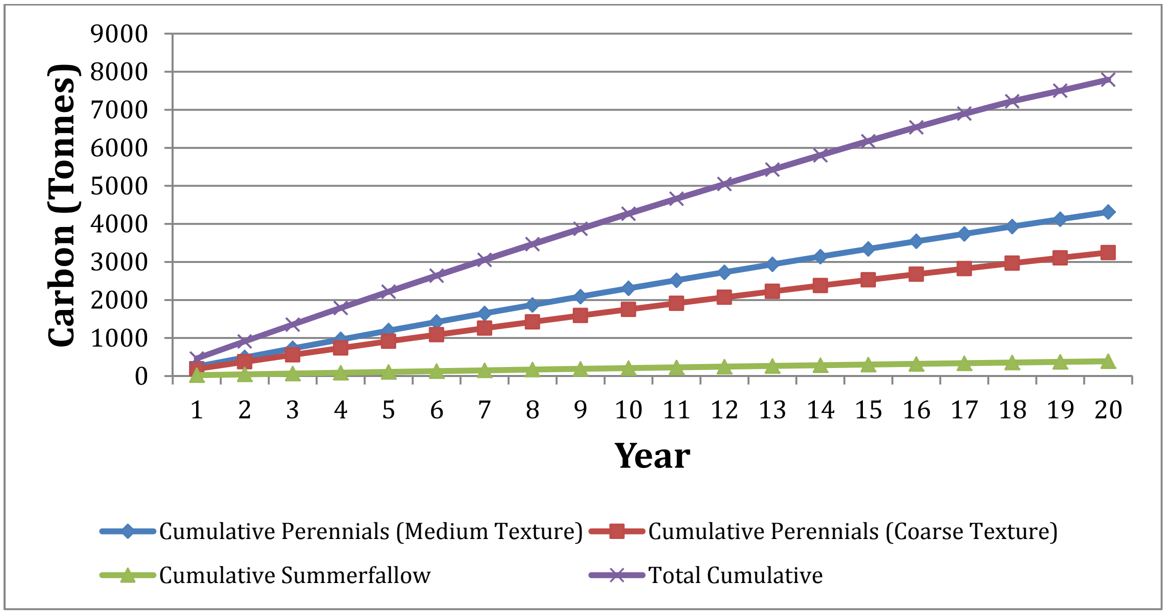
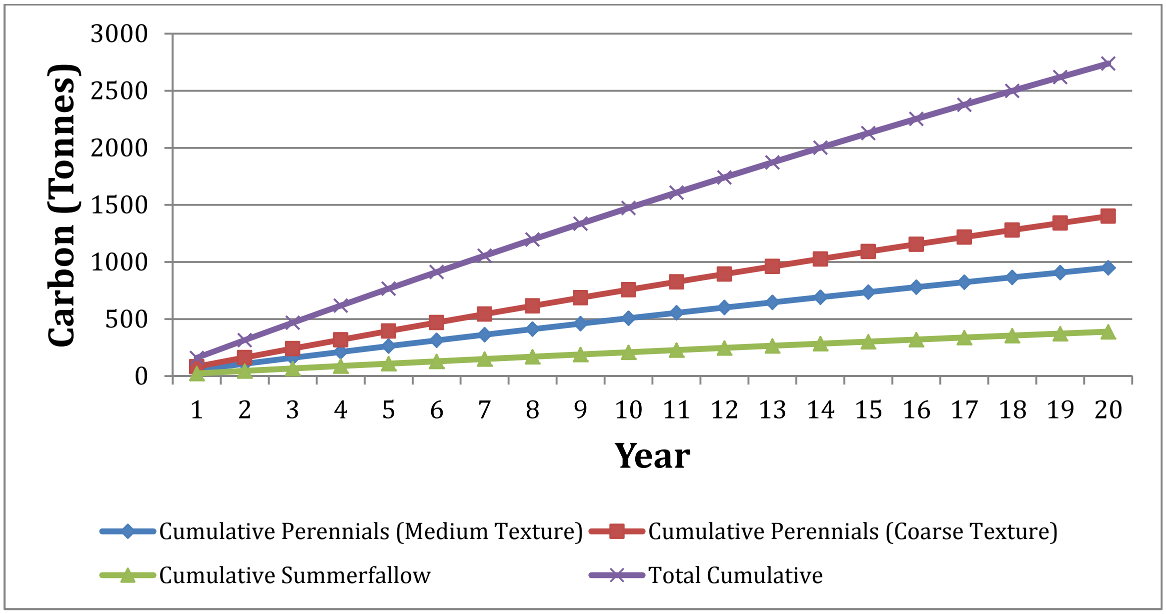


Figure 6.7. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 1b (Tonnes C).



6.2.2 Results for Scenario 2: Legume Green Manuring and Field Peas

A alternative set of rotational BMPs was also introduced to the LLB watershed in Scenario 2. In each of the three versions, fababeans were grown on irrigated fields and plowed into the soil as a legume green manure. Additionally, a different BMP rotation was adopted on dryland fields in each version. First, in Scenario 2a, field peas were introduced into the base dryland rotation. In Scenario 2b, red clover was grown as a legume green manure on dryland fields between annual crops. Finally, in the third version, Scenario 2c, both field peas and red clover were grown as part of an eight-year annual crop rotation on dryland fields.

Summary economic and biophysical results from the three versions of Scenario 2 are displayed in Table 6.5. The land use changes in each version of Scenario 2 resulted in a decrease to the mean NPV (total watershed) of agricultural activity in the LLB when compared to the Baseline Scenario. Mean NPV ranged from $12,486,672 to $12,770,583, and the annualized mean NPV per acre ranged from $220 (Scenario 2b) to $225 (Scenario 2a). Because the irrigated cropping and pastureland activities remained the same across the three versions, dryland cropping was the sole source of variation within

Scenario 2. As such, the mean annualized NPV per acre of dryland cropping was calculated to be $126, $108, and $131 for Scenario 2a, Scenario 2b, and Scenario 2c, respectively. Scenarios 2a and 2c result in increases to producer returns from dryland fields compared to the Baseline ($111 per acre), but Scenario 2b results in a slight decrease. Whereas field peas are able to provide both a source of revenue from crop sales as well as savings from N credits to subsequent crops, the N benefits from using red clover as a green manure was not able to offset the cost of forgoing revenue for that year as well as higher input costs for red clover compared to a summerfallow period. In Scenario 2c, the complete elimination of summerfallow from the rotation in favour of both field peas and red clover also had a positive impact on annualized mean NPV (20%), although slightly less so that of Scenario 2a. Trautman (2012) reported similar findings, as including field peas in crop rotations in the Dark Brown zone increased the annualized mean NPV by 33% and the addition of a legume green manure decreased NPV by 12%.

The use of fababean as a green manure had a negative impact on the economic returns to producers generated by cropping on irrigated fields. The mean annualized NPV per acre decreased from an average of $536 (between the two base irrigated rotations) to $481 (between the two BMP irrigated rotations), a decrease of 10.36%. This result is similar to that of Xie (2014), who observed negative impacts ranging from 10-31% depending on the rotation.

Figure 6.8 displays the net cashflow generated by cropping on dryland fields in each version of Scenario 2. Figure 6.9 shows the total watershed net cashflow in each version along with the that of the Baseline Scenario.

Table 6.5. Summary Economic and Biophysical Results, Baseline and Scenarios 2a, 2b, and 2c.

	Baseline	**Scenario 2a**	**Scenario 2b**	**Scenario 2c**
Mean NPV ($) (*Total Watershed*)	$14,675,018	$12,706,640	$12,486,672	$12,760,783
Standard Deviation	-	$63,092	$120,322	$63,050
Mean NPV ($ acre^{-1})	$2,197	$1,902	$1,869	$1,910
Mean Annualized NPV ($ acre^{-1})	$258	$223	$220	$224
Percent Difference (Relative to Baseline)	-	-13.41%	-14.91%	-13.04%
Mean Annual N Balance (kg acre^{-1})	10.11	12.63	14.80	13.64
Cumulative N Balance (Tonnes) (*Total Watershed*)	1,350	1,687	1,977	1,822
Standard Deviation (Tonnes)	-	4.70	4.73	4.40
Percent Difference (Relative to Baseline)	-	24.95%	46.41%	34.94%
Mean Annual P Balance (kg acre^{-1})	2.09	2.67	3.06	2.90
Cumulative P Balance (Tonnes) (*Total Watershed*)	279	356	409	388
Standard Deviation (kg)	-	167	0	118
Percent Difference (Relative to Baseline)	-	27.75%	46.42%	38.76%
Net SOC Change (Tonnes) (*Total Watershed*)	0	302	907	907

Note: In Scenario 2a, field peas are grown on dryland fields. In Scenario 2b, red clover is grown as a green manure, and in 2c, both crop are used in the rotation. Fababean is grown on irrigated fields in each scenario.

Figure 6.8. Modified Net Cashflow, Dryland Fields, Scenarios 2a, 2b, and 2c.

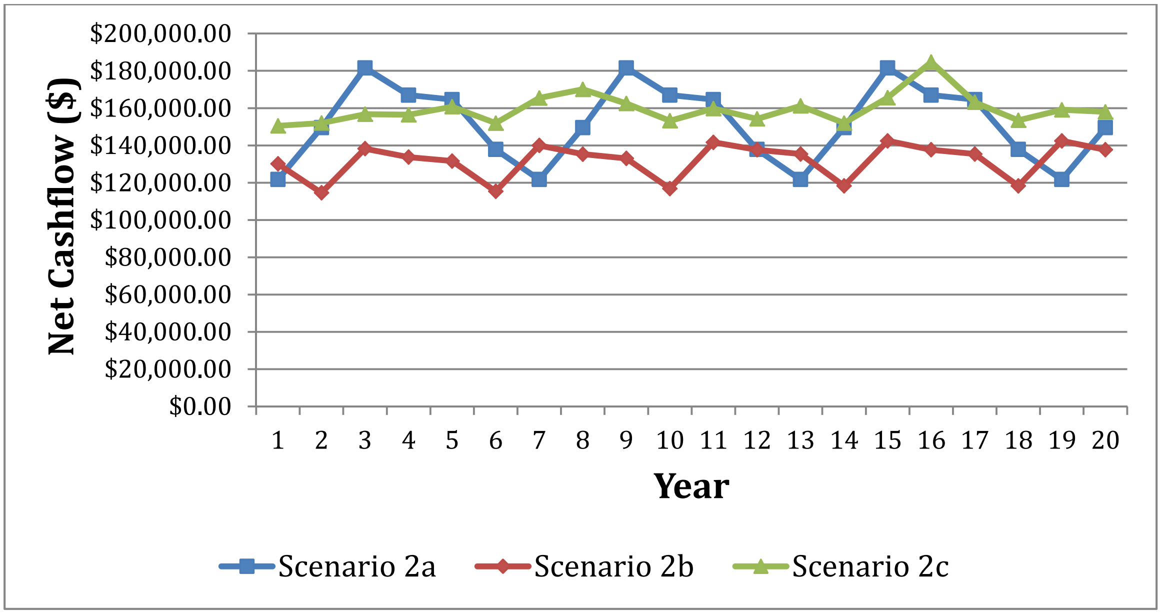
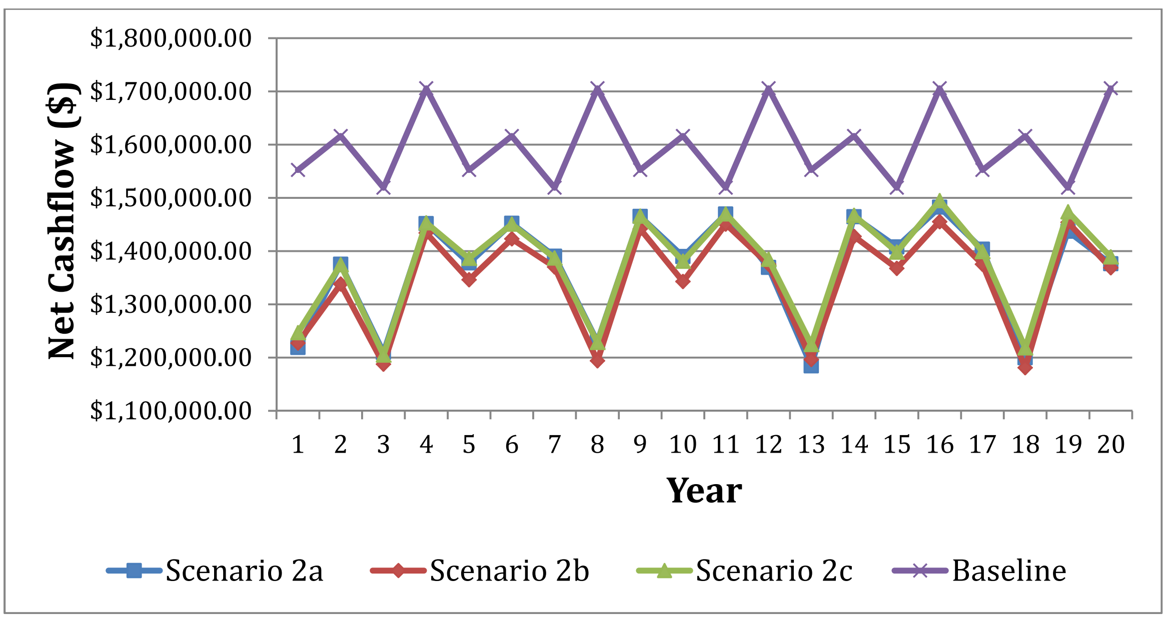


Figure 6.9. Modified Net Cashflow, Total Watershed, Scenarios 2a, 2b, 2c, and Baseline.



Overall, all three versions of Scenario 2 increased both the average annual N and P balance of the LLB watershed vis-à-vis the Baseline Scenario. The N balance increased by 25%, 46%, and 35% in Scenarios 2a, 2b, and 2c, respectively. These increases were primarily driven by growing fababean for the purposes of green manuring on irrigated fields. Shown in Table 6.6, the irrigated field-level N balances were 25.86 and 30.46 kg

N per acre for fields in the first and second irrigated rotations, respectively, increases from 17.10 and 22.64 kg per acre found in the Baseline (Table 6.2). A similar set of results was found for the watershed-level average P balance, with increases of 28%, 46%, and 39% for Scenarios 2a, 2b, and 2c, respectively. The elevated level of nutrients in the soil can be attributed to the green manure practice. When fababean is plowed down the N and P in the plant material is returned to the soil system rather than removed as a harvested product. In years when this practice occurs, the net gain in N and P is 87 and 12 kg per acre, respectively, as N is added via biological fixation from fababean, and P is added as from fertilizer application. The organic nutrients embedded within the plant material mineralize over time to become available to subsequent crops, which leads to a reduced need for chemical fertilizer application. However, the effect of this practice, at least in the short to medium term, is to enrich the soil with nutrients, leading to a positive impact on the nutrient balances calculated in this analysis. Historically, improved soil fertility on nutrient deficient soils, coupled with savings from reduced N fertilizer inputs (if the green manure crop is a legume), have been the reasons for crop producers to implement this practice (AARD, 1993). In the case of the LLB watershed, however, many of the agricultural fields used for cropping have high soil test nutrient levels due to a history of intensive use. As such, further enrichment of these soils through legume green manuring may not be an appropriate practice unless targeted at a field-specific level to fields with depleted soil nutrient levels. The price of N fertilizer or cost of manure application may also impact the feasibility of growing green manure crops in rotation, as obtaining N via biological fixation can be a cost-effective way to promote soil fertility. This possibility is investigated in the sensitivity analysis conducted later on in the chapter.

The variation in overall nutrient balance between the three versions of Scenario 2 is driven by differences in dryland crop rotation, as the land use and management practices on irrigated cropping and pastureland fields remain the same. In the Baseline Scenario, the average annual N balance on coarse and medium textured dryland fields used for cropping was 2.34 and 4.05 kg N per acre. The corresponding P balances were 0.85 and 1.59 kg per acre. Incorporating red clover as a green manure had the most significant impact, raising the average annual N balance to 8.64 and 10.35 kg per acre, and the average annual P balance to 3.07 and 4.06 kg per acre. Like fababean on irrigated fields, the N and P in red clover plant material is returned to the soil system instead of harvested, which has a positive impact on nutrient balance. The addition of field peas in Scenario 2a had the opposite effect to the N balance of dryland fields. Because fields peas are harvested like any other annual crop, the removal of plant material increases the total export of N compared to the green manuring practice. Additionally, low levels of supplemental N fertilizer are used when field peas are grown (because of the plants ability to fix N_2 in the soil as a legume), and in the following year when spring wheat is grown. This leads to a lower level of total N imports to the soil system. As such, the N balance on coarse and medium textured dryland fields in Scenario 2a is -1.18 and 0.67 kg N per acre, respectively. The finding of a nutrient deficit suggests that the expected benefit in terms of reduced fertilizer application may need to be pared back in order to maintain productivity over the long-term. However, on fields with high residual N levels, the use of field peas in annual crop rotations may help remediate this issue. The average annual P balance was comparable to the Baseline Scenario at 1.17 and 1.97 kg P per acre for coarse and medium textured fields. Lastly, the crop rotation employed on dryland fields in Scenario 2c strikes a balance between the two previous versions as both N and P levels remain fairly stable over the course of modeling timeframe. Nutrient addition via the green manure practice is offset by reduced fertilizer inputs in crops following both red clover and field peas. Tables 6.6 and 6.7 break down the average annual N and P balance on cropped fields by various field conditions and uses. Figure 6.10 tracks the cumulative addition of N in the watershed for each of the three versions of Scenario 2 in comparison with the Baseline Scenario.

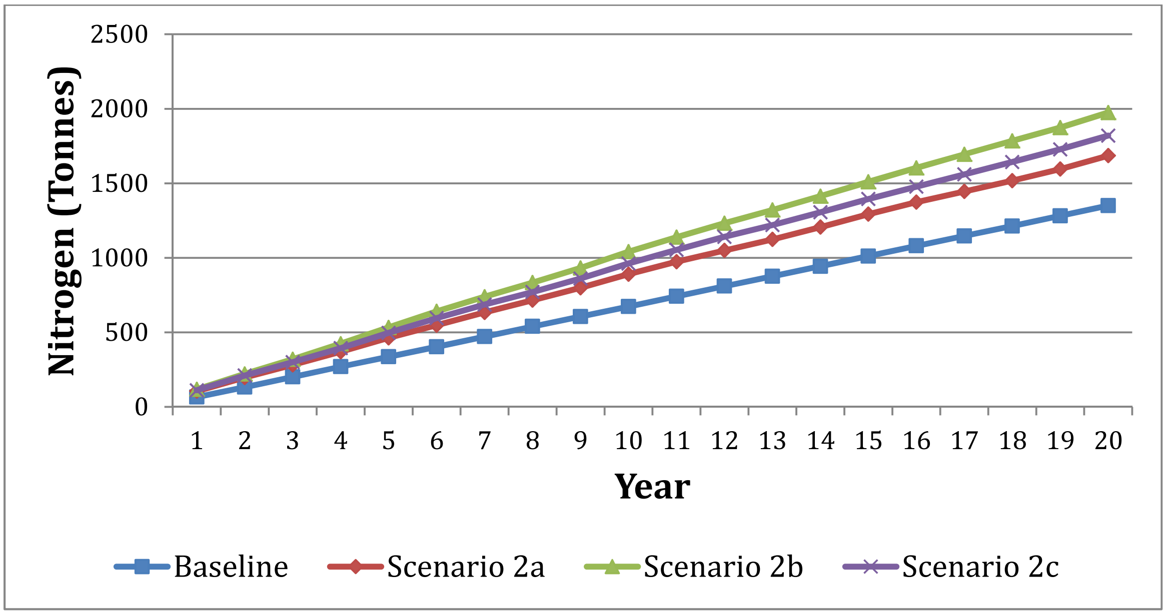
Table 6.6. Field Level Average Annual Nitrogen Balance on Cropped Fields in Scenarios 2a, 2b, and 2c, By Crop Rotation and Soil Texture (kg acre^{-1}).

Rotation	**Texture**	**Baseline**	**Scenario 2a**	**Scenario 2b**	**Scenario 2c**
Dryland	Coarse	2.35	-1.18	8.64	3.27
	Medium	4.05	0.67	10.35	3.42
Irrigated 1	-	17.10	25.86	25.86	17.10
Irrigated 2	-	22.64	30.46	30.46	22.64
Total Watershed		10.11	12.63	14.80	13.64

Table 6.7. Field Level Average Annual Phosphorus Balance on Cropped Fields in Scenarios 2a, 2b, and 2c, By Crop Rotation and Soil Texture (kg $acre^{-1}$).

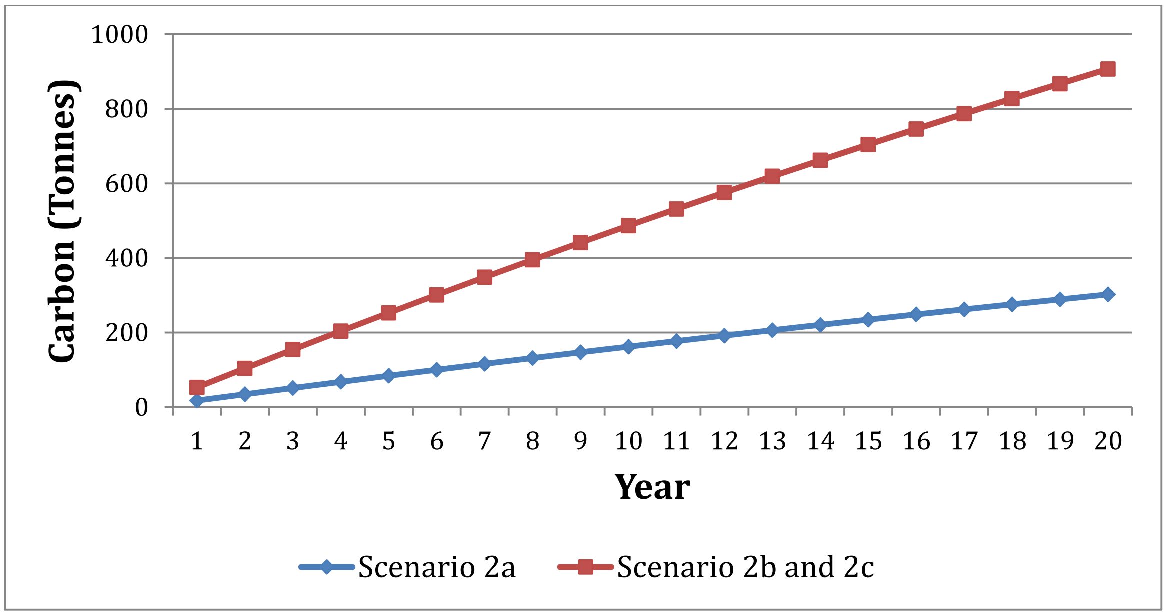
Rotation	Texture	Baseline	Scenario 2a	Scenario 2b	Scenario 2c
Dryland	Coarse	0.85	1.17	3.07	2.30
	Medium	1.59	1.97	4.06	3.24
Irrigated 1	-	2.08	3.61	3.61	3.61
Irrigated 2	-	5.74	6.74	6.74	6.74
Total Watershed		2.09	2.67	3.06	2.90

Figure 6.10. Comparison of Cumulative Nitrogen Balances in the Lower Little Bow Watershed Over the Modeling Timeframe Between Scenarios 2a, 2b, 2c, and the Baseline (Tonnes N).



A net increase in stored SOC was found in all three versions of Scenario 2. In each case, the impact on this parameter was modeled for only one change in land use practice: reduction (or elimination) of summerfallow. No perennial crops were grown in any of the three versions, nor were cropping fields converted to permanent cover. However, the proportion of summerfallow in each of the dryland BMP rotations was reduced or eliminated. In Scenario 2a, summerfallow was reduced from once every four years (1/4) to once every six (1/6). In both Scenarios 2b and 2c the practice of summerfallow was eliminated entirely. As such, the net gain in SOC for the LLB watershed in Scenario 2a was calculated to be 302 tonnes, and the net gain in Scenarios 2b and 2c is 907 tonnes. In all three cases the gains to SOC came from changes to practices on dryland fields only. Although some evidence suggests that green manure crops affect SOC levels (e.g., Lal, 2002; Fortuna et al., 2003), McConkey et al (2014) exclude this practice from Canada's GHG inventory reporting system due to lack of empirical data. As such, the effect of this practice is excluded from this analysis as well. Figure 6.11 tracks the cumulative changes to SOC calculated for each version of Scenario 2.

Figure 6.11. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenarios 2a, 2b, and 2c (Tonnes C).[a]



[a] Only changes due to reduction or elimination of summerfallow are reported; Scenarios 2b and 2c have the same change in net SOC.

6.2.3 Results for Scenario 3: Manure Management

Manure is applied to both irrigated and dryland cropped fields based on the N-requirements of spring wheat once every four years in Scenario 3. This strategy is designed to approximately satisfy the P requirements of crops over the four year period, with the use of supplemental chemical N fertilizer to fully meet N requirements. In comparison to the BAU Baseline Scenario, which approximates the typical manure application practices of producers in the LLB watershed, the strategy presented in Scenario 3 is considered a BMP because a lower total amount of manure is applied to each field. This reduces the possibility of nutrient buildup, particularly P, in the soil, which in turn reduces the risk of ground and surface water contamination.

The calculations used to determined the amount of manure applied can be found in section 5.2.4. On the 880 acres of coarse textured dryland used for cropping, 8.36 tonnes of manure per acre are applied when a field is seeded to red spring wheat. On the 440 acres of medium textured dryland fields, 8.95 tonnes of manure per acre are applied. A rate of 19.22 tonnes per acre is used on irrigated fields. On average, 15,893 tonnes of manure are used each year on cropped fields of the LLB watershed, which equates to a total of 317,852 tonnes over the 20 year modeling timeframe. This manure is spread over 4,040 acres of cropland. In comparison, a total of 298,000 tonnes are applied to 2,520 acres of cropland in the BAU Baseline Scenario.

Both baseline scenarios must be used in the evaluation of Scenario 3. Because the yield benefits associated with manure application cannot be accounted for in the BAU Baseline, a comparison of economic outcomes between the two scenarios would not be valid. However, the economic impacts of the manure BMP can be evaluated using the primary Baseline Scenario as the reference case. Conversely, an assessment of nutrient balance outcomes can be done using both baseline scenarios as a point of comparison, which will help illustrate the disproportionate impact of manure application.

Table 6.8 presents the summary economic results of Scenario 3 and the Baseline. As manure is not applied in the Baseline, and crop nutrient requirements are met solely through chemical fertilizer application, the incorporation of non-nutrient yield benefits of manure to the comparison is justified. As such, the mean NPV of agricultural activity in the LLB watershed rises 7.80% to $15,819,066 over the 20-year period. This increase is driven by two factors. First, the non-nutrient crop yield benefit of manure application increases yields by 1-5%, which increases revenue to crop producers. This benefit is derived from improvements to certain soil properties, such as increased soil organic matter or better moisture-holding capacity (Lupwayi et al., 2005). Second, as demonstrated in the comparison between the two baseline scenarios, manure is a less expensive source of nutrients than chemical fertilizer. Therefore, the substitution of a portion of N and P inputs from chemical fertilizer to manure will decrease total nutrient costs and increase the economic returns generated. The mean annualized NPV per acre increases by $20, from $258 in the Baseline Scenario to $278 in Scenario 3. Figure 6.12 displays the modified net cashflow found for each year of the modeling timeframe in both the Baseline and Scenario 3. A positive net cashflow is generated each year, with minor variation due to the number of acres requiring manure application and thus increased cost.

Table 6.8. Summary Economic Results, Baseline and Scenario 3.

	Baseline	**Scenario 3**
Mean NPV ($) (*Total Watershed*)	$14,675,018	$15,819,066
Standard Deviation	-	$54,033
Mean NPV ($ acre^{-1})	$2,197	$2,368
Mean Annualized NPV ($ acre^{-1})	$258	$278
Percent Difference (Relative to Baseline)	-	7.80%

Figure 6.12. Modified Net Cashflow, Total Watershed, Scenario 3 and the Baseline.

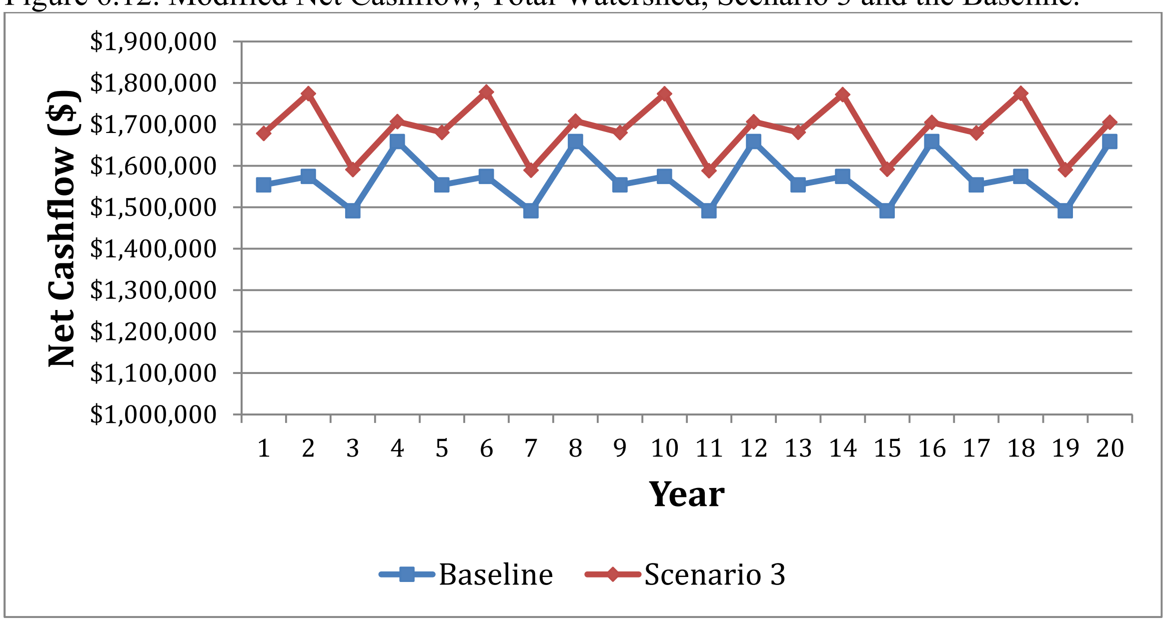


Table 6.9. Summary Biophysical Results, BAU Baseline, Baseline, and Scenario 3.

	BAU Baseline	Baseline	Scenario 3
Mean Annual N Balance (kg acre^{-1})	18.70	10.11	21.73
Cumulative N Balance (Tonnes) (*Total Watershed*)	2,498	1,350	2,904
Standard Deviation	-	-	2,931
Percent Difference (Relative to BAU Baseline)	-	-	16.21%
Percent Difference (Relative to Baseline)	-	-	114.97%
Mean Annual P Balance (kg acre^{-1})	3.93	2.09	2.21
Cumulative P Balance (Tonnes) (*Total Watershed*)	525	279	295
Standard Deviation (kg)	-	-	554
Percent Difference (Relative to BAU Baseline)		-	-43.76%
Percent Difference (Relative to Baseline)	-	-	5.75%
Net SOC Change (Tonnes) (*Total Watershed*)	0		n/a

Averaged across the study area, the implementation of the manure application BMP increased the annual and cumulative N balance compared to both baseline scenarios. Compared to the BAU Baseline Scenario, the average annual N balance increased 16.21%, from 18.70 to 21.73 kg N per acre. Importantly, however, the manure used in Scenario 3 was spread over a larger land base. While the total amount of manure used increased slightly, pulling the watershed average higher, the variation in N balance at a field level decreased substantially. As shown in Table 6.10, manured fields in the BAU Baseline receive an annual surplus of approximately 38 kg of N per acre (both dryland and irrigated); however, this surplus drops to between 2-4 kg N on dryland fields and 17-23kg N on irrigated fields when manure is not used. The BMP employed in Scenario 3 reduces the N surplus found on manured dryland fields to 12.05 and 14.45 kg N for coarse and medium textured fields. This balance, however, is higher than non-manured fields in the BAU Baseline. Notably, annual N balance increased by 3.14 and 8.24 kg per acre on fields used for irrigated 1 and irrigated 2 rotations, respectively, compared to irrigated fields receiving manure in the BAU Baseline.

When compared to the N balance outcomes of the practices employed in the BAU Baseline, the manure BMP was successful in reducing N surplus on dryland fields initially targeted for manure application. However, increases to the annual N balance were found on every other field type. Manure is not applied in the Baseline Scenario, and as such the N balance found in Scenario 3 is more than double (115% greater).

When interpreting the implications of the N balances calculated in this analysis, it is important to note the significance of the NH_4-N retention factors assumed when calculating nutrient availability from manure. Per Table 5.10 (section 5.2.4.2), loss of NH_4-N via volatilization to the atmosphere is highly dependent on method of manure application and weather conditions at the time. For average conditions, and if incorporated into the soil within one day, 75% of available NH_4-N is retained on dryland conditions. For irrigated fields, the retention rate decreases to 65%. Although the volatilization of NH_4-N (into NH_3) can lead to other environmental issues[24], the proportion not retained in the soil is not at risk of leaching into ground or surface water. However, the total of N contained in the manure source, including all NH_4-N, is included in the calculation of N balance. In reality, of the 10 kg of N present in each tonne of cattle manure (2.6 kg of which is NH_4-N), 0.65 kg are lost to the atmosphere when used on dryland fields. Similarly, 0.91 kg are lost when applied to irrigated fields. When calculated for Scenario 3, this reveals that a total of 275 tonnes of N considered as part of the cumulative watershed N surplus is in fact not added to the soil system. This value is 254 tonnes N in the BAU Baseline Scenario. While removing these values would moderate the surplus of N found in both scenarios to some extent, it is important to keep in mind that these results are meant to be indicative of N that can potentially be lost to the overall environment, which includes the atmosphere. Therefore, in a broad sense, the inclusion of volatilized NH_4-N is valid when evaluating total environmental impact.

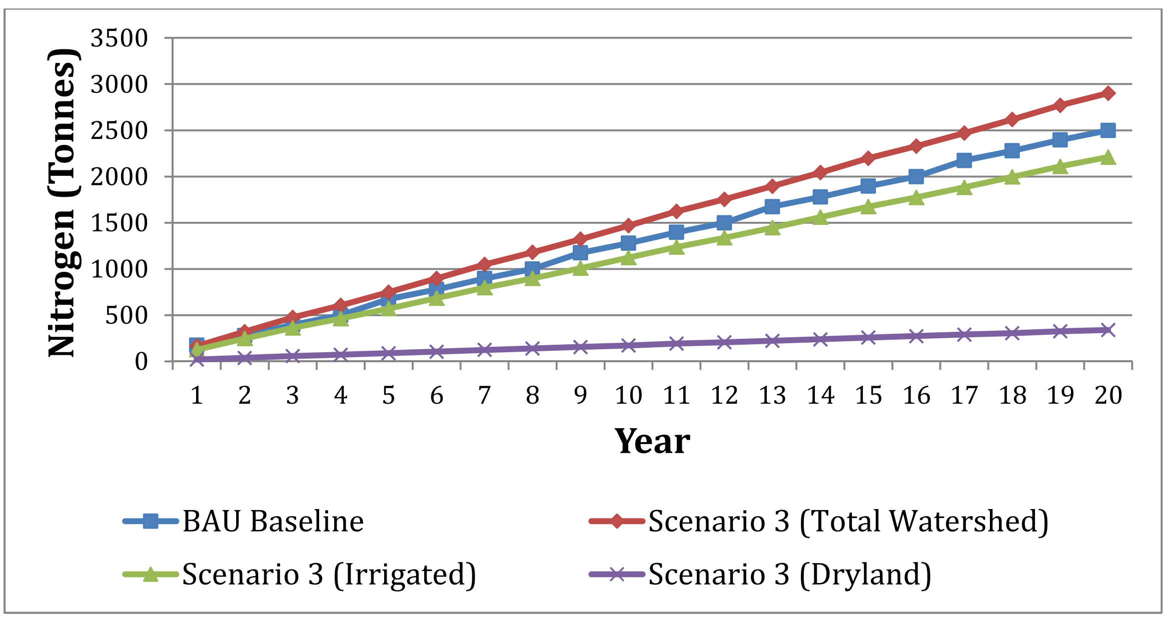
[24] For instance, NH_3 leads to acidification and eutrophication of water bodies when combined with water in the atmosphere or deposited on water bodies (Sutton et al., 2011).

Table 6.10. Field Level Average Annual Nitrogen Balance on Cropped Fields [a] in the BAU Baseline and Scenario 3, By Crop Rotation, Soil Texture, and Nutrient Source (kg $acre^{-1}$).[b]

Rotation	Texture	Nutrient Source	BAU Baseline	Scenario 3
Dryland	Coarse	Chemical	2.35	-
	Medium	Chemical	4.05	-
	-	Manured	38.18	-
	Coarse	Manured	-	12.05
	Medium	Manured	-	14.45
Irrigated 1	-	Chemical	17.1	-
	-	Manured	37.64	40.78
Irrigated 2	-	Chemical	22.64	-
	-	Manured	37.68	45.92
Total Watershed			18.70	21.73

[a] N balance on pasture fields remains the same between the BAU Baseline, Baseline, and Scenario 3, and can be found in Table 6.2.
[b] Average N balance on cropped fields of the Baseline Scenario is identical to that of fields reported for 'Chemical' in the BAU Baseline.

Figure 6.13. Comparison of Cumulative Nitrogen Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 3 and the BAU Baseline (Tonnes N).



Accumulation of P in soil is a notable concern when manure is applied to meet the N requirements of crops each year (Sharpley et al., 1998). As such, the manure management BMP in Scenario 3 was constructed to reduce total manure application and better match the P requirements of the annual crops grown in the modeled rotations. This strategy was successful in reducing the annual surplus of P averaged across the watershed, from 3.93 kg P per acre in the BAU Baseline to 2.21 kg P per acre. This 44% reduction resulted in a decrease of 230 tonnes of P imported into the study area over the 20 year period. Table 6.11 breaks down the change in P balance from the BAU Baseline Scenario to Scenario 3 across the different land uses. On both coarse and medium textured dryland fields the P balance was comparable to dryland fields in the BAU Baseline scenario that did not receive manure application. However, the P balance was dramatically reduced when compared to dryland fields receiving manure in the baseline. Similarly, the surplus of P was reduced on manured fields for both irrigated rotations, with a reduction of 3.25 kg per acre on fields used for the first irrigated rotation and 4.28 kg per acre on fields seeded to the second rotation. Interestingly, the P balance was also reduced vis-à-vis non-manured fields allocated to the second irrigated rotation.

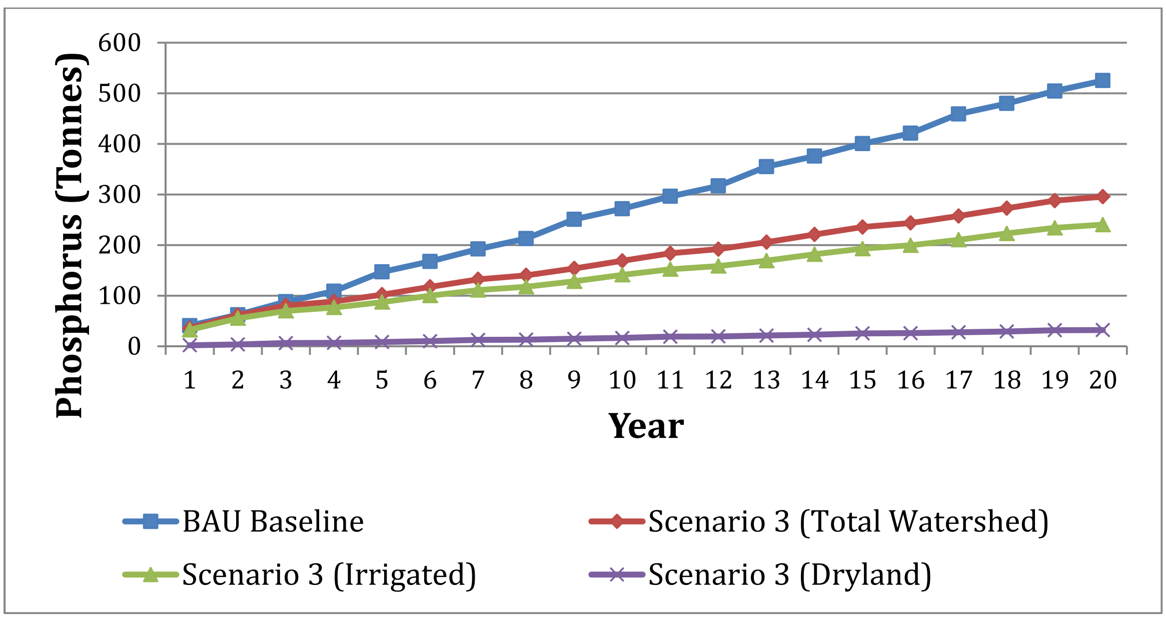
The manure management BMP was successful in controlling P surplus when compared to the Baseline Scenario as well, demonstrating that the application of manure can be done without accumulating high amounts of P in the soil. Overall, P balance across the LLB watershed increased only 6%. In Table 6.11, the P balance values on non-manured fields of BAU Baseline are equivalent to that of the Baseline. Only on irrigated 1 fields is the P balance increased by the BMP manure application strategy. The remaining cropped fields, including coarse dryland, medium dryland, and irrigated 2, show in fact a decrease in annual P surplus. If a crop producer were looking to incorporate manure application on fields that had previously only received chemical fertilizer, the results below demonstrate that soil test P levels can be managed if the proper application practice is put in place. The incidence of high existing soil test P levels in the LLB watershed may also be remedied by a modification to manure application levels.

Table 6.11. Field Level Average Annual Phosphorus Balance on Cropped Fields [a] in the BAU Baseline and Scenario 3, By Crop Rotation, Soil Texture, and Nutrient Source (kg acre^{-1}).[b]

Rotation	Texture	Nutrient Source	BAU Baseline	Scenario 3
Dryland	Coarse	Chemical	0.85	-
	Medium	Chemical	1.59	-
	-	Manured	7.24	-
	Coarse	Manured	-	0.66
	Medium	Manured	-	1.42
Irrigated 1	-	Chemical	2.08	-
	-	Manured	7.91	4.66
Irrigated 2	-	Non-Manured	5.74	-
	-	Chemical	8.49	4.21
Total Watershed			3.93	2.21

[a] P balance on pasture fields remains the same between the BAU Baseline, Baseline and Scenario 3, and can be found in Table 6.2.
[b] Average P balance on cropped fields of the Baseline Scenario is identical to that of fields reported for 'Chemical' in the BAU Baseline.

Figure 6.14. Comparison of Cumulative Phosphorus Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 3 and the BAU Baseline (Tonnes P).



As previously discussed, the net change in SOC storage is not calculated for this scenario. Manure application can locally increase the SOC levels (Helgason et al., 2005), which has private economic benefits in certain regions and conditions (Belcher et al., 2003). However, these benefits are largely private and are partially accounted for in the calculation of non-nutrient crop yield increases. The primary focus of this aspect of the analysis is on changes in net C sequestration from the atmosphere, which can be considered a public benefit. In this context, the small to non-existent change in net atmospheric C stock makes the calculation of this parameter inconsequential for Scenario 3.

6.2.4 Results for Scenario 4: Permanent Forage

Scenario 4 is designed to reduce the intensity of agriculture in the LLB watershed primarily through the conversion of land use. In place of annual cropping, which currently makes up a majority of acreage in the watershed (61%), perennial crops are grown instead. Alfalfa, timothy grass, and an alfalfa-grass mix are grown on cropped land as feedstock in Scenario 4a. Stands of alfalfa and timothy hay are rotated on irrigated fields, whereas a permanent cover of alfalfa-grass mix is grown on dryland fields. In Scenario 4b, cropped irrigated fields remain planted to annual crops (as in the baseline scenarios), but dryland cropped fields are 'retired' to the perennial vegetation of the alfalfa-grass mix. Lastly, in Scenario 4c, only dryland fields considered to be marginally productive for annual crop production are converted to perennial vegetation. Fields considered to be marginally productive were those within soil polygons marked as having a 'T' (slope) limitation[25]. A total of seven fields encompassing 560 acres were identified as being marginally productive.

The conversion away from annual cropping leads to decreases in the mean NPV of agricultural activity in the watershed in every version of Scenario 4. The complete conversion of all annual crops in Scenario 4a leads to a 70% decline in mean NPV compared to the Baseline Scenario, with an annualized mean NPV per acre of $77. The elimination of high-value annual crops such as canola, potatoes, and wheat carries a significant opportunity cost to producers. Additionally, many of the benefits of integrating a perennial legume such as alfalfa into an annual crop rotation are not realized when full conversion takes place. For instance, the yield increases to subsequent annual

[25] This limitation indicates the presence of slope steep enough to incur a risk of water erosion or to limit full cultivation. See section 5.2.5.3 for a detailed explanation.

crops following an alfalfa stand are a significant benefit and go a long way to offsetting the opportunity cost of reducing the proportion of the aforementioned high-value annual crops. In Scenario 4b, where annual cropping is preserved on irrigated fields, a significant portion of the NPV is retained. The mean total NPV in Scenario 4b is $13,046,283, with an annualized per acre value of $229. This represents an 11% decrease in producer returns generated. Finally, targeting only marginal fields in the LLB watershed for perennial cropping lessens the negative private economic impacts even more. The mean total NPV of Scenario 4c is $14,307,976, with an annualized per acre value of $252. It should be reiterated (see section 5.2.5.3) that the negative impacts to producers may be overstated in Scenario 4c, as the marginal cropland targeted for retirement may in fact be less productive than other dryland fields, which is not accounted for here. In this case, the opportunity cost of conversion would be lower.

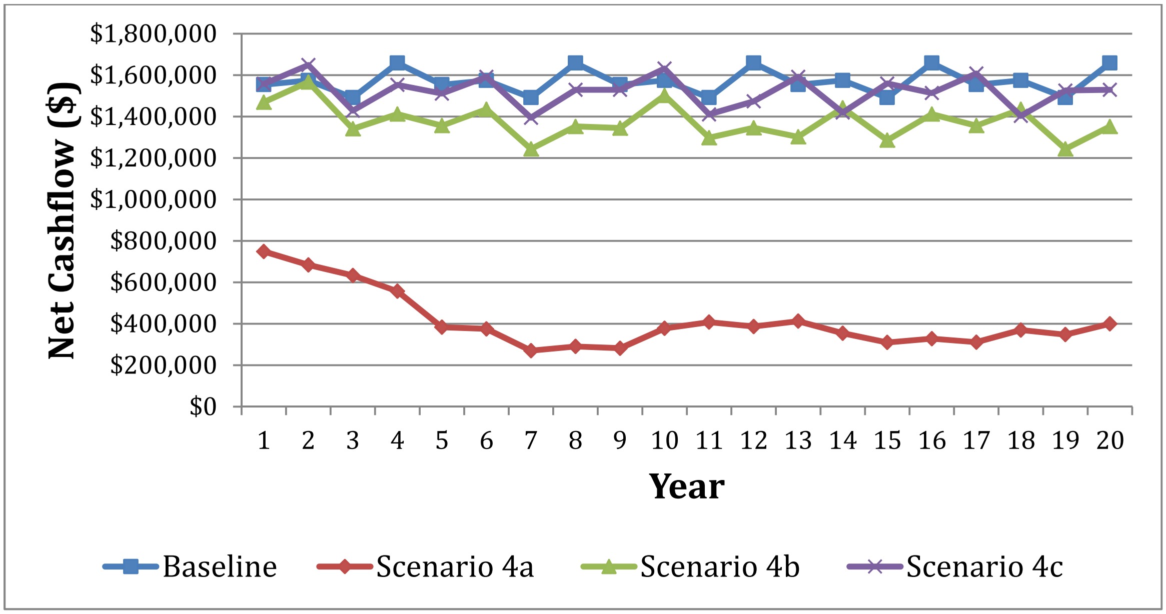
Table 6.12 presents the economic and biophysical summary results of each version of Scenario 4, and Figure 6.15 compares the modified net cashflow observed in each version of Scenario 4 along with that of the Baseline Scenario. The net cashflow observed in Scenario 4a decreases steadily over the first five years as annual crops are phased out and replaced with perennials, before stabilizing around $300,000 on an annual basis. In Scenario 4b, net cashflow remains below that of the Baseline in each of the 20 years, whereas in certain years the net cashflow of Scenario 4c eclipses that of the Baseline. However, when canola makes up a larger proportion of the dryland field acreage the net cashflow of the Baseline clearly exceeds that of Scenario 4c. For instance, 400 of the 1320 total dryland cropping acres are seeded to canola every four years, corresponding to the spikes in net cashflow that can be observed in Figure 6.15. Conversely, the maximum acreage devoted to canola on dryland fields in any given year of Scenario 4c is 320 acres.

Table 6.12. Summary Economic and Biophysical Results, Baseline and Scenarios 4a, 4b, and 4c.

	Baseline	**Scenario 4a**	**Scenario 4b**	**Scenario 4c**
Mean NPV ($) (*Total Watershed*)	$14,675,018	$4,358,590	$13,046,283	$14,307,976
Standard Deviation [a]	-	$7,542	0	0
Mean NPV ($ $acre^{-1}$)	$2,197	$653	$1,953	$2,142
Mean Annualized NPV ($ $acre^{-1}$)	$258	$77	$229	$252
Percent Difference (Relative to Baseline)	-	-70.30%	-11.10%	-2.50%
Mean Annual N Balance (kg $acre^{-1}$)	10.11	4.71	10.42	10.22
Cumulative N Balance (Tonnes) (*Total Watershed*)	1,350	629	1,392	1,365
Standard Deviation [a] (Tonnes)	-	19.47	2.86	1.41
Percent Difference (Relative to Baseline)	-	-53.41%	3.07%	1.08%
Mean Annual P Balance (kg $acre^{-1}$)	2.09	0.57	2.32	2.18
Cumulative P Balance (Tonnes) (*Total Watershed*)	279	76	309	291
Standard Deviation [a]	-	0	0	0
Percent Difference (Relative to Baseline)	-	-72.92%	10.81%	4.31%
Net SOC Change (Tonnes) (*Total Watershed*)	0	18,179	6,389	2,834

[a] The standard deviation of mean NPV is only reported for Scenario 4a, the only version requiring simulation due to the variability of alfalfa hay N_2 fixation and thus subsequent fertilizer input to timothy hay. Calculation of revenues and costs from crop production in the Baseline, Scenario 4b, and Scenario 4c do not require random draws from a distribution of values. However, the variability of fixation levels in alfalfa/grass mix does necessitate the simulation of N balance outcomes in the latter two scenarios. P Balance remains unaffected, and thus no measure of variation is reported.

Figure 6.15. Modified Net Cashflow, Total Watershed, Scenarios 4a, 4b, 4c, and Baseline.



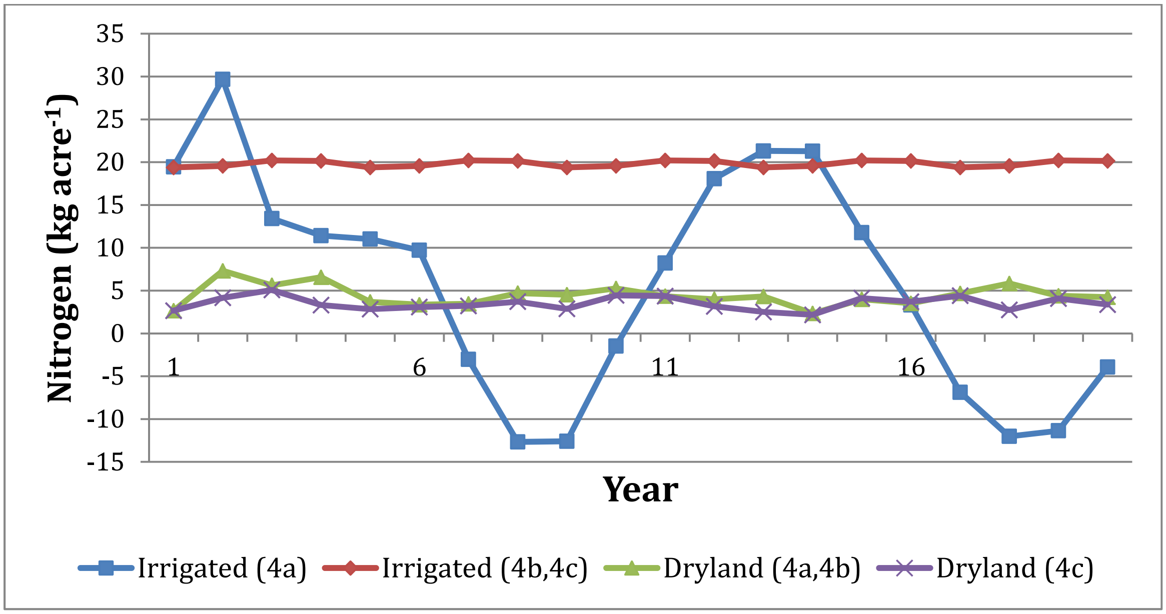
As expected, the full conversion modeled in Scenario4a away from more intensive forms of land use (i.e., annual cropping) decreased the overall watershed balance of both N and P. However, this decrease was exclusively driven by change in land use on irrigated fields. When left unchanged, as in Scenarios 4b and 4c, watershed N and P balance increased slightly. The mean annual N balance on a per acre basis was 4.71 kg in Scenario 4a, a decrease of 53%. In Scenarios 4b and 4c, N surplus increased 3% and 1%, respectively.

While the above values reflect the average net difference between the import and export of N to and from the LLB watershed, the dissimilarity of N balance over time between the versions of Scenario 4, particularly on irrigated fields, is obscured. Figure 6.16 displays the change in average annual N balance from year to year over the 20 year timeframe on cropped fields in each version of Scenario 4. The most variation exists on irrigated fields in Scenario 4a. This fluctuation in N balance exists due to the rotation between alfalfa and timothy grass. In years where alfalfa is more prevalent, particularly later on in the stand life when biological fixation levels are still high and the amount of harvested material begins to decline, total import of N exceeds export. In some years (e.g., 13 and 14), the residual soil N level on irrigated fields in Scenario 4a surpasses that of the other versions as well as the Baseline Scenario. As irrigated fields are transitioned to timothy hay, the export of N via harvested crop material begins to exceed N import. This trend is driven by the fertilizer reduction benefits from alfalfa in the first three years of the timothy stand (see section 5.2.5.6), as fertilizer applications are reduced because of the high residual soil N levels left by the perennial legume. On average, however, the annual surplus of N on irrigated fields of Scenario 4a is 5.74 kg per acre, down from an average of 19.82 kg per acre in the Baseline and Scenarios 4b and 4c (see Table 6.13).

The 1320 acres of dryland fields are converted to permanent cover in both Scenarios 4a and 4b, resulting in an average annual N balance of 4.52 kg per acre. However, when only marginal fields are converted to this use the per acre N surplus declines to 3.5 kg[26]. In the Baseline Scenario, dryland fields have an average annual balance of 2.35 and 4.05 kg N per acre (for coarse and medium fields, respectively), lower than that of dryland fields in Scenarios 4a and 4b. This reveals that, under the specified assumptions of the baseline chemical fertilizer application levels and the contribution of biological fixation from the alfalfa/grass mix to soil nutrient status, this BMP does not decrease residual soil N levels on dryland fields over the long term when compared to annual cropping. However, in a case where a field had high existing levels of residual N (such as NO_3-N), introducing a perennial alfalfa/grass mix may still be effective in mitigating the associated environmental risks (Entz et al., 2001). This is because a perennial legume plant such as alfalfa can effectively 'mine' the soil for plant available N, reaching deeper into the soil profile than an annual crop due to more expansive root growth.

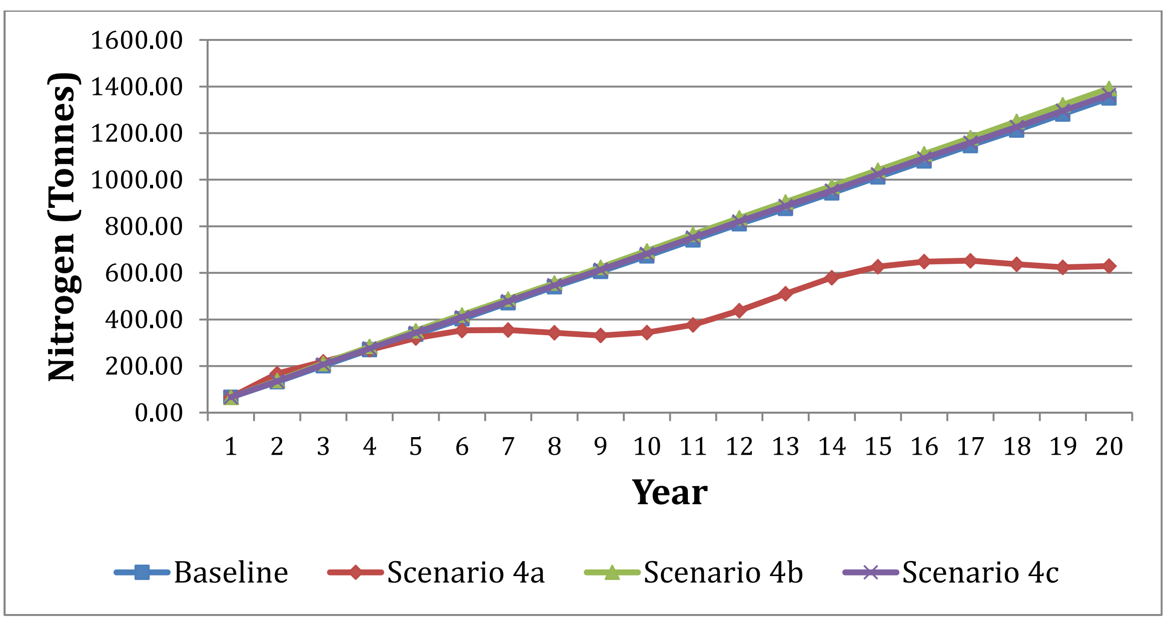
[26] This value averages across both coarse (600 acres) and medium (160 acres) dryland fields under baseline annual crop rotations receiving baseline fertilizer application rates, and marginal (560 acres) dryland fields with alfalfa-grass mix land use.

Figure 6.16. Average Annual Nitrogen Balance on Irrigated and Dryland Cropped Fields, Scenarios 4a, 4b, and 4c (kg acre^{-1}).[a]



[a] The N balance of irrigated fields in Scenario 4a differs from that of irrigated fields in Scenarios 4b and 4c. Conversely, the N balance of dryland fields in Scenario 4c differs from that of dryland fields in Scenarios 4a and 4b.

Figure 6.17. Comparison of Cumulative Nitrogen Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenarios 4a, 4b, 4c, and the Baseline Scenario (Tonnes N).



A similar pattern was found for P in each version of Scenario 4. Compared to the Baseline, the net surplus of P into the watershed was dramatically reduced (73%) in Scenario 4a, but moderately increased in both Scenario 4b and 4c (11% and 14%). Annual balance averaged 0.57, 2.32, and 2.37 kg P per acre across the study area in Scenarios 4a, 4b, and 4c, respectively. In general, annual P balance across all land uses is comparable to the levels found in the Baseline (Table 6.13). One notable exception is irrigated fields in Scenario 4a, which post a slight P deficiency averaged over time. In the long run this would make the application of supplemental P fertilizer above baseline levels necessary to maintain productivity, unless this BMP was introduced to a field with a high pre-existing soil test P level. The cumulative total watershed P balance for each version of Scenario 4 is displayed and compared to the Baseline in Figure 6.18.

Figure 6.18. Comparison of Cumulative Phosphorus Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenarios 4a, 4b, 4c, and the Baseline Scenario (Tonnes P).

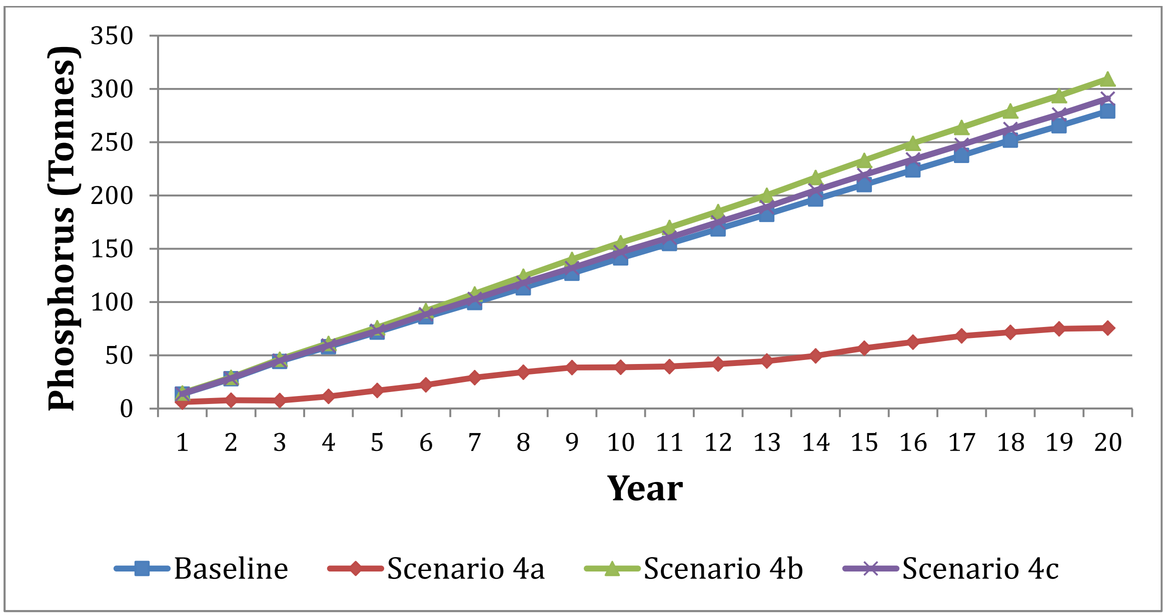


Table 6.13. Field Level Average Annual Nitrogen Balance on Cropped Fields in Scenarios 4a, 4b, 4c, and the Baseline, By Crop Rotation and Soil Texture (kg $acre^{-1}$).

Rotation	Texture	Baseline 2	Scenario 4a	Scenario 4b	Scenario 4c
Dryland	Coarse	2.35	4.52	4.52	2.35
	Medium	4.05	4.52	4.52	4.05
	Marginal	-	-	-	4.52
Irrigated 1	-	17.10	5.74	17.10	17.10
Irrigated 2	-	22.64	5.74	22.64	22.64
Total Watershed		10.11	4.71	10.42	10.22

Table 6.14. Field Level Average Annual Phosphorus Balance on Cropped Fields in Scenarios 4a, 4b, 4c, and the Baseline, By Crop Rotation and Soil Texture (kg $acre^{-1}$).

Rotation	Texture	Baseline 2	Scenario 4a	Scenario 4b	Scenario 4c
Dryland	Coarse	0.85	2.24	2.24	0.85
	Medium	1.59	2.24	2.24	1.59
	Marginal	-	-	-	2.24
Irrigated 1	-	2.08	-0.28	2.08	2.08
Irrigated 2	-	5.74	-0.28	5.74	5.74
Total Watershed		2.09	0.57	2.32	2.18

In terms of increased SOC storage, Scenario 4a provides the most benefits. The net SOC gain is calculated to be 18,179 tonnes, the majority of which is due to the displacement of annual cropping in favour of permanent forage. Of this total, 10,063 tonnes of C are gained on medium textured fields and 7,209 tonnes C on coarse textured fields (all of which were formerly used for annual cropping). The elimination of summerfallow provides 907 tonnes of increased SOC on dryland fields. The total net accumulation of SOC in the LLB watershed in Scenario 4a is displayed in Figure 6.19, along with the cumulative change due to each land use change considered.

The SOC sequestration benefits are tempered in Scenario 4b, but a net gain is still found. Overall, the total net gain in SOC is 6,389 tonnes. In this case, because coarse textured fields make up a majority of (formerly annually cropped) dryland fields, the gain from fields with coarse textured soils is greater than the gain from medium textured fields. Irrigated fields assigned to annual cropping in the baseline are not converted to permanent forage in Scenario 4b, the primary reason for diminished SOC storage benefits. Figure 6.20 presents the cumulative change in SOC in the LLB watershed in Scenario 4b.

The land use change in Scenario 4c results in the lowest gain in SOC among the three versions of Scenario 4. The net gain is calculated to be 2,834 tonnes. Figure 6.21 details the net gain in SOC in Scenario 4c, and Figure 6.22 compares the each of the three versions in terms of this biophysical parameter. Table 6.15 summarizes the amount of SOC gain each land use change is responsible for in each version of Scenario 4.

Figure 6.19. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 4a (Tonnes C).

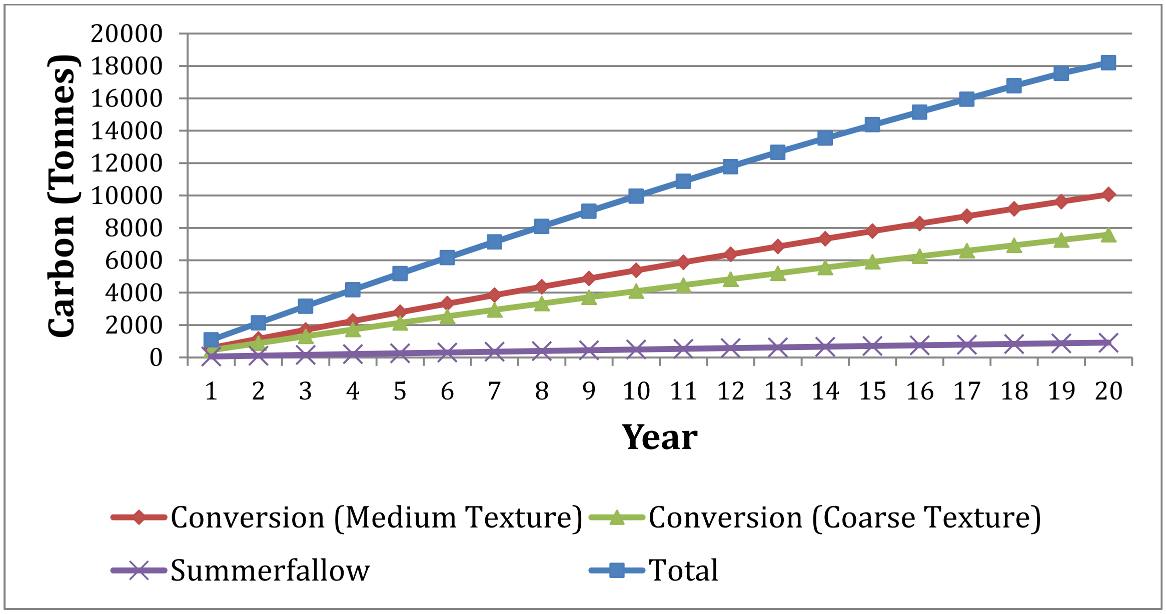


Figure 6.20. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 4b (Tonnes C).

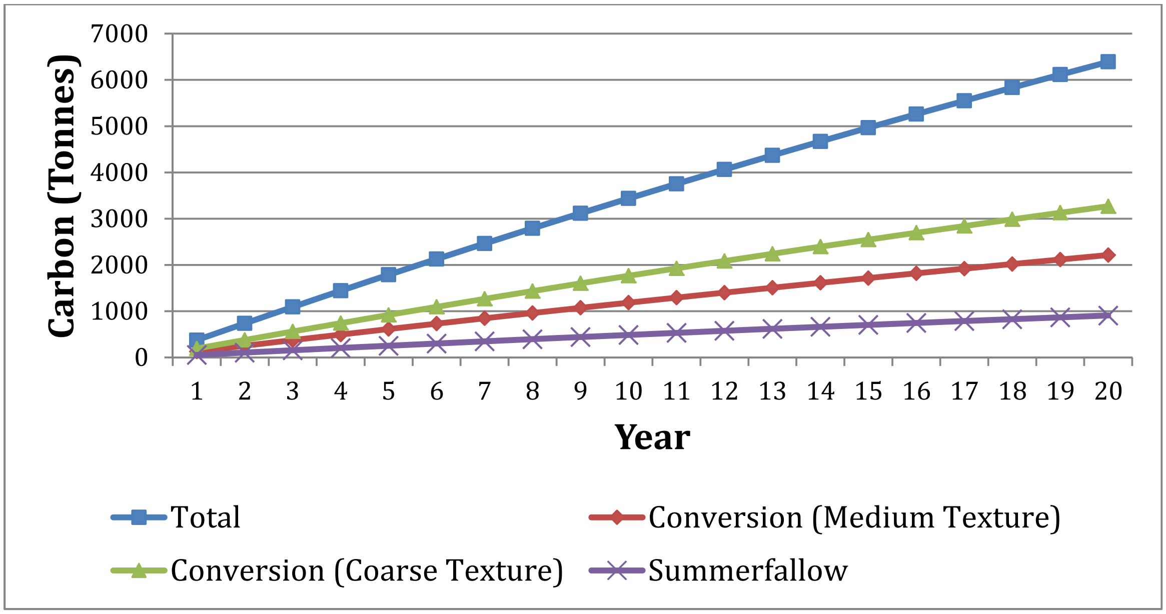


Figure 6.21. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 4c (Tonnes C).

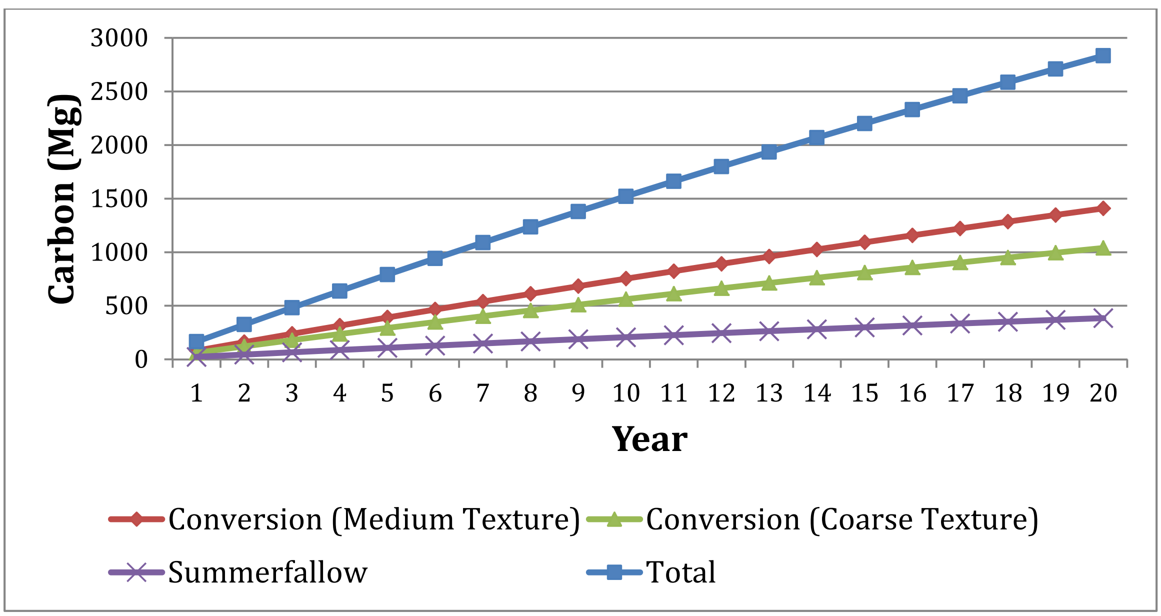


Figure 6.22. Comparison of Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenarios 4a, 4b, and 4c (Tonnes C).

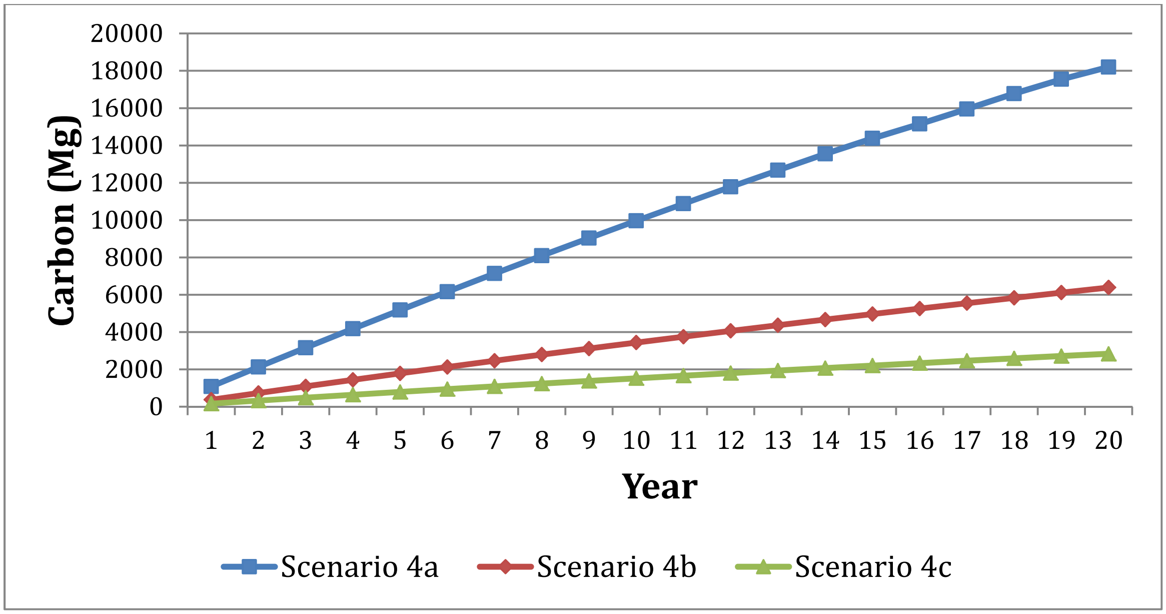


Table 6.15. Net Changes in Soil Organic Carbon From Various Land Use Changes in Scenarios 4a, 4b, and 4c (Tonnes C).

Land Use Change	**Scenario 4a**	**Scenario 4b**	**Scenario 4c**
Reduction of Summerfallow	907	907	385
Conversion to Annual (Medium Texture)	10,063	2,214	1,409
Conversion to Annual (Coarse Texture)	7,209	3,269	1,040
Total Watershed	18,179	6,389	2,834

6.2.5 Results for Scenario 5: Cultivation of Pastureland

The final set of scenarios concerning changes to land use in the LLB watershed involve the conversion to more intensive forms of use. In Scenario 5a, fields initially designated as pastureland in the Baseline Scenario are converted into use for annual crop cultivation. This land use change takes place on all pasture fields (2640 acres), including both native range and tame grass, and base annual crop rotations are implemented. In Scenario 5b, the crop rotations are modified to include the rotational alfalfa BMP introduced in Scenario 1a. These two versions of Scenario 5 are evaluated using the Baseline Scenario as the reference case, since manure application is not present.

A third version, Scenario 5c, is also developed for comparison with the BAU Baseline Scenario. In this case, the baseline manure application practice is maintained on select fields originally allocated to cropping. However, like Scenario 5a, only chemical fertilizers are used on pastureland converted to cropping. Using this strategy, the amount of manure applied across the watershed is held constant between Scenario 5c and the BAU Baseline.

Scenario 5 was modeled in order to provide a contrasting state of land use in the LLB watershed relative to both the Baseline Scenario and other BMP scenarios, where more input-intensive activities are utilized. As such, Scenario 5 is not a 'BMP' scenario in the same sense as Scenarios 1-4, but rather an alternative land use scenario. This hypothetical set of land use changes will be of benefit in the formation of tradeoff curves, which are presented in section 6.4.

Table 6.16 presents the summary economic and biophysical results of Scenarios 5a and 5b along with that of the Baseline Scenario. Economic returns generated by agricultural activity in the watershed increase in both versions. This result is unsurprising, as pasture activities (low return per acre) are substituted for annual cropping (higher return per acre). The mean NPV of activity in the watershed increased 34% and 11% in Scenarios 5a and 5b, respectively. The mean annualized NPV per acre for the two versions was $344 and $287, compared to $258 per acre in the baseline. Although the introduction of alfalfa in Scenario 5b reduced the proportion of higher-value annual crop on original cropland fields, the influx of higher returns from the conversion of pastureland along with the benefits of growing alfalfa (e.g., reduced fertilizer costs) more than offset this opportunity cost. The growth in producer returns (11%) is similar to that of Scenario 1b (7%).

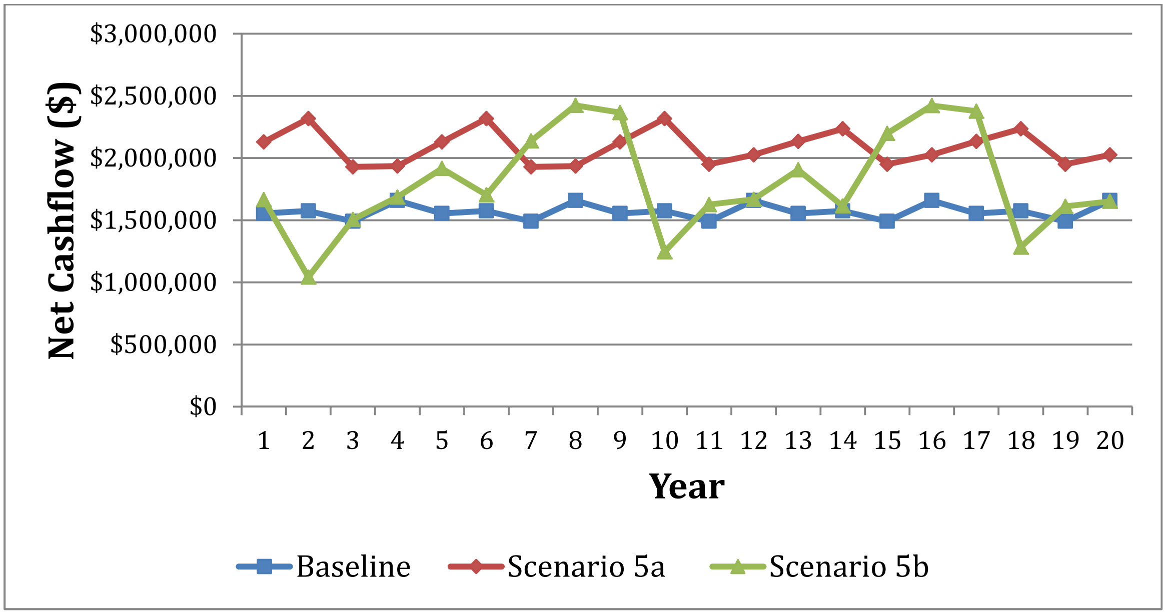
Figure 6.23 displays the modified net cashflow in each year of the modeling timeframe for Scenarios 5a and 5b compared to the Baseline Scenario. Net cashflow in Scenario 5a is higher than in the Baseline for each year, consistently remaining between $1,900,000 and $2,400,000. Year to year variability is highest in Scenario 5b, ranging from a high of $2,424,000 to a low of $1,042,000. This trend in net cashflow mirrors that of Scenario 1a, where periodic dips (e.g., years 2, 10, and 18) are a result of a lack of potatoes planted on irrigated fields in those years.

Table 6.16. Summary Economic and Biophysical Results, Baseline and Scenarios 5a and 5b.

	Baseline	**Scenario 5a**	**Scenario 5b**
Mean NPV ($) (*Total Watershed*)	$14,675,018	$19,562,221	$16,340,385
Standard Deviation [a]	-	0	$246,505
Mean NPV ($ acre^{-1})	$2,197	$2,928	$2,446
Mean Annualized NPV ($ acre^{-1})	$258	$344	$287
Percent Difference (Relative to Baseline)	-	33.59%	11.35%
Mean Annual N Balance (kg acre^{-1})	10.11	12.04	1.40
Cumulative N Balance (Tonnes) (*Total Watershed*)	1,350	1,608	187
Standard Deviation (Tonnes)	-	0	26
Percent Difference (Relative to Baseline)	-	19.08%	-86.15%
Mean Annual P Balance (kg acre^{-1})	2.09	2.50	1.51
Cumulative P Balance (Tonnes) (*Total Watershed*)	279	335	201
Standard Deviation (Tonnes)	-	0	3.22
Percent Difference (Relative to Baseline)	-	19.82%	-27.75%
Net Soil Carbon Change (Tonnes) (*Total Watershed*)	0	-12,690	698

[a] Similar to other BMP scenarios involving alfalfa, standard deviation is included as a measure of variance when reporting the results of Scenario 5b due to the distribution of values for N fixation and crop yield increase.

Figure 6.23. Modified Net Cashflow, Total Watershed, Scenarios 5a, 5b, and the Baseline.



Scenario 5c also increased the returns to producers generated by agricultural activity compared to the Baseline Scenario. The mean NPV increased from $14,929,842 to $19,870,447, a gain of 33%. On an annualized per acre basis, mean NPV increased by $86.

Nutrient balances were impacted in different directions by Scenarios 5a and 5b. The net surplus of N increased by 19% in Scenario 5a vis-à-vis the Baseline, resulting in an additional 258 tonnes of N entering the LLB watershed agro-environmental system. This result was expected since the land use changes modeled involved the conversion of a large acreage of native range, which receives no supplemental nutrients, to a more intensive use (annual cropping) with increased use of chemical fertilizer. However, the net surplus of N decreased by 86% when the alfalfa crop rotation BMP was introduced in Scenario 5b. As seen in Scenario 1a, the inclusion of alfalfa, a perennial forage legume, in annual crop rotations has a dramatic impact on N balance. Export of N rises as crop yields increase, and the imports via biological fixation are offset by reductions in fertilizer application. Additionally, despite relatively low levels of N fertilizer addition, dry tame grass pasture has a higher baseline N balance than cropped dryland fields because N exports via animal weight gain are extremely low. As such, the conversion of dryland tame grass fields (280 acres) to cropland can in fact reduce the N surplus found. However, the difference is minimal and therefore unlikely to substantially change the environmental risk.

Table 6.17. Summary Economic and Biophysical Results, BAU Baseline and Scenario 5c.

	BAU Baseline	**Scenario 5c**
Mean NPV ($) (*Total Watershed*)	$14,929,842	$19,870,447
Standard Deviation	-	0
Mean NPV ($ acre^{-1})	$2,235	$2,975
Mean Annualized NPV ($ acre^{-1})	$263	$349
Percent Difference (Relative to Baseline)	-	33.09%
Mean Annual N Balance (kg acre^{-1})	18.70	20.63
Cumulative N Balance (Tonnes) (*Total Watershed*)	2,498	2,756
Standard Deviation	-	0
Percent Difference (Relative to Baseline)	-	10.31%
Mean Annual P Balance (kg acre^{-1})	3.93	4.32
Cumulative P Balance (Tonnes) (*Total Watershed*)	525	578
Standard Deviation	-	0
Percent Difference (Relative to Baseline)	-	10.01%
Net Soil Carbon Change (Tonnes) (*Total Watershed*)	0	-12,690

The year to year change in average N balance for irrigated and dryland fields of both Scenario 5a and 5b is shown in Figure 6.24. The inclusion of alfalfa increases the variability of annual N balance on both irrigated and dryland fields. Because all pastureland is converted, the total number of acres devoted to dryland and irrigated cropping in this scenario is 3,600 and 3,080, respectively. It is important to note that the trends displayed in Figure 6.24 are averages of the two rotations on irrigated fields, as well as averages of the textural categories on dryland fields. Figure 6.25 shows the cumulative N surplus over the 20 year modeling timeframe. Table 6.18 reports the specific average annual N and P balance found for each category of field.

Figure 6.24. Average Annual Nitrogen Balance on Irrigated and Dryland Cropped Fields, Scenarios 5a and 5b (kg acre^{-1}).

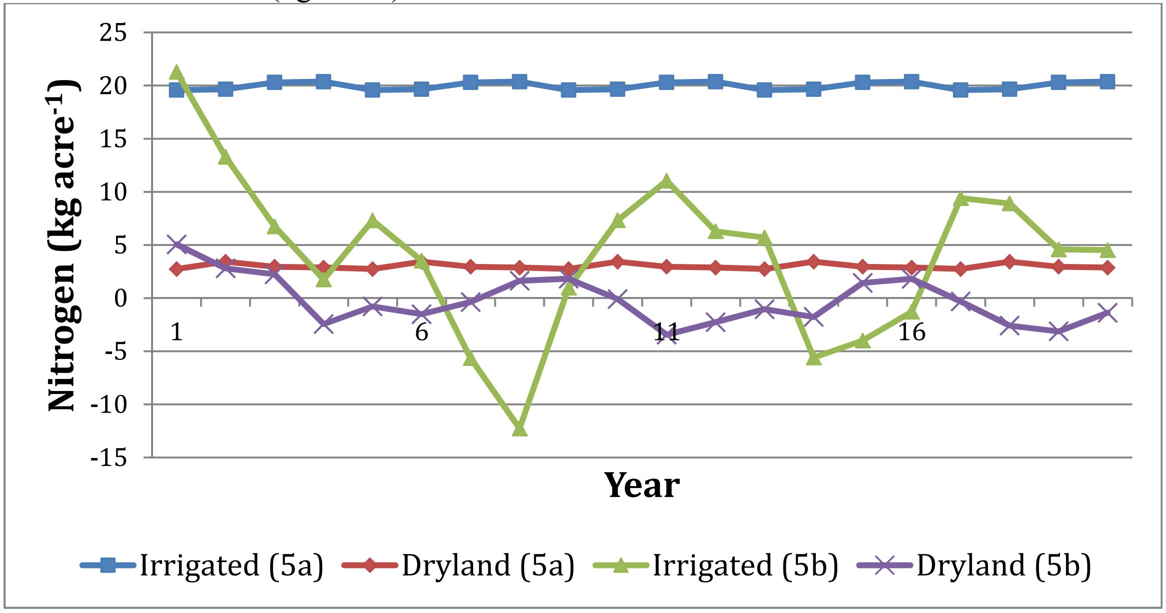
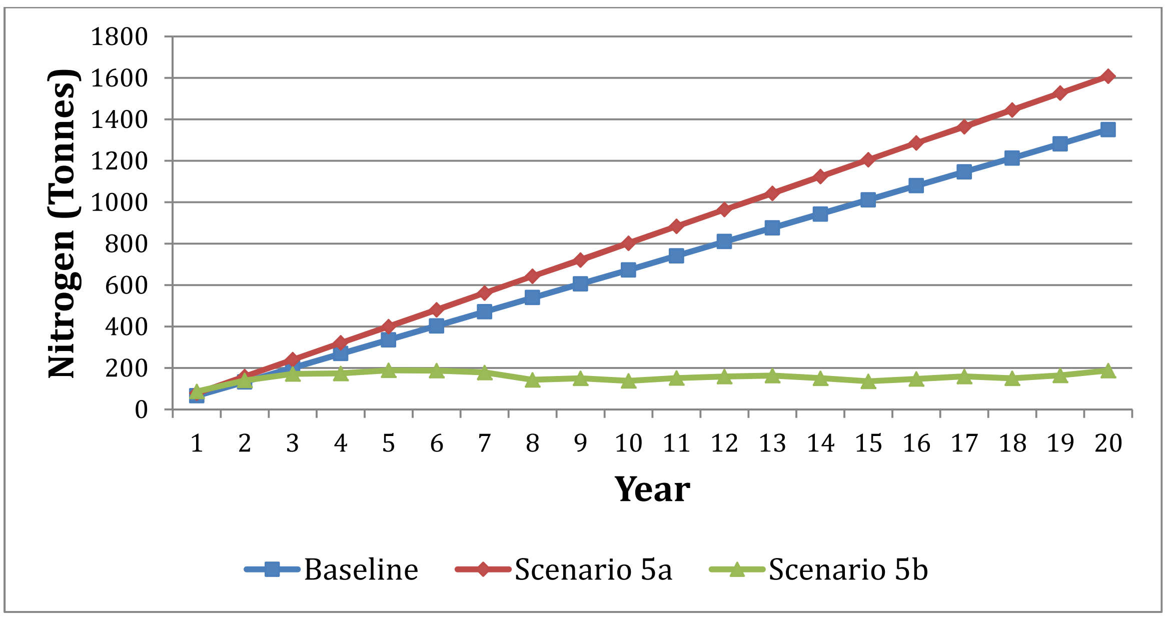


Figure 6.25. Comparison of Cumulative Nitrogen Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenarios 5a, 5b, and the Baseline Scenario (Tonnes N).



Conversion of pastureland in Scenario 5a also increased the balance of P in the watershed. Overall, P surplus increased by 20%, leading to a net gain of 55 tonnes of P over the 20 years. In contrast, P balanced decreased by 28% with the implementation of the alfalfa rotational BMP across the watershed in Scenario 5b. Of note regarding P balance is that, in Scenario 5b, the average annual balance on irrigated fields is lower than that of dryland fields. The modeled increases in yield of annual crops following alfalfa increase the total export of P (and N) from the watershed, which drives reductions in P balance on irrigated fields. However, this effect is not enough to offset the increased P inputs to alfalfa crops on dryland fields, as the proportion of summerfallow (when no fertilizer is added) is also reduced compared to the baseline. Overall, the surplus of P on dryland fields increases slightly, from 1.13 to 1.95 kg per acre (averaged over the different soil textures). Figure 6.26 shows the P balance found on the different categories of cropped fields in Scenarios 5a and 5b, and Figure 6.27 charts the accumulation of P over the course of the modeling timeframe.

Figure 6.26. Average Annual Phosphorus Balance on Irrigated and Dryland Cropped Fields, Scenarios 5a and 5b (kg acre^{-1}).

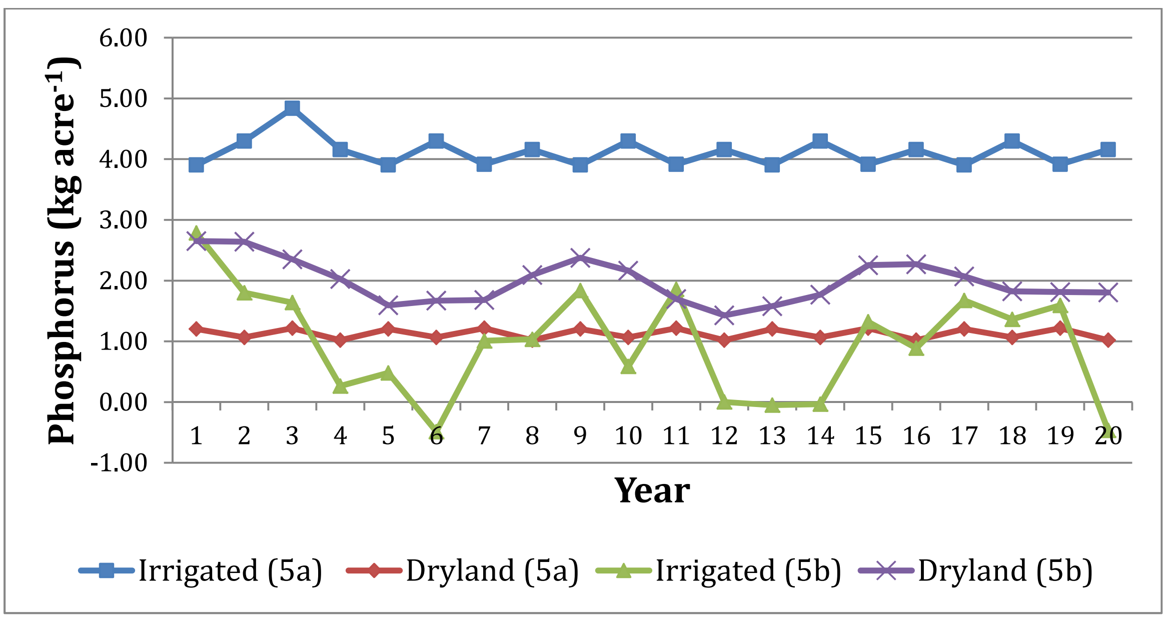


Figure 6.27. Comparison of Cumulative Phosphorus Balance in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenarios 5a, 5b, and the Baseline Scenario (Tonnes P).

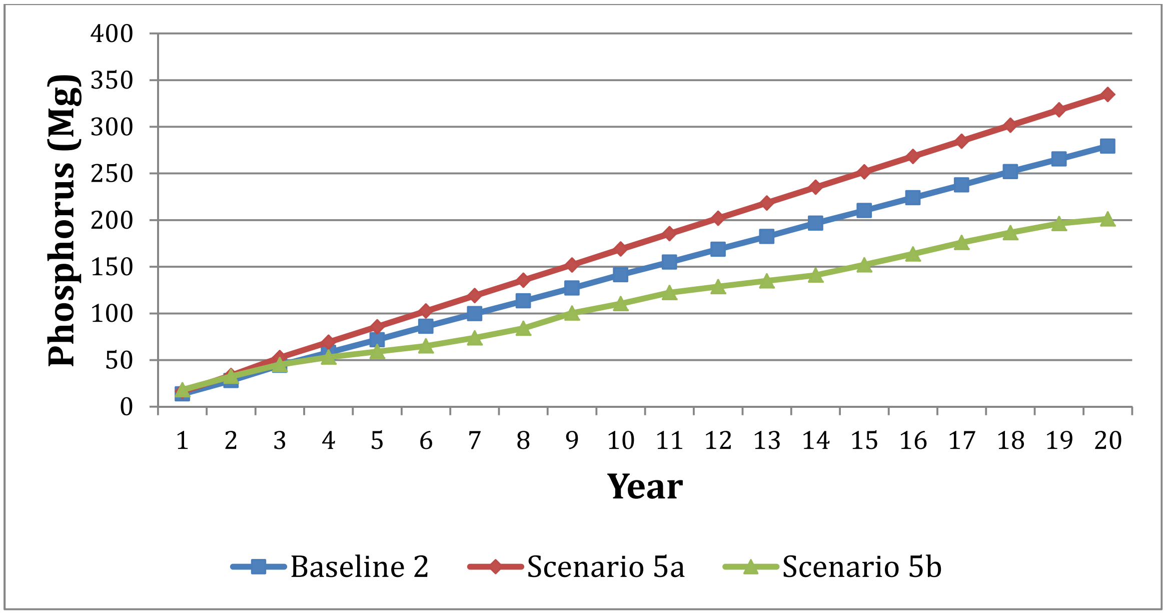


Table 6.18. Field Level Average Annual Nitrogen and Phosphorus Balance on Cropped Fields in Scenarios 5a and 5b, By Crop Rotation and Soil Texture (kg acre^{-1}).

		Scenario 5a		**Scenario 5b**	
Rotation	**Texture**	N Balance	P Balance	N Balance	P Balance
Dryland	Coarse	2.35	0.85	-0.13	0.90
	Medium	4.05	1.59	0.71	3.70
Irrigated 1	-	17.10	2.08	3.62	-0.13
Irrigated 2	-	22.24	5.74	4.63	1.82
Total Watershed		12.04	2.50	1.40	1.51

As shown in Table 6.17, N and P balance both increased in scenario 5c compared to the BAU Baseline Scenario as well. Although the percentage increases were lower (10.3% and 10.03% for N and P, respectively) than the analogous comparison between Scenario 5a and the Baseline Scenario, the absolute level of the increase in N and P was the exact same. The increase in N accumulation was 258 tonnes and the increase in P was 55 tonnes. Overall, the land use and practices modeled in Scenario 5c result in the highest cumulative surplus of N and P over the 20 year timeframe: 2,756 tonnes of N and 578 tonnes of P. If crop producers were to use manure on newly cultivated fields, the corresponding nutrient balance would increase even further.

Among the alternative land use scenarios modeled in this analysis, a decrease in SOC storage is exclusive to Scenario 5a.[27] The cultivation of fields formerly set aside as native range, or managed as tame grass, for pasture activities has the effect of reducing the SOC storage capabilities of the soil (Smith et al., 2001). The impacts of an increase in summerfallow practice are accounted for as well. The net decrease in SOC for the whole watershed in Scenario 5a is 12,690 tonnes. This decrease is mitigated in Scenario 5b by the implementation of the alfalfa BMP, which introduces perennial vegetation into annual crop rotations. The BMP is used on both the pasture fields that were recently cultivated, as well as fields initially devoted to annual cropping. Therefore, the decrease in SOC storage on former pasture fields is lessened and the cumulative total is boosted by the increase in perennials on fields formerly used exclusively for annual crops in the Baseline. Both the increase (on former pasture) and decrease (on initial annually cropped fields) of summerfallow practice are accounted for as well. The net effect is positive, as 698 tonnes of C are gained overall in soils of the LLB watershed. Table 6.19 presents the net change in SOC due to each land use change in both versions of Scenario 5.

Table 6.19. Net Changes in Soil Organic Carbon From Various Land Use Changes in Scenarios 5a and 5b (Tonnes C).

Land Use Change	**Scenario 5a**	**Scenario 5b**
Reduction of Summerfallow	-	389
Increase in Summerfallow	-1,566	-895
Conversion to Annual Crop (Medium Texture)	-5,032	-2,875
Conversion to Annual Crop (Coarse Texture)	-6,092	-3,481
Increase in Perennial Crop (Medium Texture)	-	4,313
Increase in Perennial Crop (Coarse Texture)	-	3,248
Total Watershed	-12,690	698

[27] The same SOC calculation apples to Scenario 5c as 5a.

Figure 6.28. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 5a (Tonnes C).

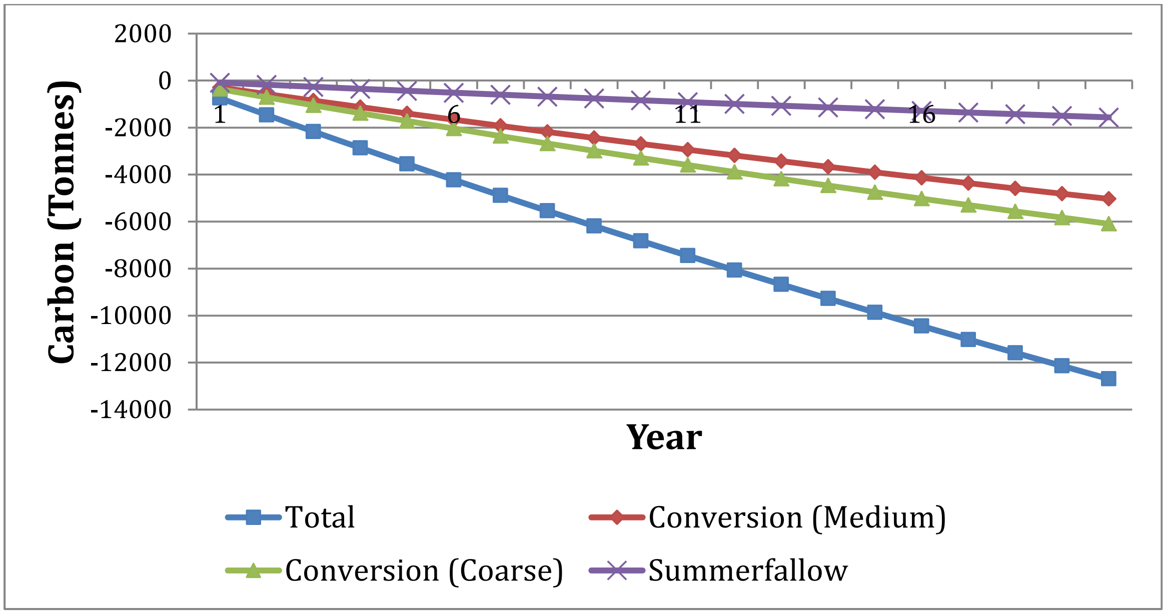


Figure 6.29. Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenario 5b (Tonnes C).

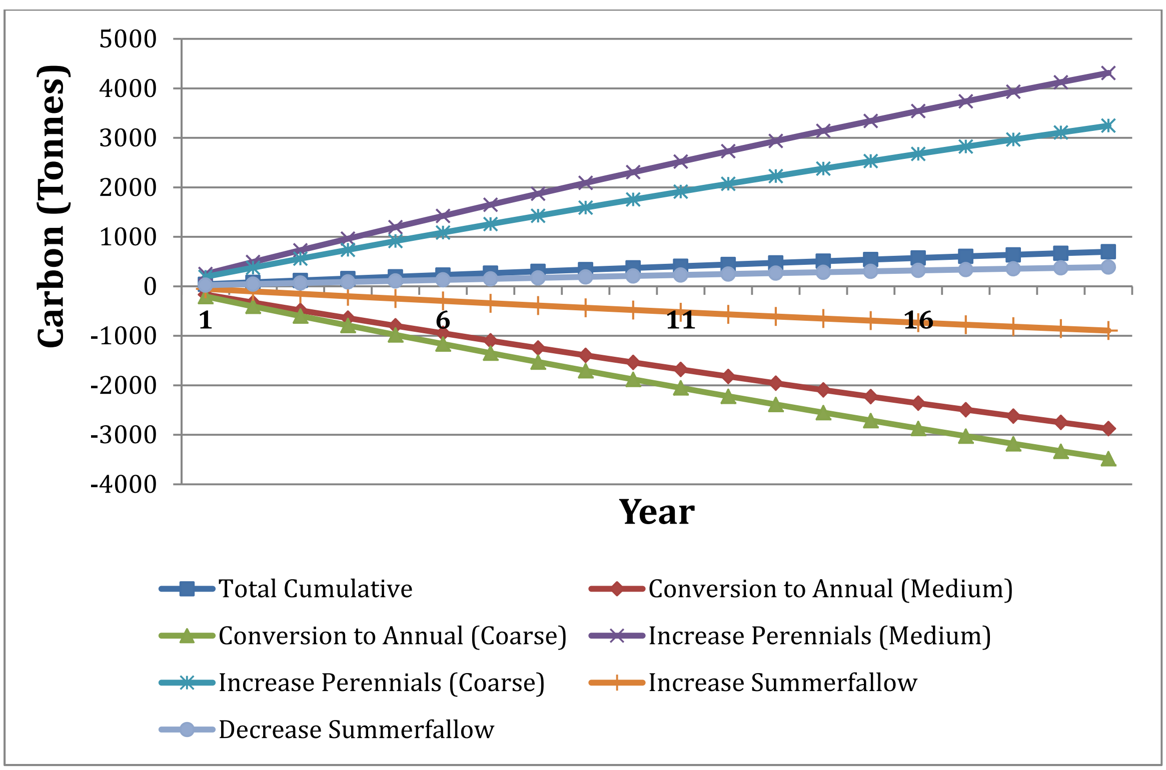
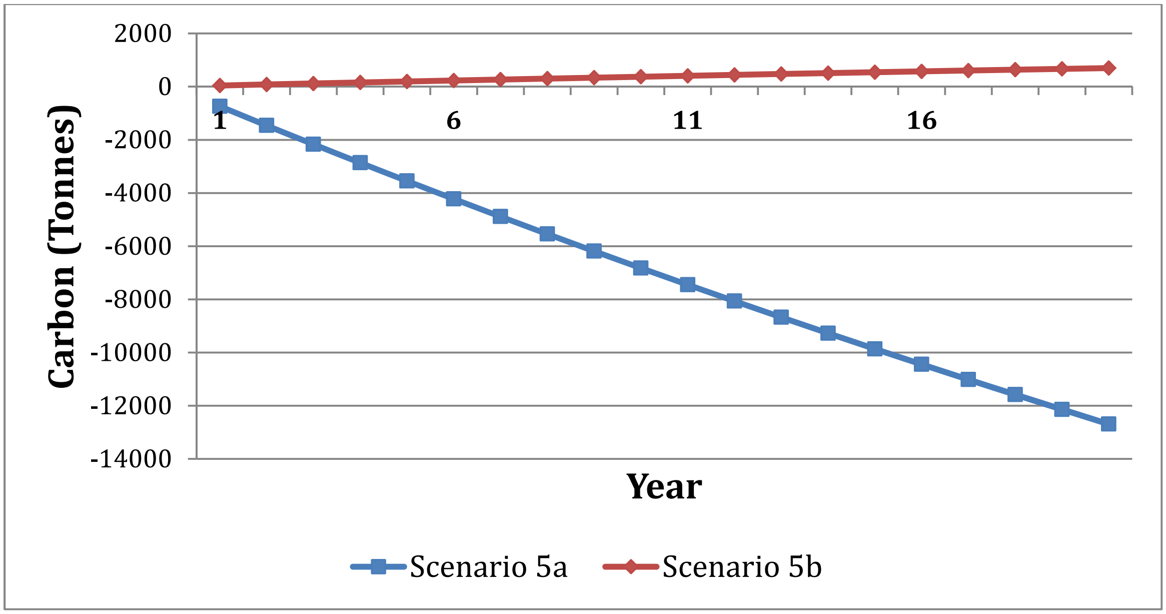


Figure 6.30. Comparison of Cumulative Net Change in Soil Organic Carbon Storage in the Lower Little Bow Watershed Over the Modeling Timeframe, Scenarios 5a and 5b (Tonnes C).



6.3 Summary Comparisons

The following sections provide a brief overview of the economic and biophysical outcomes produced by the baseline and BMP scenarios of land use in the LLB watershed.

6.3.1 Economic Outcomes

A wide discrepancy exists with respect to economic outcomes produced by the land use scenarios considered in this analysis. Shown in Figure 6.31 and Table 6.20, the mean NPV of total agricultural activity over the 20-year timeframe in the LLB watershed ranges from a low of $4,358,589 (Scenario 4a) to a high of $19,870,447 (Scenario 5c).

The difference in choice of crop nutrient source between the two baseline scenarios led a slight difference in mean NPV. The BAU Baseline, where manure is utilized for a majority of cropped fields, produced an annualized NPV per acre of $263, a $5 per acre increase over that found for the Baseline Scenario. This finding can be attributed to the difference in cost between crop nutrient sources.

Using the Baseline Scenario for comparison, four alternative land use scenarios produced an increase to producer returns generated in the watershed: Scenario 1b (7.47% greater), Scenario 3 (7.80%), Scenario 5a (33.59%), and Scenario 5b (11.35%). The increases in watershed level wealth in Scenario 1b were primarily driven by the yield increases to annual crops following alfalfa stands on dryland fields, reductions in fertilizer N costs, and a decline in the proportion of summerfallow practice in the crop rotation (which produces no revenue). These benefits offset the opportunity cost of devoting acreage to the lower-value alfalfa/grass feedstock crop. In Scenario 3, manure application produces yield benefits to crops grown, which increases revenue. Additionally, at current N and P fertilizer prices, manure is a less expensive nutrient source. Lastly, in both versions of Scenario 5 the conversion of low-value pastureland to more intensive cropping activities increases the returns to producers.

The adoption of BMP practices specified in the other scenarios involve an economic cost to producers in the LLB watershed. This includes the use of legume green manures and field peas in crop rotations, alfalfa on irrigated fields, and conversion of cropped land to permanent forage. Many of these results are corroborated well with the findings of previous work on the costs of BMP adoption (e.g., Trautman, 2012; Xie, 2014). The remaining land use scenarios (1a, 2a, 2b, 2c, 4b, and 4c) result in NPV decreases ranging from 2.50% to 15.05% when compared to the Baseline Scenario. However, Scenario 4a represents a more extreme change in land use, and consequently results in a 70.30% decrease in NPV, the largest of any watershed land use scenario.

Figure 6.31. Comparison of Producer Returns Across the BMP Scenarios and the Baseline Scenario (million $).

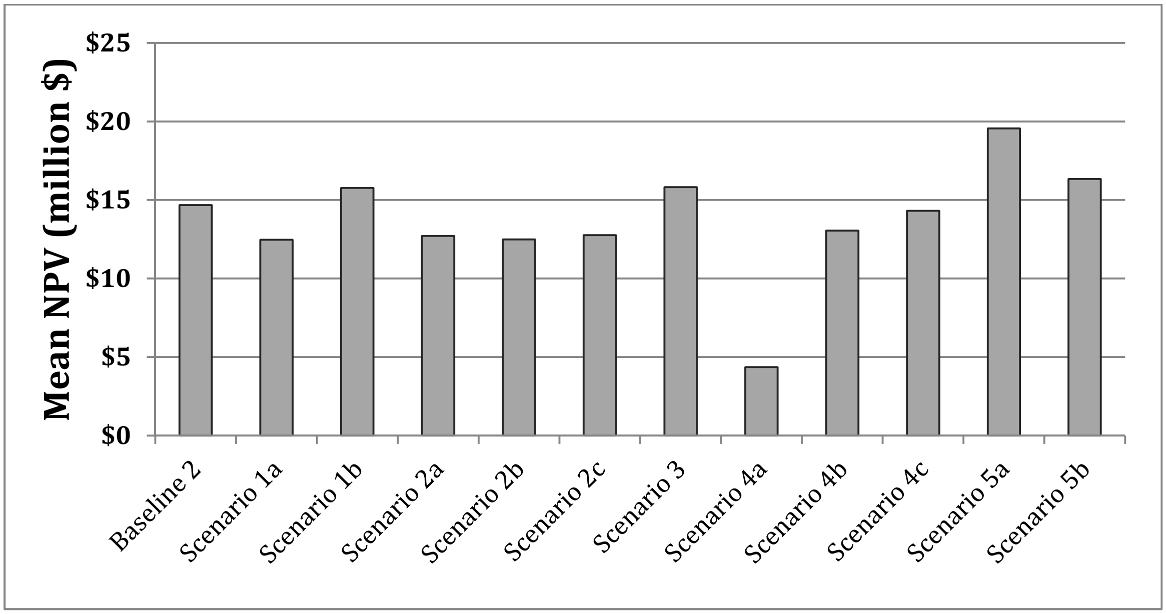


Table 6.20. Percent Change in Producer Returns, BMP Scenarios Relative to the Baseline Scenario.

Scenario	Mean NPV	Annualized NPV *($ acre^{-1})*	Percent Change
Baseline	**$14,675,018**	**$258**	-
1a	$12,466,043	$219	-15.05%
1b	$15,771,111	$277	7.47%
2a	$12,706,640	$223	-13.41%
2b	$12,486,672	$220	-14.91%
2c	$12,760,783	$224	-13.04%
3	$15,819,066	$278	7.80%
4a	$4,358,590	$77	-70.30%
4b	$13,046,283	$229	-11.10%
4c	$14,307,976	$252	-2.50%
5a	$19,562,221	$344	33.59%
5b	$16,340,385	$287	11.35%

The mean NPV of Scenario 5c is compared to that of the BAU Baseline Scenario due to the identical manure application practice. As explained in previous sections, the non-nutrient yield benefit to crops following manure application is not accounted for in either scenario. This limitation precludes the comparison of either scenario to the Baseline or other BMP scenarios. However, when manure application levels are kept constant, the cultivation of pastureland in the watershed produces a 33.09% increase to total NPV. This value is similar and analogous to the comparison between the Baseline and Scenario 5a. Because manure is less expensive nutrient source, however, the gain is slightly lower (0.5%) when only chemical fertilizer is used on newly cultivated cropping fields.

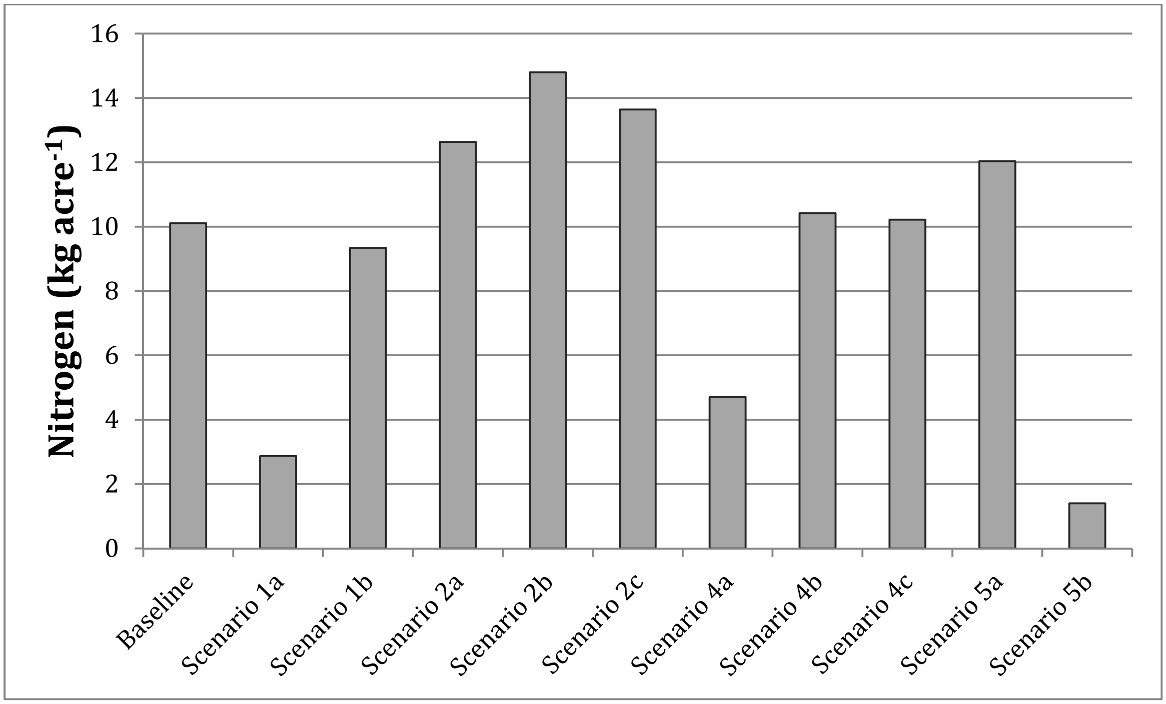
6.3.2 Nutrient Balance Outcomes

The following sub-sections provide a summary comparison of watershed-wide N and P balance outcomes from the each of the land use scenarios. The BAU Baseline, which includes representative manure application, is used to evaluate the nutrient balance outcomes in Scenarios 3 and 5c. The Baseline Scenario is used in the evaluation of the remaining BMP scenarios, all of which do not receive manure. However, the tradeoff between outcomes of N and P balance in Scenario 3 is illustrated via comparison to both baseline scenarios.

6.3.2.1 Nitrogen

Each scenario results in an overall N surplus for the LLB watershed, ranging from a cumulative total of 629 tonnes to 2,756 tonnes of excess N over the course of the modeling timeframe. For scenarios evaluated with the Baseline Scenario as a reference point, differences in average annual N balance are shown in Figure 6.32 and listed in Table 6.21.

Figure 6.32. Comparison of Average Annual Nitrogen Balance Across the BMP Scenarios and the Baseline Scenario (kg $acre^{-1}$).



The addition of a perennial legume like alfalfa, especially to irrigated crop rotations, lowers the surplus of residual N found in agricultural soils the LLB watershed. Scenarios 1a, 1b, and 4a all result in a decrease in the watershed balance of N. Several significant factors contribute to this outcome. First, through the biological process of N_2 fixation, alfalfa responds to the nutrient status of soil and can be used to regulate the import of N. Second, N credits to annual crops after terminating an alfalfa stand can lead to reduced chemical fertilizer inputs in following years, a benefit to producers. Lastly, yield increases to crops grown after the alfalfa stand result in an increase in N exports.

Conversely, the use of legume green manures in crop rotations can add a significant amount of N to soil system since the plant material is plowed down and added directly. As such, increases to watershed N levels were seen in each of the versions of Scenario 2. Reduced fertilizer applications were not enough to offset the increase in N import from the green manure crops.

Table 6.21. Percent Change in Nitrogen Balance, BMP Scenarios Relative to the Baseline or BAU Baseline Scenarios.

Scenario	Cumulative Balance [a] *(tonnes N)*	Annual Balance *(kg N acre^{-1})*	Percent Change
Baseline	**1,350**	**10.11**	-
1a	384	2.87	-71.61%
1b	1,247	9.34	-7.60%
2a	1,687	12.63	24.95%
2b	1,977	14.80	46.41%
2c	1,822	13.64	34.94%
4a	629	4.71	-53.41%
4b	1,392	10.42	3.07%
4c	1,365	10.22	1.08%
5a	1,608	12.04	19.08%
5b	187	1.40	-86.15%
BAU Baseline	**2,498**	**18.70**	-
3	2,904	21.73	16.20%
5c	2,756	20.63	10.31%

[a] Cumulative over the 20-year time period.

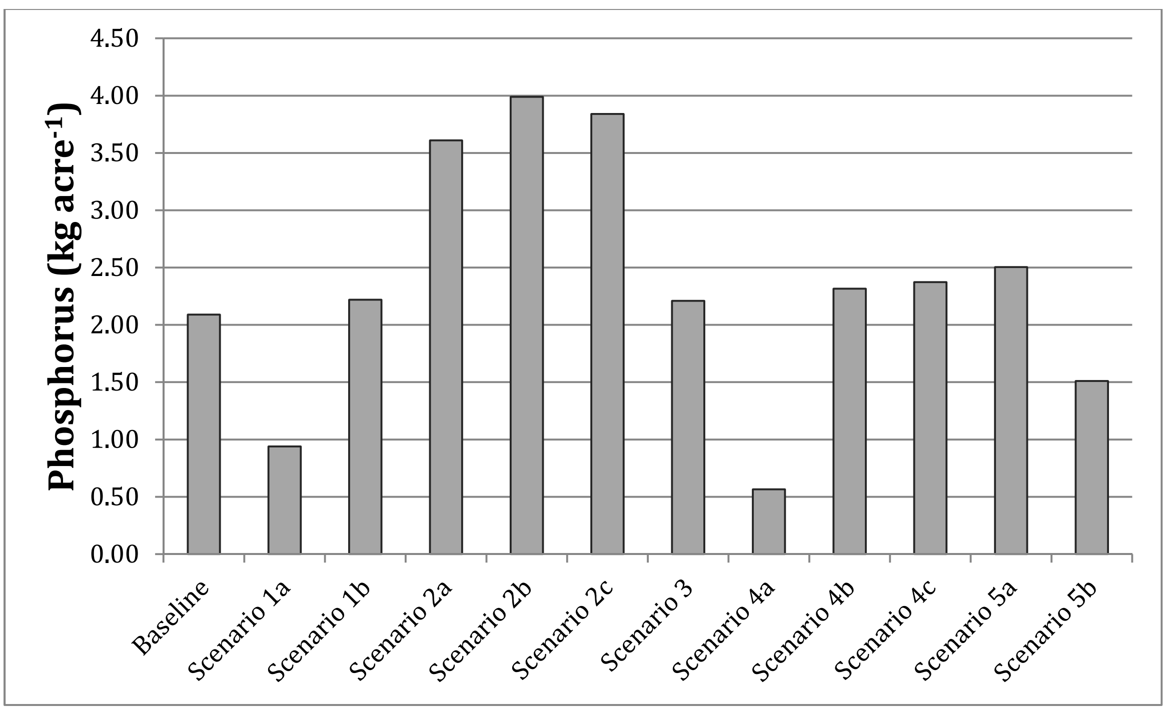
Scenarios involving manure application (BAU Baseline, Scenario 3, and Scenario 5c) were compared separately and have the highest average surplus of N. Because the organic N in manure fertilizer sources mineralize at a slow rate, a large amount of total N must be added to a field to cover the N requirements of the crop in the first year. When large quantities of manure are regularly applied to an area, soil N levels will begin to rise as various pools of N (organic, mineral) build up (e.g., Rodvang et al., 1998; Olson et al., 2010). The manure management BMP did not reduce the overall watershed N balance, although decreases of residual N on dryland fields were found. Olson (2015) suggests that consistent monitoring and soil testing is necessary to make sure residual N levels do not build up in the soil if manure is continually applied.

6.3.2.2 Phosphorus

Total watershed imports of P exceed total exports of P in each of the land use scenarios. However, certain alternative land uses were able to reduce the net amount of residual P in the watershed. Scenarios 1a, 4a, and 5b decreased the watershed P balance through the inclusion of or conversion to perennial crops such as alfalfa. Other scenarios, such as the entire suite of Scenario 2, increased the watershed P balance. In Scenario 2, this result was driven mainly because of the use of green manure crops, which are worked back into the soil instead of harvested. Because of this, there is no export of P from those fields in the year that the green manure crop is grown. Unlike N, this analysis did account for the possibility that organic P factions may mineralize and become plant-available in later years.[28] The possibility of this effect would enable a crop producer to reduce to P fertilizer inputs in following years, thus reducing the import of P and calculated P balance of the soil system.

Figure 6.33 compares the average annual P balance found across the different BMP scenarios (excluding Scenario 5c). Scenario 3 is included to exhibit the efficacy of the manure management BMP in controlling residual P levels.

Figure 6.33. Comparison of Average Annual Phosphorus Balance Across the BMP Scenarios and the Baseline Scenario (kg acre^{-1}).



[28] Some research shows that legume green manures, such as fababean, may impact the P uptake of subsequent crops (OACC, 2007).

The manure management BMP used in Scenario 3 was successful in controlling the surplus of P found in the watershed. Relative to the BAU Scenario, implementing the BMP across cropped fields of the watershed reduced residual P levels by 44%. The more efficient utilization of manure P for crop use improved this biophysical outcome. For producers considering the use of manure to replace chemical fertilizer as a nutrient source, the results of Scenario 3 relative to the Baseline Scenario demonstrate that it is possible to do so without the accumulation of excessive levels of residual P in the soil.

Table 6.22. Percent Change in Phosphorus Balance, BMP Scenarios Relative to Baseline or BAU Baseline Scenarios.

Scenario	**Cumulative Balance** [a] *(tonnes P)*	**Annual Balance** *(kg acre^{-1})*	**Percent Change**
Baseline	**279**	**2.09**	
Scenario 1a	125	0.94	-55.02%
Scenario 1b	297	2.22	6.22%
Scenario 2a	356	3.61	72.73%
Scenario 2b	409	3.99	90.91%
Scenario 2c	388	3.84	83.73%
Scenario 4a	76	0.57	-72.92%
Scenario 4b	309	2.32	10.80%
Scenario 4c	291	2.18	13.53%
Scenario 5a	335	2.50	19.81%
Scenario 5b	201	1.51	-27.75%
BAU Baseline	**525**	**3.93**	
Scenario 3	295	2.21	-43.76%
Scenario 5c	578	4.54	15.52%

[a] Cumulative over the 20-year time period.

6.3.3 Soil Organic Carbon Storage Outcomes

Impacts to SOC storage was the final biophysical metric accounted for in this analysis. In addition to improving soil quality and fertility, increased storage of C in soil is a public benefit due to the increased sequestration of CO_2 from the atmosphere. The greatest impact to net SOC storage came from the conversion of land use from annual cropping and permanent forages. The highest net gain in SOC was found in Scenario 4a, where all fields previously used for annual cropping (4,040 acres) were converted to perennial vegetation. Similarly, the highest net loss in SOC took place in Scenario 5a when all fields previously used for pasture activities (2,640 acres) were cultivated for annual crops. The alternative land uses in the remaining BMP scenarios each resulted in small to modest gains in SOC in the watershed, primarily from the changes in frequency of summerfallow practice and increases in proportion of perennial vegetation in crop rotations.

Figure 6.34. Comparison of Cumulative Net Change in Soil Organic Carbon Storage Across the BMP Scenarios (Tonnes C).

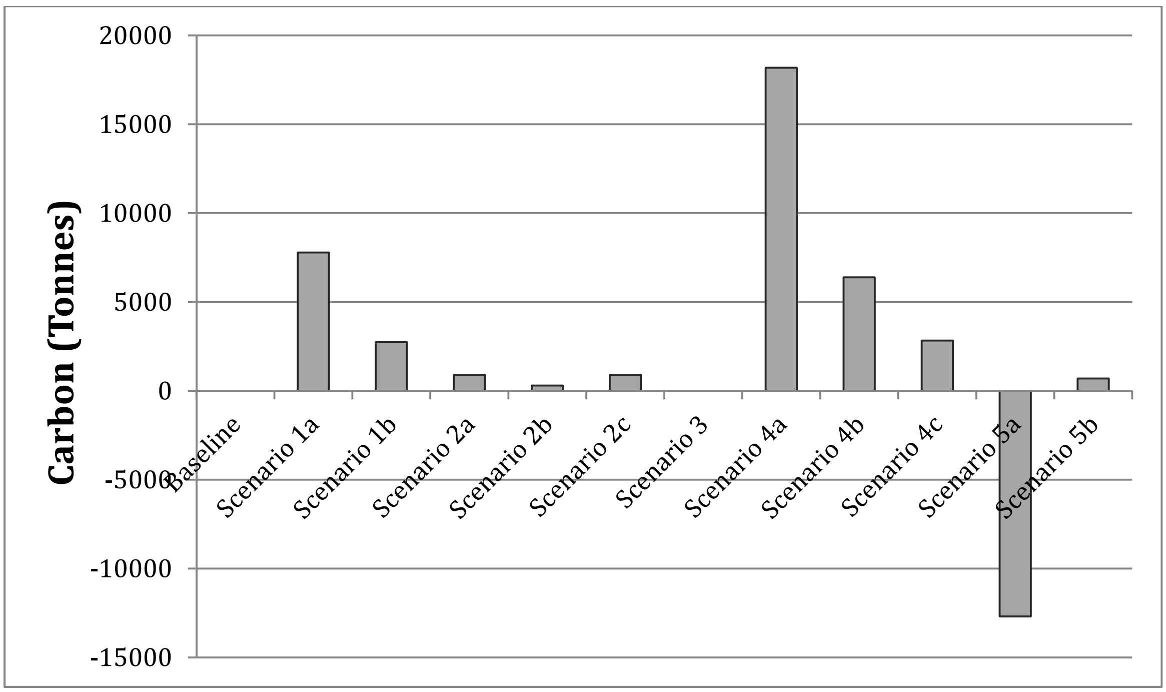


Table 6.23. Net Change in Soil Organic Carbon Storage in BMP Scenarios, Relative to the Baseline (Tonnes C).

Scenario	**Net Change** *(tonnes C)*
1a	7,791
1b	2,738
2a	907
2b	302
2c	907
4a	18,179
4b	6,389
4c	2,834
5a	-12,690
5b	698

6.4 Tradeoff Curves

Part of the value in concurrently determining the impacts to both producer returns (private benefits) and various biophysical metrics (public benefits) for each scenario lies in the capacity to construct tradeoff curves. The effectiveness of a policy-maker (i.e., regulator) depends on their ability to balance the health of the environment and the agricultural economy most efficiently. According to Weersink et al (2002), plotting economic indicators (in monetary terms) against environmental indicators (in physical terms) for alternative production systems can be a viable method for presenting information. Displaying economic and biophysical tradeoffs in such a manner can be an alternative to a conventional benefit-cost framework when quantifying agricultural impact on the environment. This method is often made necessary by the lack of available information regarding the monetary value of environmental improvements (or degradation). Although the optimal level of each indicator is not selected as it would be in a more complete economic framework (i.e., benefit-cost analysis), information gleaned from tradeoff curves may still be useful to policy-makers when choosing a desired set of agricultural management practices for both production and environmental outcomes (Weersink et al., 2002).

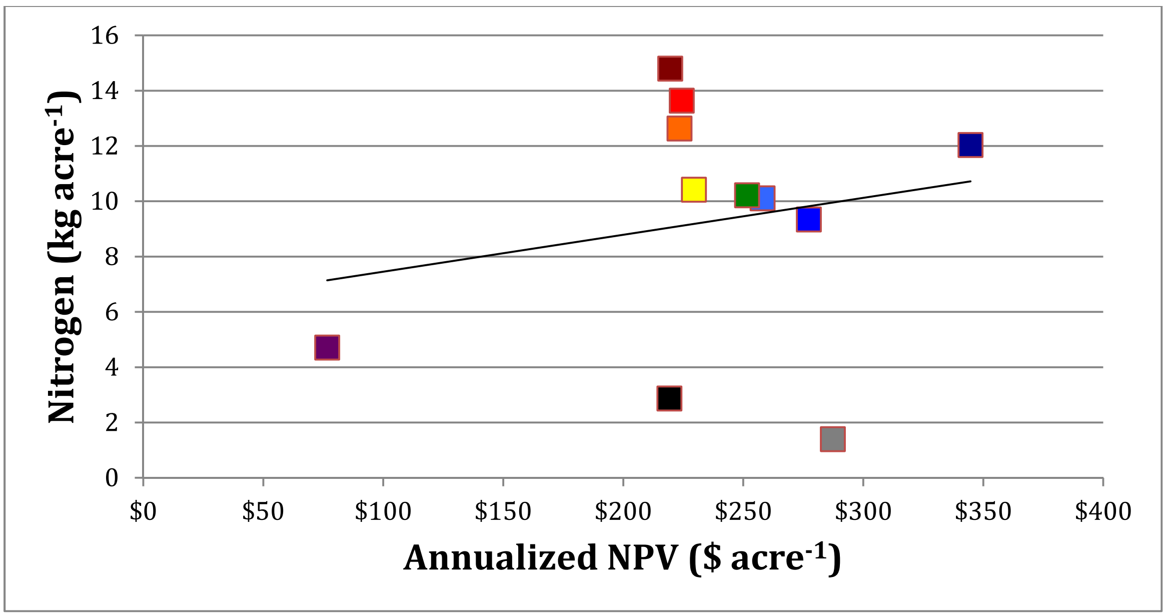
In each case, the following tradeoff curves were constructed with producer returns (presented as annualized NPV per acre) plotted against one of the biophysical metrics. At a watershed level, average N balance (kg per acre), average P balance (kg per acre), and net change in SOC storage (Mg) were evaluated. Additionally, the balance of N and P were assessed against producer returns on cropping fields only. Because most of the BMPs modeled were targeted towards cropping activities, and thus the range of nutrient status on cropped fields is highest, these activity-specific tradeoff curves help isolate and present information pertinent to crop production policy. For the tradeoffs at a watershed level, a linear function was fitted to the plotted points and an R^2 value was calculated. For tradeoffs on cropping fields, an exponential (double-logged) function was fitted instead due to better fit.

Figure 6.35 displays the tradeoff between the watershed residual N balance and the annualized NPV per acre found in 12 of the LLB watershed land use scenarios. Each point in the figure represents one of the scenarios, using the watershed N balance and NPV results discussed in the sections above. The BAU Baseline and Scenario 5c were excluded from the tradeoff curve, as the Baseline Scenario was used as the common point of reference. A visual inspection confirms that, for the watershed as a whole, residual N balance is only distantly related to producer returns in the LLB watershed. Significant reductions in N balance can be achieved through the selection of appropriate BMPs, many of which do not require the sacrifice of a significant share of producer returns. For example, if a policy-maker were to identify residual N as an environmental concern for a particular field, a suitable BMP would be the inclusion of alfalfa in annual crop rotations instead of the use of a legume green manure crop. Because the former option preserves, and even enhances, producer returns, the selection of policy may include extension and education to encourage producers to adopt the appropriate set of practices, rather than the introduction of monetary incentives (Pannell, 2008). If producers in the watershed were to intensify their operations in a manner modeled in Scenario 5a (conversion of all pastureland to cropping), the highest returns on a per acre basis could be achieved. The selection of the alfalfa BMP on all cropped fields would not only mitigate potential concerns of elevated residual N levels, but would maintain over 80% of the private returns generated (Scenario 5b).

Table 6.24. Colour of Scenario Points in Figures of Tradeoff Curves.

	Marker Colour	BMP Evaluated
Baseline	Teal	n/a
Scenario 1a	Black	Alfalfa (all cropped fields)
Scenario 1b	Blue	Alfalfa (dryland fields)
Scenario 2a	Orange	Legume green manure, field peas
Scenario 2b	Maroon	Legume green manure
Scenario 2c	Red	Legume green manure, field peas
Scenario 4a	Purple	Conversion to permanent forage (all)
Scenario 4b	Yellow	Conversion to permanent forage (dryland)
Scenario 4c	Green	Conversion to permanent forage (dryland marg)
Scenario 5a	Navy	Conversion of pastureland to cropland
Scenario 5b	Gray	Conversion of pastureland to cropland w/ alfalfa

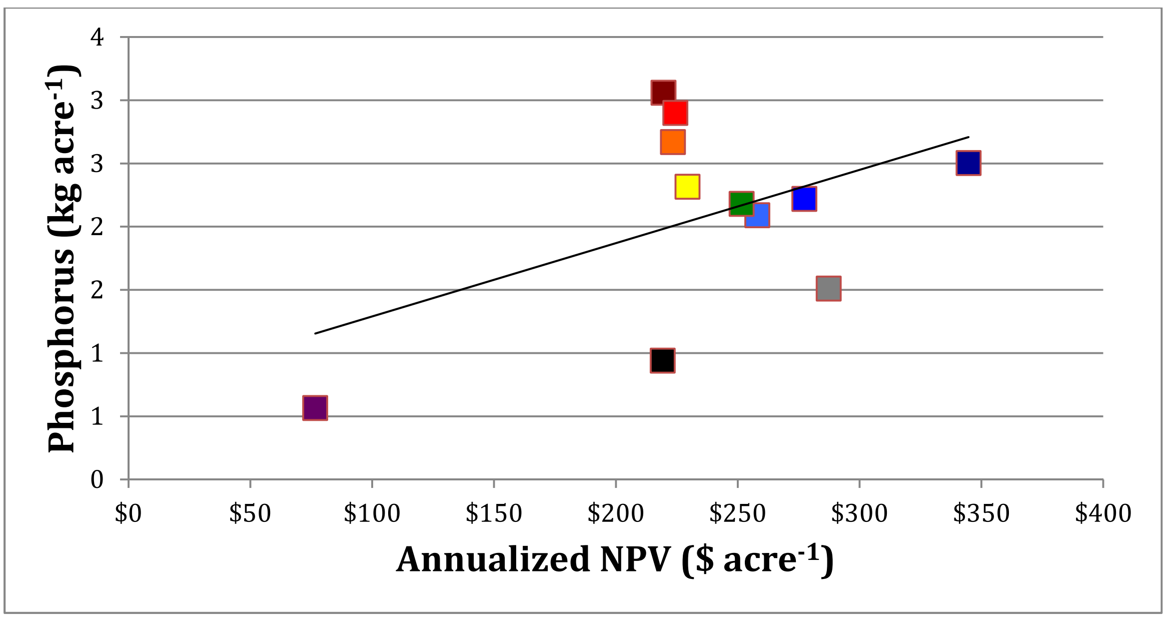
Figure 6.35. Tradeoffs Between Residual Nitrogen Balance (kg $acre^{-1}$) and Annualized NPV ($ $acre^{-1}$) in the Lower Little Bow Watershed.[a]



[a] Each point in the figure represents one of the eleven watershed scenarios, i.e. the calculated residual watershed N balance and annualized NPV are plotted for each scenario. Refer to section 6.3 for exact scenario NPVs (6.3.1) and N balance (6.3.2.1), and Table 6.24 for the marker colour key.

In terms of physical impacts to the environment, the residual P balance of the watershed is more closely related to the level producer returns. This indicates that producers may have incentives to engage in more ‘P-intensive’ practices. As Figure 6.36 shows, the scenario posting the lowest returns per acre also boasts the lowest residual P levels in soil (Scenario 4a). Similarly, the scenario which generates the highest returns also results in the fourth highest residual P levels (Scenario 5a). If the green manuring practice of Scenario 2 is taken out of consideration (as this practice is responsible for the highest three P balances found among scenarios included), Scenario 5a would then be responsible for the highest residual P balance. With this being the case, for the regulator concerned with the accumulation of P in the soil, the appropriate selection of policy may not be extension and education as in the case of N. Rather, the use of incentives, or taxation, would likely be more appropriate given the negative impact on the returns to producers when implementing BMPs designed to reduce P input. According to Pannell (2008), this policy choice will depend on the relative magnitude of each impact.

Figure 6.36. Tradeoffs Between Residual Phosphorus Balance (kg acre^{-1}) and Annualized NPV (\$ acre^{-1}) in the Lower Little Bow Watershed.[a]

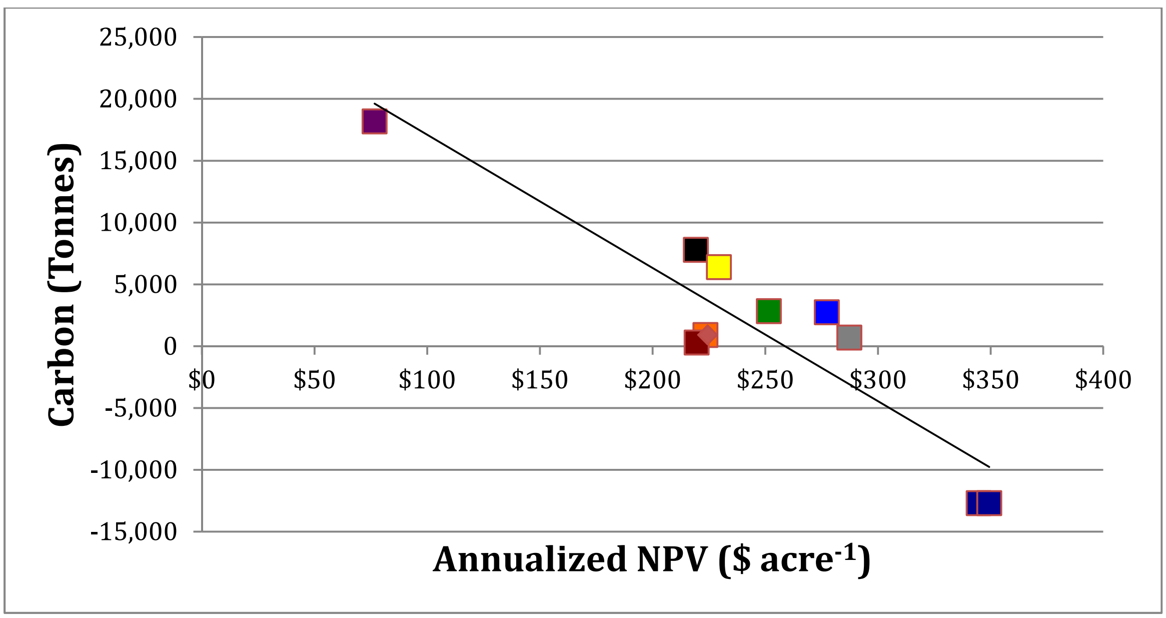


[a] Each point in the figure represents one of the eleven watershed scenarios, i.e. the calculated residual watershed P balance and annualized NPV are plotted for each scenario. Refer to section 6.3 for exact scenario NPVs (6.3.1) and P balance (6.3.2.2), and Table 6.24 for the marker colour key.

Finally, the clearest association between producer returns and an environmental metric in the LLB watershed can be found in Figure 6.37. Unlike N and P balance, SOC storage is inversely related to producer returns. That is, a higher level of SOC storage is found in scenarios with land management practices that generate lower producer returns.

For instance, the elimination of annual cropping activities in Scenario 4a produces the most gains in SOC storage (a public benefit), but the least in terms of producer wealth. Conversely, the scenario in which the highest returns are generated (Scenario 5a) comes at the expense of SOC storage, which can be thought of as a cost to society. Fitting a linear regression line to the data points reveal that, with an R^2 value of -0.84, 84% of the variation in SOC storage around its mean can be explained by the annualized NPV. To some extent, improving the carbon sink potential of agricultural soils in the LLB watershed will involve eliminating certain activities and thus the productivity capacity of producers. As such, economic incentives will likely be necessary to induce land use changes consistent with goals of SOC accumulation.

Figure 6.37. Tradeoffs Between Net Soil Organic Carbon Storage (Tonnes C) and Annualized NPV (\$ $acre^{-1}$) in the Lower Little Bow Watershed.[a]



[a] Each point in the figure represents one of the eleven watershed scenarios, i.e. the calculated net SOC change and annualized NPV are plotted for each scenario. Refer to section 6.3 for exact scenario NPVs (6.3.1) and net SOC change (6.3.3), and Table 6.24 for the marker colour key.

The same tradeoff curves were formulated using only the private and public outcomes on fields used for cropping activities, including both dryland and irrigated fields. In a majority of scenarios, these activities account for 4,040 of the 6,680 total acres in the watershed; however, the conversion of all pastureland in Scenarios 5a and 5b increase this amount to the full 6,680 acres. Net returns on cropped fields were calculated along with the N and P balances for each BMP scenario along with the Baseline. Table 6.24 lists the annualized NPV per acre of cropping activities along with the corresponding N and P balances on cropped fields for 11 BMP scenarios and the Baseline.

Table 6.25. Average Producer Returns and Nutrient Balance on Cropped Fields in 11 BMP Scenarios and the Baseline Scenario. [a]

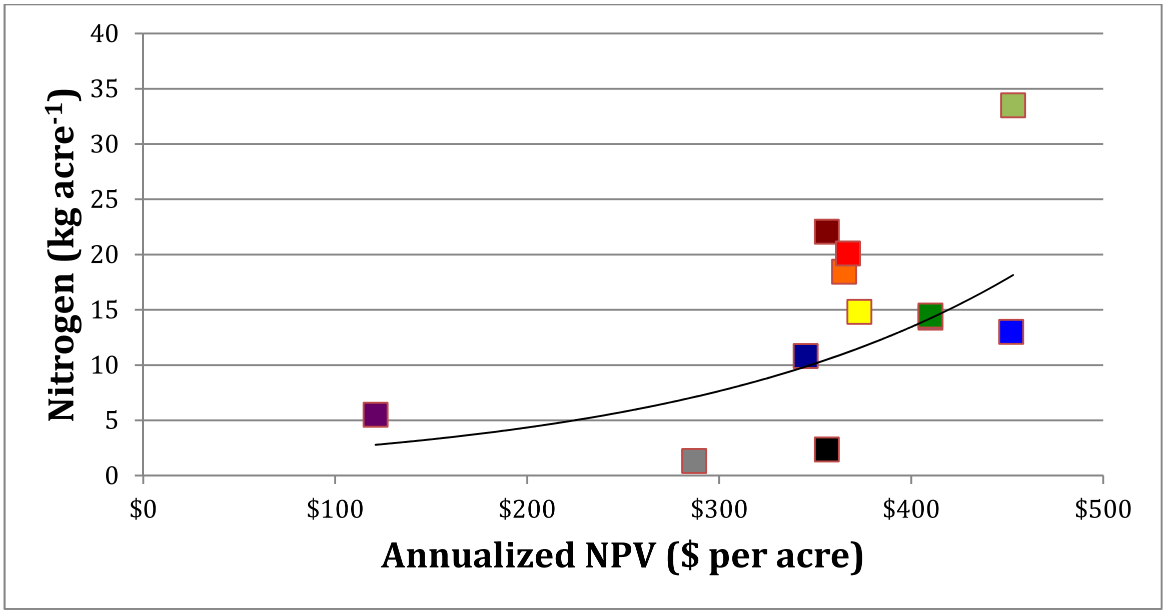
Scenario	**Cropped Acres**	**Annualized NPV** *($ acre^{-1})*	**N Balance** *(kg acre^{-1})*	**P Balance** *(kg acre^{-1})*
Baseline	4,040	$410	14.30	3.06
1a	4,040	$356	2.36	1.14
1b	4,040	$452	13.01	3.26
2a	4,040	$365	18.45	4.01
2b	4,040	$356	22.07	4.66
2c	4,040	$367	20.10	4.40
3	4,040	$453	33.50	3.27
4a	4,040	$121	5.50	0.54
4b	4,040	$373	14.82	3.44
4c	4,040	$410	14.48	3.21
5a	6,680	$345	10.82	2.50
5b	6,680	$287	1.32	1.51

[a] Each metric (annualized NPV, N balance, P balance) is averaged across both dryland and irrigated fields used for cropping activities. Fields converted to perennial crops in Scenario 4 are considered to be cropped fields.

When the returns from pastureland (which is a lower-value activity) are excluded from the calculation of producer benefits, the average per acre NPV increases. However, net N and P balance on pastureland is similar to that of dryland cropping fields (see Table 6.2) or even slightly higher. Therefore, a different relationship is found between nutrient balance and producer returns on cropped land compared to the watershed as a whole. Figures 6.38 and 6.39 show the revised tradeoff curves for cropping production with respect to residual N and P balance, respectively, with each point again representing one of the land use scenarios modeled. Instead of a linear regression form, the data points are better fitted to an exponential (double-log) functional form. In this case, the marginal gain in annualized NPV per acre decreases as residual nutrient levels increase. This demonstrates that, in crop production, the marginal damage to the environment (in the form of elevated soil nutrient levels) increases as producer returns increase. A fitted regression line is included in both Figure 6.38 and 6.39. Scenarios falling below the line can be thought of as “efficient” in terms of N or P input use and corresponding producer returns. For N balance, Scenarios 5b, 1a, and 1b fall definitively into this category, demonstrating the effectiveness of the alfalfa BMP in controlling residual N levels. For P balance, Scenarios 1a, 1b, 3, 4c, and 5b were below the fitted regression line and can be considered beneficial in terms of residual P outcomes. Conversely, scenarios that lie above the respective lines represent a set land use practices in the LLB watershed that are not cost-effective means for the goal of improving residual soil nutrient outcomes. The three versions of Scenario 2 each lie above the regression lines of both N and P tradeoff curves, indicating BMPs that may be unsuitable for environmental improvement in the LLB watershed.

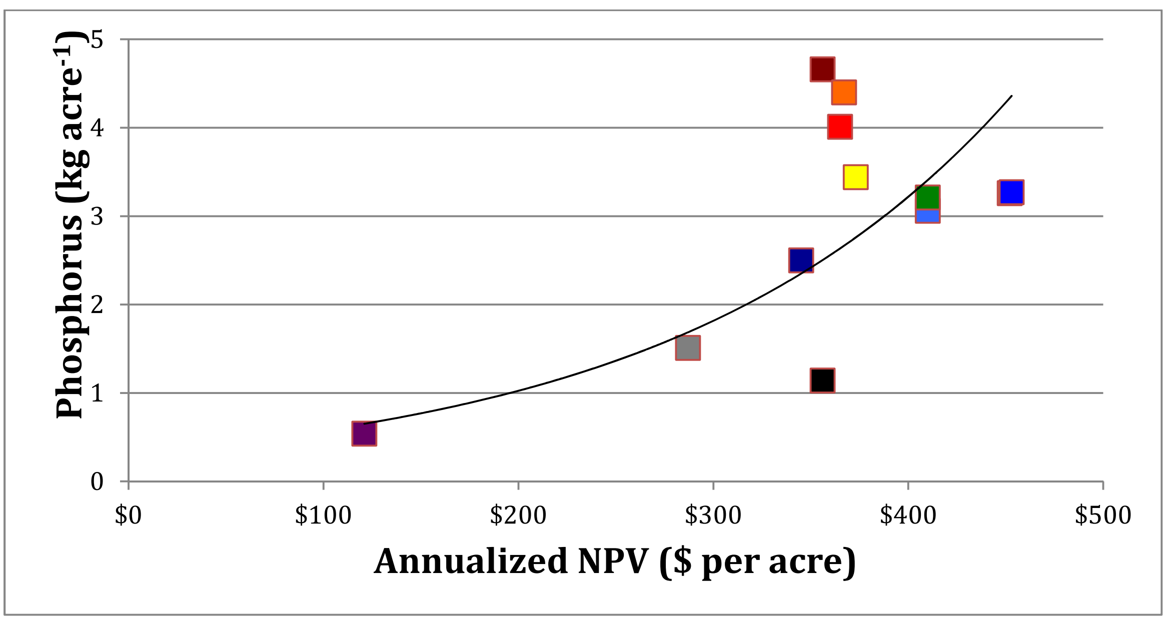
One interesting tradeoff worth discussing further is that of the manure management BMP (Scenario 3). As the tradeoff curves below reveal, this crop production BMP is on one hand efficient and potentially beneficial in terms of P balance outcome, yet also potentially damaging in terms of the high residual N balance outcome. A regulator will have to decide on the applicability and significance of this tradeoff for agricultural management in the LLB watershed. It is worth noting that other tradeoffs may also be at play in a policy decision, including the welfare of livestock producers with large amounts of manure that must be dealt with (e.g., Smith and Miller, 2008).

Figure 6.38. Tradeoffs Between Residual Nitrogen Balance (kg acre^{-1}) and Annualized NPV ($ acre^{-1}) on Cropped Fields in the Lower Little Bow Watershed.[a]



[a] Each point in the figure represents one of the eleven watershed scenarios, i.e. the calculated residual cropped fields N balance and annualized NPV generated by cropping activity are plotted for each scenario. Refer to Table 6.25 for the exact values, and Table 6.24 for the marker colour key.

Figure 6.39. Tradeoffs Between Residual Phosphorus Balance (kg acre^{-1}) and Annualized NPV ($ acre^{-1}) on Cropped Fields in the Lower Little Bow Watershed.[a]



[a] Each point in the figure represents one of the eleven watershed scenarios, i.e. the calculated residual cropped fields P balance and annualized NPV generated by cropping activity are plotted for each scenario. Refer to Table 6.25 for exact values, and Table 6.24 for the marker colour key.

6.5 Public Benefit Valuation

As alluded to in the previous section, the conversion of physical impacts of agricultural production on the environment into monetary terms permits a more direct comparison of private and public benefits. Doing so allows for the construction of a more conventional cost-benefit framework because money is used as a common frame of reference. In this section, literature estimates of the value of these impacts (either as benefits or costs) are discussed and incorporated into the calculation of mean NPV for each BMP scenario. First, the value of the climate change mitigation service provided by enhanced SOC storage is addressed. Second, the public value of improvements to water quality provided by reduced residual nutrient levels is discussed and included in the BMP scenario evaluation.

6.5.1 Climate Change Mitigation Service

The increased sequestration of atmospheric CO_2 and storage as SOC in agricultural soils can reduce the climate footprint of agricultural activities and provide climate change mitigation services to society (Burney et al., 2010). The value of this service can be determined using estimates of the social cost of carbon (SCC), which is used to convert the biophysical measure (tonnes of C) to a monetary value. The SCC is a measure of the present value of the expected future stream of marginal damages from climate change produced by an incremental (one tonne) increase in CO_2 emissions to the atmosphere (ECCC, 2016). Conversely, the SCC can also be thought of as the value of avoided damages from a decrease in CO_2 emissions. Incorporated in this measure are a variety of impacts, including increased health costs and the prospect of diminished economic output (Tol, 2009). Estimates of the SCC can vary a great deal due to a multitude of uncertainties in climate science, damage projections, and treatment of inter-generational equity issues (Johnson et al., 2012). The removal of CO_2 from the atmosphere due to increased sequestration in soils of the LLB watershed is a public benefit, and can be monetized and accounted for in this analysis using estimates of the SCC.

A range of SCC estimates have been produced in the literature. Among the first to estimate this value, Stern (2006) concluded that the urgency of global climate change required a SCC value of \$354[29] per tonne of C.[30] This high value was calculated as a result of an estimation of large future damages should action on this issue not proceed immediately, as well as the use of a low discounting rate. Compared to a higher discount rate, a low discount rate raises the value of future benefits or costs relative to the present. Nordaus (2007) suggested a more conservative SCC value, beginning at \$38 per tonne and rising 2% annually until the year 2050. More recently, ECCC (2016) also establishes a national SCC estimate to be used in cost-benefit analyses for policy decision-making in Canada. Drawing from a number of sources and models, the newly revised central EC (2016) estimate is \$149 per tonne of C in 2016. This value increases over time in five year increments, reaching \$219 per tonne in 2035. Another measure of interest is the Government of Alberta's recently released Climate Leadership Plan, which proposes to put a price on CO_2 emissions beginning in 2017 at \$20 per tonne of CO_2-equivalent (\$73 per tonne of C) and rising to \$30 by 2018 (\$110 per tonne of C) (GoA, 2015). Although not a measure of the SCC per se, this value is relevant in the sense that the public benefits associated with increased SOC storage in the LLB watershed may be priced at this level in the future (e.g., through payments for carbon offsetting). Other studies have used the commodity price of carbon found in commercial offset marketplaces such as the Chicago Climate Exchange (e.g., Kulshreshtha et al., 2015) to value this service. However, the prices found in such forums are often very low and fail to capture the full social components of climate change mitigation. As such, more robust estimates of the SCC are used in this analysis.

---

[29] Unless specified otherwise, each value cited for the SCC has been converted to 2014 Canadian Dollars to maintain internal consistency.

[30] Each value cited is also expressed as \$ per metric tonne of C, which is most convenient for this analysis. The SCC is also commonly expressed as \$ per metric tonne of CO_2 equivalent. To convert to tonne of C, a conversion factor of 3.67 is used. The atomic weight of C is 12 atomic units, and the weight of CO_2 is 44 units since each of the two oxygen (O) atoms weigh 16. Therefore, one tonne of C equals 3.67 tonnes of CO_2 (44/12 = 3.67).

For each BMP scenario, the gain in SOC (in tonnes of C) in each year of the 20-year modeling period is determined. To incorporate uncertainty in economic and scientific variables, a range of SCC estimates are used to place a value on this environmental impact. Values reported in Stern (2006), Nordhaus (2007), and ECCC (2016) are used, as well as the carbon price put forward by the Government of Alberta. Each measure of the SCC is multiplied by the net gain or loss in SOC calculated for each year, and this value is discounted at the same 10% rate used in the calculation of producer returns.[31] Table 6.25 summarizes the calculated value of the change in climate mitigation service for a range of SCC estimates. The NPV of this stream of benefits (or costs, in the case of Scenario 5a) is added to the mean NPV of producer returns calculated for each BMP scenario (see section 6.5.3).

Table 6.26. Net Present Value of Climate Change Mitigation Services Provided By Increased Soil Organic Carbon Storage of BMP Scenarios Under Four Social Cost of Carbon Estimates (thousand $).

Scenario	**Net Change (tonnes C)**	**Stern (2006)[a]**	**ECCC (2016)[b]**	**GoA (2015)[c]**	**Nordhaus (2007)[d]**
1a	7,791	$1,379	$637	$428	$167
1b	2,738	$475	$220	$148	$58
2a	302	$52	$24	$16	$6
2b	907	$157	$73	$49	$19
2c	907	$157	$73	$49	$19
4a	18,179	$3,193	$1,473	$992	$387
4b	6,389	$1,109	$512	$345	$135
4c	2,834	$491	$227	$153	$60
5a	-12,690	-$2,203	-$1,017	-$684	-$267
5b	698	$120	$56	$37	$15

[a] Using a SCC value of $354 per tonne C.
[b] Using a SCC value of $149 per tonne C in years 1-5, $166 in years 6-10, $183 in years 11-15, and $200 in years 16-20.
[c] Using a SCC value of $110 per tonne C.

[31] To maintain internal consistency within this analysis, a discount rate of 10% is used in the evaluation of future public benefits as well as private benefits. However, it should be noted that the costs of climate change impacts are more commonly evaluated using lower discount rates to account for equity issues. For instance, EC (2016) uses rates of 5%, 3%, and 2% when determining the SCC. However, because this analysis features private decisions made using a rate of 10%, that same rate is applied.

[d] Using a SCC value of $38 per tonne C in year 1, and rising by 2% in each year of the modeling timeframe.

The SCC values reported in Stern (2006) result in the highest value of climate mitigation services provided by each BMP scenario. In present value terms, the public benefit of this service in Scenario 4a, which features the highest net gain in SOC among the BMP scenarios, ranges from approximately $3,200,000 when using the Stern (2006) SCC value to $387,000 when using Nordhaus (2007). Under the Government of Alberta's new Climate Leadership Plan, this stream of benefits would have a present value of $992,000. Lower values of this benefit are found for BMP scenarios that do not result in as high a net gain in SOC. In the most extreme case, the climate mitigation services provided by the sequestration of 302 tonnes of C over the 20-year period in Scenario 2a is worth only $6,000 in present value terms using the SCC value of Nordhaus (2007). In Scenario 5a, the land use changes modeled result in a net loss of SOC, which represents a cost to society. Under the specified set of SCC values, this cost ranges from $267,000 to $2,200,000.

6.5.2 Value of Water Quality Improvements

A significant amount of uncertainty exists with respect to the benefits of water quality improvement or costs of degradation. Unlike estimates of the SCC, which combine many factors involved in the service of climate change mitigation, there is no unified measure of the value of changes to water quality. Water quality is impacted by the presence and magnitude of many factors, including nutrient levels, bacteria, sediment, and various toxins (Johnson et al., 2012). Additionally, the variety of ways in which humans use and benefit from clean water complicates the valuation process. The provision of clean drinking water is often treated as a separate service from the provision of recreational benefits, leading to the application of different valuation approaches (Johnson et al., 2012). For instance, stated preference methods have been used to ascertain the willingness-to-pay (WTP) among the public for measures to reduce nutrient concentrations in surface water for the purposes of recreation (e.g., Nelson et al., 2015). Stated preference methods, which generally involve either a contingent valuation or choice experiment survey, are used to determine an individuals WTP based on the answers they give ('state') in an experimental setting. These methods have been used in the context of drinking water improvements in the Canadian prairies (Dias and Belcher, 2015). Hedonic pricing methods are also used to assess the value of water quality. For example, Krystel et al (2003) looked at lakeshore property prices in the Mississippi headwaters region of Minnesota to determine the effect of lake water quality on property prices. Another method that has been used is the avoided cost approach, which quantifies the treatment costs that would be necessary to restore degraded water back to drinking quality. Belcher et al (2001) use this method to estimate the benefits of reduced nutrient (nitrogen and phosphorus) load in water bodies in select Canadian watersheds.

In cases where primary, site-specific valuation is not feasible (due to cost or time constraints, for example), the benefit-transfer method can be applied. Benefit transfer involves taking the values calculated at one site (the 'experiment' site) and transferring them to a different site (the 'policy' site). While this method can be cost-effective for the researcher and policy practitioner, there are potential pitfalls. Namely, the transfer of values can be an unreliable way to estimate the benefits of enhanced environmental health if the sites do not share a broad range of similar characteristics. For example, the values reported in Nelson et al (2015) apply to surface water quality improvements for recreational purposes in Utah, such as swimming and boating. These values would be a poor indicator of the public benefits of water quality improvements in LLB watershed of southern Alberta, a rural site with water bodies rarely used for recreational purposes. In this case, the preferences (i.e., values) of Utah residents for water quality improvements would likely not be transferrable to the preferences of southern Alberta residents.

In this analysis, the benefit transfer method is used to quantify the benefits of improved water quality in the watershed. The values cited in Belcher et al (2001) and Olewiler (2004) are used to value the reductions to nutrient loading of ground and surface water. These studies were chosen because they were conducted in a Canadian agricultural context and provide measures of the welfare impacts of excess nutrients in water bodies. Using an avoided treatment cost approach, Belcher et al (2001) determined that the cost of removing P from a water supply in southern Ontario range from $5 to $500 per kg. For their analysis, a median cost of $50 was used. Olewiler (2004), estimating the benefits of wetland preservation in the Fraser Valley of British Columbia, used the same approach and determined that the treatment costs in the case of excess N were between $3 and $8.50 per kg. The treatment costs either incurred or avoided from changes in N and P load can be thought of as a societal welfare change. These values are only representative of foregone expenditure, however, and do not reflect the total willingness of society to pay for water quality (Belcher et al., 2001). Therefore, it is likely that these estimates are conservative measures of the value of reducing residual N and P levels.

To estimate this value, the difference between the watershed N balance in the Baseline Scenario and each BMP scenario is calculated for each year of the modeling timeframe. The same is done with watershed P balance. To be consistent with earlier parts of the analysis, it is assumed that 10% of residual N and 5% of residual P in the soil are lost to groundwater or adjacent surface water bodies via leaching or runoff on irrigated portions of the watershed (Janzen et al., 2003). On dryland areas, this proportion is reduced to 5% and 1% for residual N and P, respectively. These percentages are applied to the calculated differences in nutrient loading between the Baseline and BMP scenarios in each year, multiplied by the per kg treatment cost estimates, then discounted over time at a 10% rate. For this analysis, per kg treatment costs of $30 and $5 for P and N, respectively, are applied.

These costs are lower than the values reported in Belcher et al (2001), although still within the range reported. These values were chosen for the following reasons. First, improved technological capacity for the abatement of water pollution over time is reasonable to expect. A more recent report (MRC, 2010) cited abatement costs of $36.85 per kg P for new wastewater treatment capacity at Lake Winnipeg, Manitoba, for the city of Winnipeg. Second, there is a low population base in the study region that would require water treatment. However, the Lower Little Bow river, which is located in the South Saskatchewan River Basin, does eventually join the Bow and Oldman Rivers to form the South Saskatchewan River. This river provides drinking water to several larger municipalities in southern Alberta, including Medicine Hat. As such, although improved nutrient transport modeling is required, it is possible that excess nutrients from activity in the LLB watershed can impact drinking water quality.[32] Between the exclusion of other values (e.g., non-use value[33]) of water quality improvements and the assumption that a non-zero proportion of nutrients lost to surface and groundwater impact drinking quality, these estimates ($30 and $5 per kg) are seen as a reasonable proxy of the marginal benefits of reducing residual nutrient levels and thus of changes to public welfare. Table 6.26 summarizes the calculated public benefits and costs associated with changes to residual nutrient levels in each BMP scenario.

Table 6.27. Net Present Value of Water Quality Benefits From Changes to Residual Nitrogen and Phosphorus Levels in BMP Scenarios. [a]

Scenario	Residual N Reductions [b]	Residual P Reductions [c]	Total Benefits of Nutrient Reduction
1a	$193,596	$111,848	$305,444
1b	$5,533	-$2,840	$2,693
2a	-$106,061	-$47,275	-$153,336
2b	-$140,284	-$54,609	-$194,893
2c	-$121,860	-$51,654	-$173,514
4a	$130,427	$160,018	$290,445
4b	-$5,350	-$3,847	-$9,197
4c	-$2,034	-$1,487	-$3,521
5a	-$47,928	-$27,182	-$75,109
5b	$182,839	$95,015	$277,854

[a] Relative to the Baseline Scenario;
[b] Calculated using an abatement cost of $5 per kg N;
[c] Calculated using an abatement cost of $30 per kg P.

A positive net public benefit in terms of water quality impacts is found for Scenarios 1a, 1b, 4a, and 5b, ranging from $2,693 to $305,444. Conversely, increases to the nutrient balance in Scenarios 2a, 2b, 2c, 4b, 4c, and 5a result in negative public benefits (i.e., costs), ranging from $3,521 to $194,893.

[32] Nitrates (NO_3^-) and nitrites (NO_2^-) from agriculture fertilizer runoff are listed as one of the parameters tested for that requires treatment in The City of Medicine Hat's Water Treatment Plant (Environmental Utilities Department, 2016).
[33] Non-use values are a component of total economic value, and refer to the value assigned to a good or service by individuals despite the fact they will never use or benefit directly from it (Tietenberg and Lewis, 2010).

It is important to note that several simplifying assumptions were made in the calculation of these values. Namely, a fixed percentage of the surplus of N and P in the soil each year is assumed to lost to adjacent water bodies. However, the timing, intensity, and frequency of precipitation and irrigation events can have a large impact on the downward movement of nutrients through the soil profile and into groundwater (Olson et al., 2009). Similarly, although levels of runoff are low in the LLB watershed (Rahbeh et al., 2011), irrigation and precipitation events do impact the movement of nutrients over land and into surface water. These factors, especially as it relates to precipitation, are inherently variable.

Another limiting assumption is that of landscape homogeneity. Relevant landscape features not accounted for include the soil texture of the field, the proximity of the field to a surface water body, or the presence of a shallow groundwater table. For instance, Olson et al (2009) report that shallow groundwater below coarse-textured soils is especially vulnerable to NO_3^-contamination. The extent to which groundwater contamination is linked to the degradation of surface water quality depends on hydrogeologic conditions, climatic conditions, and groundwater flow (Spalding and Exner, 1993). Evidence from Rodvang et al (2004) suggest that lateral groundwater transport from upslope locations in the LLB watershed does occur, increasing the likelihood of some degree of surface water contamination. As such, the assumption of a uniform landscape, where excess nutrients in each field have an equal opportunity to impact water quality, is simplistic and may potentially impact the results. A detailed analysis of hydrologic conditions in the watershed would be necessary to increase the accuracy of public benefit estimation regarding water quality improvements.

Lastly, the values reported in Belcher et al (2001) and Olewiler (2004) are taken from estimates of municipal water treatment costs in Ontario and British Columbia and as such may have imperfect applicability to the study site. Spatial or temporal variation in technological capacity or population preferences may effect these values.

6.5.3 Incorporation of Public Benefits in BMP Scenario Evaluation

This section concludes with a re-evaluation of BMP scenarios. The value of public benefits reported in sections 6.5.1 and 6.5.2 are used in conjunction with previous estimates of private benefits (i.e., producer returns) to perform a more conventional cost-benefit analysis of each BMP scenario. Table 6.27 summarizes the revised watershed mean NPV results for each BMP scenario. The values for the social cost of carbon (SCC) reported by ECCC (2016) is used in the calculation of the climate mitigation benefits provided by increased SOC storage, as it is the middle value between the two other SCC estimates used in section 6.5.1 (Stern, 2006; Nordhaus, 2007), and relatively close to the Government of Alberta's proposed carbon price (GoA, 2015).

Table 6.28. Watershed Mean Net Present Value Results of BMP Scenarios, Revised to Include Public Benefits.

Scenario	**Mean NPV** (Former)	**% Change**	**Mean NPV** (Revised)[a]	**% Change** (Revised)
Baseline	**$14,675,018**			
1a	$12,466,043	-15%	$13,408,432	-9%
1b	$15,771,110	7%	$15,993,376	9%
2a	$12,706,640	-13%	$12,577,506	-14%
2b	$12,486,672	-15%	$12,364,385	-16%
2c	$12,760,783	-13%	$12,659,874	-14%
4a	$4,358,590	-70%	$6,121,587	-58%
4b	$13,046,283	-11%	$13,549,423	-8%
4c	$14,307,976	-3%	$14,531,489	-1%
5a	$19,562,221	33%	$18,469,795	26%
5b	$16,340,385	11%	$16,673,861	14%

[a] Includes the public benefits associated with changes to SOC storage and residual N and P levels. The SCC values reported by EC (2016) are used for the valuation of the climate mitigation benefits from SOC storage.

The inclusion of public benefits in the evaluation of BMP scenarios did not change the overall direction of the impact to mean NPV initially calculated using only private benefits. Instead, the inclusion of public benefits diminished the loss in NPV for certain scenarios and augmented the overall gain in others. For instance, increases to watershed mean NPV were found in Scenarios 1b and 5b, furthering the case that these land use changes would have a positive net benefit. Alternatively, a slight decrease was found in the mean NPV of Scenario 5a, although the overall change was still positive relative to the Baseline Scenario. This finding suggests that the increases to producer returns from the conversion of pastureland outweigh the public costs of reduced SOC storage and increased watershed N and P balance. Other BMP scenarios that initially involved a decrease in mean NPV, such as Scenarios 1a, 4a, 4b, and 4c, still result in an overall decrease in welfare when public benefits are incorporated. However, the loss in welfare is tempered by the inclusion of public impacts. For example, adding alfalfa to irrigated fields, whether as part of annual crop rotations (Scenario 1a) or as permanent forage (Scenario 4a), reduces producer returns to an extent that is not made up by gains in public benefits. Lastly, implementation of the BMPs in Scenario 2, namely legume green manuring, reduces both private and public benefits according to this analysis. However, it should be noted that these BMPs have other impacts that may offset some of the public and private costs calculated here. Additionally, because they generally are in less mobile forms (organic vs. mineral), the nutrients added to the soil in the plant material of the legume green manure crop may not present as high a risk of being leached as other nutrient sources.

6.6 Sensitivity Analysis

The assumptions involved in the selection of certain parameters used in this analysis may have a significant influence on the final results and subsequent interpretations of BMP and alternative land use outcomes. As such, sensitivity analysis was performed in order to explore these impacts and identify potential areas where an assumption may prove crucial to the final result. Three parameters were identified and altered for the purposes of the sensitivity analysis. First, because improved nutrient management is the focus of many of the BMPs modeled, the price of common chemical fertilizers will affect the tradeoff faced by producers when making the BMP implementation decision. The price of fertilizer has been highly variable, both at different times of year and between years. Therefore, the sensitivity of the mean NPV calculations to fertilizer prices is investigated. The other two parameters under scrutiny in this analysis are average crop prices and crop yields. In Chapter 5, the decision was made to use the average crop prices of 2004-2013, the last ten years that data were available, as representative prices for the analysis. The impacts on mean NPV of selecting the average of the last five years (2009-2013) and twenty years (1994-2013) instead were explored. Similarly, the detrended average crop yields of the last ten years were chosen initially, and the five and twenty year average values are now used instead. Additionally, however, choice of crop yield will impact nutrient balance outcomes since the amount of N and P exported from the watershed is tied to yield. Therefore, changes to N and P balance are also investigated in the sensitivity analysis for crop yield parameters.

A thorough report and explanation of each parameter change is conveyed in Appendix E, sections E.1 – E.3. The revised fertilizer prices, crop prices, and crop yields used are reported, as well as changes in the mean annualized NPV per acre results for both baseline scenarios and each BMP scenario. Generally speaking, the findings of the analysis were not affected by changes in these parameters. Regarding fertilizer prices, none of the changes impacted the overall direction of mean NPV change related to the Baseline Scenario due to the implementation of a BMP. In some cases, the magnitude of mean NPV results did change, either enhancing or reducing the attractiveness of the tradeoff between obtaining N or P from chemical fertilizer or another source. Similarly, using variants of crop price and crop yield parameters did not change the mean NPV or nutrient balance results in a significant way. The changes were generally between +/- 5% compared to using the original price and yield values.

6.7 Chapter Summary

In this chapter, the results of the baseline and BMP scenarios were presented, including the economic returns to producers from agricultural activity, the balance of N and P at the field and watershed level, and the change in net SOC storage. Following that, each outcome (producer returns, N balance, P balance, net SOC storage) was compared across the land use scenarios, and tradeoff curves were constructed. Valuation of the public benefits provided by BMP implementation were discussed, and the benefit transfer method was utilized to incorporate public benefits into the evaluation of BMP scenarios. Lastly, sensitivity analysis was conducted on fertilizer price, crop prices, and annual crop yields.

Chapter 7: Synthesis and Conclusions

A summary of results from the baseline and BMP scenarios are provided in this chapter. Included in this summary is a discussion of the implications for future environmental outcomes, in particular water quality. Following this, the overall tradeoffs between private economic returns to producers and improvements in environmental quality in the LLB watershed are discussed. Next, this chapter also discusses the potential for valuation of the public benefits considered in this analysis, including net gains in carbon sequestration in soil and reductions to nutrient loading in ground and surface water. This chapter concludes with a discussion of the implications for agriculture production and policy in the LLB watershed, based on the findings of this study. Lastly, the limitations and assumptions made in the development of the land use scenarios are discussed, along with suggestions for areas of future research.

7.1 Summary of Results

The intensity of agricultural production in the LLB watershed necessitates the consideration of environmental impacts. This intensity is reflected in a number of ways, many of which can have adverse impacts to the environment if not managed properly. In this region, nutrient (primarily N and P) build up in soil has been identified as an area of concern, especially as it relates to water quality. The prevalence of livestock operations and continuous annual cropping production in the area has historically resulted in levels of manure and chemical fertilizer application beyond that which can be utilized for crop growth. As such, the resulting residual soil N and P surplus presents a risk to the local environment. This issue has been studied extensively in southern Alberta, and in Lethbridge County in particular (Rodvang et al., 1998; Little et al., 2003; Kohn et al., 2015).

Various Beneficial Management Practices (BMPs) have been proposed to address this issue in the LLB watershed. Included among the BMPs evaluated for the WEBs project, for instance, were streambank fencing and off-stream watering for cattle, conversion to greencover (perennial forage), manure management, and the incorporation of buffer strips. Several other alternative land uses and practices were introduced in this analysis as well, including crop rotation BMPs and land use conversions. While previous studies have quantified and evaluated the private economic impacts of BMP adoption on crop producers (e.g., Trautman, 2012), the objective of this study was to evaluate and provide a measure of public impacts at the watershed level, as well. The dual assessment of both private and public impacts allows for the evaluation of tradeoffs between the two, a valuable source of information to policymakers. Depending on the result, choice of policy might include increased extension and education, direct incentives, or taxation to achieve a more optimal balance between private and public benefits. The public benefits of focus in this analysis were water quality improvements and increased storage of soil carbon in agricultural soil.

To accomplish the study objective, a series of alternative land use scenarios for the LLB watershed were constructed to quantify the economic impacts to producers, the corresponding changes in both watershed and field-level nutrient balance, as well as net change in soil organic carbon (SOC) levels. Historical county level crop yield and crop price data obtained from AAF, AFSC, and the CWB were used in conjunction with estimates of representative crop production practices and input costs (including fertilizer use) to determine economic returns. Baseline scenarios, where BMPs are not adopted, were built using LLB watershed-specific agricultural activities and management practices to serve as points of reference for evaluation of BMP scenarios. The economic and biophysical impacts of agricultural land use specified in each scenario were tracked over a 20-year modeling period. The economic outcomes are calculated as the NPV of net returns to private producers. The difference in NPVs between the baseline and BMP scenarios is considered to be the private benefit from BMP adoption, which can be positive (wealth increasing) or negative (incurring a cost). As proxy for risk to water quality, the resultant watershed and field-level nutrient (N and P) balance is determined, along with a calculation of changes to net SOC. The difference between the baseline and the BMP scenarios with respect to these environmental outcomes is considered to be the public benefit, or cost, of BMP adoption. These metrics are often not available or reported in monetary terms, and as such the relevant tradeoffs for policy analysis must be evaluated in a different way (section 6.4). In some cases, monetary estimates of changes in environmental quality are available, and can be used in the formation of a more traditional benefit-cost analysis (section 6.5).

7.1.1 Private Impacts of Beneficial Management Practice Adoption

The feasibility of BMP adoption for producers is evaluated by quantifying the various on-farm private costs and benefits at a watershed level arising from each land use scenario. Overall, the adoption of a majority of the BMPs modeled in this analysis will incur a cost to producers of the LLB watershed. The effect on "farm-level" (treating the LLB watershed as a single economic enterprise) income is negative for each BMP scenario except Scenario 1b and Scenario 3, which increase income. The three versions of Scenario 5 also increase producer returns. However, Scenario 5 does not feature the implementation of a traditional BMP in the sense that the primary purpose of the land use changes presented is not to enhance the provision of public benefits. Rather, the intensification of land use for private gain is modeled.

In Scenario 1 alfalfa is added to annual crop rotations. Alfalfa is a common hay crop in Alberta, and when a stand is well-managed can typically yield great quantities of high quality forage. Because alfalfa is a legume, it also has the ability to fix atmospheric nitrogen (N_2) in the soil. This is an attractive characteristic to a producer because it eliminates the need for N fertilizer during the years of the alfalfa stand, as well as provides N credits in the soil to subsequent non-legume crops. Additionally, a yield benefit is often observed to subsequent crops due to improved soil conditions and a break in disease cycles. The drawback of including alfalfa, however, is that a producer must forgo higher-value crop production during the years that alfalfa is grown. Therefore, an evaluation of these private tradeoffs is provided in Scenarios 1a and 1b, as alfalfa is first added to each cropped field in the LLB watershed (4,040 acres) and then only to dryland fields (1,320 acres). In Scenario 1a, the average watershed NPV, annualized on a per acre basis, declines \$39 relative to the Baseline Scenario[34]. The benefits in terms of increased crop yield and reduced fertilizer applications were not enough to offset the proportional loss of valuable annual crop production (e.g., potato). Although still negative overall,

[34] Each BMP scenario except for Scenario 5c uses the 'Baseline' scenario as a reference case in the evaluation of private benefits. Unlike the BAU Baseline, the Baseline does not feature manure application on a proportion of cropped fields.

sensitivity analysis revealed that the magnitude of income loss is subject to fertilizer N price, indicating that the attractiveness of the alfalfa BMP increases as fertilizer N price decreases. In Scenario 1b, a positive impact on private returns was found as annualized NPV per acre increased by $19. The reversal in results between the two versions is due to the higher value nature of irrigated cropping. On dryland fields, the alfalfa BMP is feasible for a private producer to adopt.

Scenario 2 involved the addition of a legume green manure crop (fababean) to irrigated fields and both field peas and red clover (as a green manure) to dryland fields in different combinations and rotations. Green manuring, especially with legume plants, can be considered a BMP for several reasons, namely for improvements in soil quality (including soil fertility) and crop yield benefits. However, the green manure practice necessitates the elimination of crop revenue generation in that year as the crop is plowed into the soil instead of harvested. Field peas are a viable, albeit uncommon, crop in the Dark Brown soil zone of Alberta. The inclusion of field peas in more conventional crop rotations can also be considered a BMP due to the N fertilizer and crop yield benefits to crops following it. The annualized NPVs of Scenarios 2a, 2b, and 2c were $35, $38, and $34 per acre lower than that of the Baseline Scenario, respectively, indicating there is a net cost associated with adoption. Importantly, however, further analysis of land use specific returns revealed that the green manuring BMP (on both irrigated and dryland fields) was solely responsible for the decreases in NPV found. In Scenario 2a, the annualized NPV of dryland fields increased by $25 per acre as field peas replaced summerfallow and green manuring was not included. According to this study, the overall land use changes implemented in this scenario resulted in a loss of economic returns, although the evidence suggests the inclusion of field peas in dryland cropping rotations may be economically feasible.

A manure management BMP is modeled in Scenario 3. Cattle manure is applied on cropping fields once every four years based on the one-year nitrogen requirements for spring wheat. This practice can be considered a BMP because it reduces the total amount of manure applied vis-à-vis the standard legislated practice of applying at the crop N requirement annually. In particular, this practice helps prevent the buildup of P in the soil. Improved soil quality is a benefit of manure application compared to chemical fertilizer, which can increase crop yields and thus revenue for a producer. Based on this analysis, the application of cattle manure increases the annualized NPV found for the watershed by $20 an acre. This increase is in part due to the modeled crop yield benefit as well as because manure is a less expensive source of nutrient than chemical fertilizer. This result, in terms of the directional change in annualized NPV, was not affected by a sensitivity analysis of fertilizer prices, although the benefit does diminish if fertilizer N and P prices become less expensive.

Scenario 4 featured the conversion of annual cropland to perennial forage. Specifically, irrigated cropland was used solely for alfalfa and timothy feedstock production, and dryland cropland was seeded to an alfalfa/grass mix. The reduced intensity of cropping activity, including a reduction in fertilizer use, makes this land use conversion a BMP. In Scenario 4a, where annual crops are eliminated entirely, the impact to net cashflow is negative, resulting in an annualized NPV decrease of $181. In Scenario 4b annual cropping on irrigated field is continued, but dryland fields are converted to permanent forage. This land use change results in a NPV decrease of $29 per acre each year when averaged across the watershed. Lastly, when only marginal cropland is targeted, the decrease in NPV is only $6 per acre annually. Overall, the substitution of annual crops for permanent forage is a costly BMP to producers, especially when irrigated annual cropping is involved.

The final set of scenarios featured a land use change where pastureland is converted to annual cropping. In general, the grazing of livestock on land left as native range or tame grass is a less profitable activity than cropping. Historically, high crop prices have incentivized producers to invest in the cultivation of land formerly used only for pasture. Assuming this is a feasible land use change in the LLB watershed, the impact to mean watershed NPV was calculated in Scenario 5 when the conversion of pastureland is modeled. When baseline crop rotations are used on newly cultivated fields (Scenario 5a), the mean annualized NPV per acre increases $86. Because the vast majority of newly cultivated fields are dryland, this increase is less than what would have been expected if irrigation infrastructure was present in the same proportion as originally cropped fields. When alfalfa is grown in annual crop rotations (Scenario 5b), the increase in private returns is diminished, and an annualized gain of $29 per acre is found. Not surprisingly, the increase in private benefits associated with cultivating pastureland provides an incentive for producers to proceed with this land use change. Even if this were not feasible in the LLB watershed (agronomically or otherwise), results from this analysis suggest that it may be likely in areas with a similar agriculture activity profile.

7.1.2 Public Impacts of Beneficial Management Practice Adoption

With the exception of Scenario 5, each BMP scenario was designed around the premise that the implementation of the land use change or change in management practices would enhance the provision of public benefits. Public benefits were considered in two distinct categories: reductions to residual (surplus) nutrient levels and increases in the carbon sequestration capacity of agricultural soils. Regarding the former, changes to the balance of both nitrogen (N) and phosphorus (P) were considered.

The impacts to the watershed balance of N were mixed. Of the BMPs modeled, the most significant driver of N reductions on cropped land was the inclusion of alfalfa in crop rotations on irrigated fields. This BMP was modeled across the LLB watershed in Scenarios 1a and 5b, and resulted in residual N decreases of 72% and 86%, respectively, with respect to the Baseline Scenario. These results were due primarily to the reduction in chemical fertilizer application needed, as alfalfa both fixes its own N from the atmosphere and provides N credits to subsequent crops. Additionally, the yield benefit modeled increases the total N export from the watershed. Because N balance on irrigated fields is substantially higher than dryland in the Baseline Scenario, the reduction in watershed N balance is tempered in Scenario 5b when the BMP is limited to dryland fields (8% reduction). The next most effective BMP is the conversion of irrigated fields to permanent forage, which takes place in Scenario 4a. This land use change reduces residual N levels by 53%. However, when only implemented on dryland cropping fields, the benefits in terms of watershed N balance are eliminated. In Scenarios 4b and 4c, which model this particular BMP on only dryland fields, the surplus of N in the watershed increases by 3% and 1%, respectively, relative to the Baseline Scenario. N fixation levels in the alfalfa/grass crop exceeded the N chemical fertilizer additions to annual crops modeled in the Baseline Scenario.

Other BMPs were ineffective in enhancing this particular category of public benefit. The three versions of Scenario 2 increased the net balance of N in the watershed by 25-46%. Green manuring on irrigated fields, in particular, vastly increased the total import of N, which was only slightly offset by reduced chemical fertilizer application and crop yield increases. The manure management BMP, when compared with the BAU Baseline Scenario, also increased the balance of N (16%) in the watershed. However, this particular BMP was effective in reducing the N balance on dryland fields as manure application was done more closely in line with the annual crop N requirement. Scenarios 5a and 5c both increased the watershed balance of N, which was expected due to the nature of the land use changes.

The second component of public benefit with regard to nutrient balance was residual P levels in the watershed. Overall, the impacts of BMP implementation on P balance mirrored that of N balance. Scenarios 1a, 4a, and 5b were all effective in reducing the net balance of P in the watershed. This was primarily due to the yield increases modeled for annual crops following alfalfa hay, which increased total P export. Another reason for the decrease is the lower fertilizer P application required for alfalfa and timothy hay, especially compared to annual crops such as potatoes and sugar beets. In each version of Scenario 2, as well as Scenarios 4b and 4c, the balance of P increased. In Scenario 2, green manuring was primarily responsible for this result. In Scenarios 4b and 4c, the conversion to permanent forage on all (or select marginal) dryland fields did not reduce P inputs nor increase P exports vis-à-vis the Baseline Scenario.

The sole difference in the direction of result between N and P balance impacts was found in Scenario 3. Relative to the BAU Baseline Scenario, Scenario 3 decreased the net balance of P in the LLB watershed by 44%, from an average annual balance of 3.93 to 2.21 kg P per acre. This decrease is the equivalent of a 250 tonne reduction in import of P to the watershed over the course of the 20-year timeframe. Impressively, this BMP only increased P balance by 6% when compared to the Baseline Scenario, demonstrating the efficiency of manure P utilization by crops.

Net change in SOC storage in agricultural soils of the LLB watershed was also considered as a public benefit. The sequestration of atmospheric CO_2 in soil is beneficial to society as a climate change mitigation service. As such, increased SOC storage would represent a public benefit, whereas loss of SOC would represent a public cost. Three changes in agricultural land use management were identified as being consistently impactful on the net change of SOC levels: the reduction of summerfallow, inclusion of perennial crops in crop rotations, and conversion to permanent forage. With the exception of Scenario 3, at least one of those three changes were present in every BMP scenario. Overall, the most benefit was found in scenarios involving either the inclusion of perennial crops or the conversion to permanent forage. Scenario 4a, which involves the complete conversion of annually cropped fields, produces the largest increase in SOC at 18,179 tonnes over the entirety of the modeling timeframe. In comparison, including perennial crops (i.e., alfalfa) as part of annual crop rotations yields an increase of 7,791 tonnes C in Scenario 1a and 2,738 tonnes C in Scenario 1b. Scenarios 4b and 4c, which involve complete conversion but are limited to dryland fields, generate gains of 6,389 C and 2,834 tonnes C, respectively. The versions of Scenario 2 generate the smallest benefit in terms of increased SOC, ranging from 302 tonnes C in Scenario 2b to 907 tonnes C in both Scenario 2a and 2c. This limited benefit was because only changes to summerfallow practice were present as part of the scenario BMPs and included in the SOC calculations.

Scenario 5a was unique in that the land use changes modeled involved a decrease in net SOC. This was due to the cultivation of fields originally in native range or tame grass and used for permanent forage. Over the 20-year period, SOC in the watershed declined by a net 12,690 tonnes relative to the Baseline Scenario. The impacts on SOC from the land use changes modeled are considered a negative public benefit, or cost, to society.

Several methods were used to attach monetary values to the above public benefits in order to add supplemental information to the evaluation of private and public tradeoffs. First, various estimates of the social cost of carbon (SCC) were used to estimate the value of the climate change mitigation service provided by increased SOC storage in the watershed. A range of values were reported, reflecting the discrepancy and uncertainty of the SCC estimates, as well as the disparity between SOC storage outcomes among the BMP scenarios. Using the SCC values published by ECCC (2016), the present value of this public benefit ranged from a high of $1,473,000 in Scenario 4a to a low of -$1,017,000 in Scenario 5a (which represented a public cost). The value of the expected water quality improvements induced by reductions in residual nutrient levels in the LLB watershed was also estimated. The benefit transfer method was employed. Specifically, the values derived by other studies (Belcher et al., 2001; Olewiler, 2004) using an avoided cost approach were utilized in the estimation of changes to public welfare from water quality improvement. Under a broad range of assumptions, the total benefits ranged from a present value of $305,000 in Scenario 1a to a low of -$194,893 in Scenario 2b, which can be interpreted as a public cost.

The inclusion of monetary estimates of public impacts in the analysis of the BMP scenarios did not have a substantial effect on the overall conclusions. Each of the BMP scenarios that involved a private cost to producers in terms of reduced wealth still resulted in a net loss despite the inclusion of monetized public benefits (or costs). However, the net loss was, in several cases, reduced. For instance, inclusion of public benefits increased the watershed mean NPV of Scenario 4a by approximately $1,800,000, and the percent change relative to the Baseline Scenario went from -70% when only private impacts were accounted for to -58%. In other scenarios, such as Scenario 1b, the gains to producers are augmented by the inclusion public benefits, highlighting the positive impacts that the adoption of these BMPs might have.

7.2 Implications for Agricultural Production and Policy in the Lower Little Bow Watershed

The objective of this study was to increase understanding of both the private and public benefits of incorporating BMPs in agricultural operations and changing land use throughout the LLB watershed. When both private and public benefits are evaluated, the tradeoffs between the two can be assessed in order to more efficiently quantify the impact of agricultural production on the environment. In the case of agricultural activity in the LLB watershed, the implementation of the BMPs and land use changes of interest in this study will produce mixed results with respect to both private and public benefits. BMPs that are costly for producers generally involve the inclusion of non-marketable crops (e.g., those used instead for green manuring), lower-value crops (e.g., alfalfa or timothy hay on irrigated fields), or removal of land from annual crop production entirely. It is unlikely that producers will adopt these BMPs voluntarily, as they involve a private cost. When this is the case, a policy-maker or regulator can look to the estimation of biophysical impacts produced by this analysis to provide guidance as to whether the BMP is worth pursuing from a societal point of view. Achieving balance between the health of the private farm economy and the health of the environment is imperative. As such, tools such as the tradeoff curves developed in section 6.4 or the public valuation estimates provided in section 6.5 can be useful in the policy development process. If it were determined that the implementation of a certain BMP in the LLB watershed would be socially beneficial, economic incentives could be offered to producers to encourage BMP adoption (Pannell, 2008). Ideally, the incentive would be enough to cover the annualized reduction per in NPV calculated in the BMP scenario results, but still produce a net benefit to society (i.e., the incentive paid would be less than the total value of the public benefits, such as climate change mitigation or water quality services).

In certain cases, the on-farm benefits, such as a reduction in input costs or gains in output, can offset or outweigh the costs of implementation. An example of this is the inclusion of alfalfa in dryland crop rotations, which was modeled in Scenario 1b. When a BMP causes a positive private net benefit to a producer, the calculated value of the annualized increase per acre in NPV of economic returns can be used instead for education and extension programs targeted to encourage producers to adopt the BMPs (Pannell, 2008). Because these BMPs increase wealth, adoption is more likely. If public benefits are also increased, as is the case in Scenario 1b for example, then the policy-maker or regulator has additional cause to devote resources to the encouragement of BMP adoption among private producers.

7.3 Study Limitations

Certain limitations to the study design should be noted and taken into consideration. Although representative production practices, economic parameters, and biophysical metrics were used, various simplifying assumptions were made due to the restriction of modeling techniques or a lack of precise data. These assumptions may bias the results or partially obscure the economic and environmental reality of agricultural production in the LLB watershed.

First, one limitation of the analysis in this thesis is a lack of an explicit link between residual N and P surplus found in the soil and the corresponding impact to water quality. Considered in isolation, residual N and P in soil is not necessarily “bad”, and in fact can be beneficial to future crop production when accounted for properly. The issue, however, is the risk that excess nutrients may be lost to the surrounding environment, specifically ground or surface water. In this study, it was assumed that a fixed percentage of N and P is removed from the soil system via either leaching and downward movement through the soil profile or over land in runoff. However, without improved nutrient transport modeling, a more precise determination of water quality impacts is not possible.

Second, lack of information specific to the study area limited the quantification of various economic and biophysical impacts from BMP adoption. For instance, the lack of literature information regarding the level of crop yield gain following manure application required the analysis to make certain simplifying assumptions. Similarly, the use of values obtained through the benefit transfer method for the value of water quality improvements may not be fully applicable to the study area and consequently do not adequately represent the change in public welfare arising from changes in nutrient balance. Other limitations include the assumptions involved in crop yield benefits following alfalfa, the availability of nutrients following legume green manures, and the level of biological fixation attributed to the different legume species.

Third, a large and important set of possible BMPs were not investigated for various reasons. Chief among these BMPs are those involving the management of pastureland, such as off-stream cattle watering, fencing of riparian areas, or changing of stocking rates. Because pasture and livestock grazing are such prominent land uses in the LLB watershed, the exclusion of this set of BMPs is a limitation, and was due to restrictions in the modelling techniques used. For instance, off-stream cattle watering does not impact the calculation of nutrient balances. It does, however, have an impact on nutrient transport to water bodies, which is equally important yet not possible to model solely using estimations of nutrient balance. Similarly, this study assumes that conversion of pastureland is a plausible outcome of future land use in the LLB watershed. Agronomic or ownership constraints may limit the likelihood of this conversion ever happening.

7.4 Further Research

Continued research and improved modeling in several areas can improve the analysis of economic and environmental outcomes in the LLB watershed. For instance, the field-specific targeting of BMPs for areas with demonstrated environmental issues (e.g., high residual soil P or N levels) may prove more cost-effective than the wholesale implementation of a BMP across the entire watershed. In this way, remediation of 'degraded' fields can occur. Future modeling of watershed scenarios could conduct up-to-date assessments of biophysical parameters in order to include this variation.

Another area of research may be an improvement in the evaluation of nutrient transport or loading in certain areas, which can help identify susceptible areas of the watershed. For example, PEWC and AAF (2014) recently produced a study to assess the economics and environmental impacts of nutrient management in two watersheds in Alberta. The BMPs implemented were very site-specific and specialized to the issues present in the geographic area.

Improved modeling of nutrient transport throughout the watershed will be essential to fully understand the downstream impacts of increased nutrient loads. Quantifying the nutrient contribution of each field via both subsurface and surface flow under a range of conditions would further enable the targeting of BMPs in the watershed and provide policymakers with a more advanced assessment of relevant tradeoffs.

An additional area of further research could include the incorporation of production and/or price risk to the economic modelling. This could be done using an expanded version of the simulation model, which would be capable of introducing additional management considerations. The introduction of risk would allow the analysis to more closely model the conditions faced by producers in the LLB watershed.

Lastly, conducting primary economic valuation at the study site of water quality improvement benefits would improve the economic evaluation of the BMP scenarios. The values used in this study were transferred from other study sites and likely do not fully capture the true public welfare impacts from changes in water quality. A more accurate assessment would provide decision-makers with more information from which to base their policy decision on.

References

Aasen, A. and M. Bjorge. 2009. Alberta Forage Manual. 2nd Edition. Alberta Agriculture and Forestry, Edmonton, Alberta.

Adderley, D.R., J.J. Schoenau, R.A. Holm, and P. Qian. 2006. Nutrient Availability and Yield of Wheat Following Field Peas and Lentil in Saskatchewan, Canada. *Journal of Plant Nutrition,* 29: 25-34.

Agriculture and Agri-Food Canada (AAFC). 2000. Agricultural Best Management Practices: Agriculture and Water Quality. Government of Canada, Ottawa, Ontario.

Agriculture and Agri-Food Canada (AAFC). 2013. Lower Little Bow River Watershed. http://www.agr.gc.ca/eng/?id=1298404927669. (Last Accessed May 20, 2016).

Agriculture and Agri-Food Canada (AAFC). 2014. Agriculture and Water Quality. http://www.agr.gc.ca/eng/science-and-innovation/agricultural-practices/water/watershed-protection/agriculture-and-water-quality/?id=1371491033072. (Last Accessed May 10, 2016).

Agronomic Interpretations Working Group (AIWG). 1995. Land Suitability Rating System for Agricultural Crops: 1. Spring-seeded small grains. Edited by W.W. Pettapiece. Tech Bull 1995-6E. Centre for Land and Biological Research Resources Research, Agriculture and Agri-Food Canada, Ottawa.

Alberta Agriculture and Forestry (AAF). 1993. Legume Green Manuring. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex133. (Last Accessed February 1, 2016).

Alberta Agriculture and Forestry (AAF). 2001. Fertilizer Requirements of Irrigated Alfalfa. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex739. (Last Accessed May 15, 2016).

Alberta Agriculture and Forestry (AAF). 2002. Fertilizer Application and Placement. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex621. (Last Accessed February 4, 2016).

Alberta Agriculture and Forestry (AAF). 2004a. Improving Soil Fertility With Green Manure Legume Crops – Frequently Asked Questions. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/faq7979. (Last Accessed February 1, 2016).

Alberta Agriculture and Forestry (AAF). 2004b. Alberta Fertilizer Guide. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex3894. (Last Accessed February 4, 2016).

Alberta Agriculture and Forestry (AAF). 2004c. Beneficial Management Practices: Environmental Manual for Crop Producers in Alberta. Alberta Agriculture and Forestry, Edmonton, Alberta.

Alberta Agriculture and Forestry (AAF). 2005a. Agricultural Land Resource Atlas of Alberta – Soil Groups of Alberta. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex10307. (Last Accessed May 15, 2016).

Alberta Agriculture and Forestry (AAF). 2005b. Agricultural Land Resource Atlas of Alberta – Manure Production Index for the Agricultural Area of Alberta. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex10335. (Last Accessed May 20, 2016).

Alberta Agriculture and Forestry (AAF). 2005c. Agricultural Land Resource Atlas of Alberta – Groundwater Quality Risk for the Agricultural Area of Alberta. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex10339. (Last Accessed May 20, 2016).

Alberta Agriculture and Forestry (AAF). 2005d. Soil and Nutrient Management of Alfalfa. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex10073. (Last Accessed May 15, 2016).

Alberta Agriculture and Forestry (AAF). 2005. Economics of Riparian Area Grazing. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/aesa9957. (Last Accessed January 27, 2016).

Alberta Agriculture and Forestry (AAF). 2006. Barley Production in Alberta. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/crop4933. (Last Accessed May 20, 2016).

Alberta Agriculture and Forestry (AAF). 2007a. Nutrient Management Planning Guide. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/epw11920. (Last Accessed February 1, 2016).

Alberta Agriculture and Forestry (AAF). 2007b. Alberta Soil Phosphorus Limits Project. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/sag11864. (Last Accessed May 20, 2016).

Alberta Agriculture and Forestry (AAF). 2008a. Pea-Wheat Rotation in Southern Alberta. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex12403/$file/100_515-1.pdf?OpenElement. (Last Accessed May 10, 2016).

Alberta Agriculture and Forestry (AAF). 2008b. Reference Guide – Agricultural Operation Practices Act (AOPA). Alberta Agriculture and Forestry, Environmental Stewardship Division, Edmonton, Alberta.

Alberta Agriculture and Forestry (AAF). 2009a. Fertilizer and Nutrient Management of Timothy Hay.

http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex12785. (Last Accessed February 4, 2016).

Alberta Agriculture and Forestry (AAF). 2009b. Nutrient Management on Intensively Managed Pastures. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/agdex12813/$file/130_538-1.pdf?OpenElement. (Last Accessed February 4, 2016).

Alberta Agriculture and Forestry (AAF). 2013. Alfalfa Autotoxicity – Frequently Asked Questions. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/faq14355. (Last Accessed February 3, 2016).

Alberta Agriculture and Forestry (AAF). 2014. Agriculture Statistics Yearbook 2013. http://www1.agric.gov.ab.ca/$Department/deptdocs.nsf/all/sdd15054. (Last Accessed February 4, 2016).

Alberta Agriculture and Forestry (AAF). 2015a. AgriProfit$ Pasture Cost and Returns Profiles. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/econ14302. (Last Accessed February 4, 2016).

Alberta Agriculture and Forestry (AAF). 2015b. AgriProfit$ Cropping Alternatives. http://www1.agriculture.alberta.ca/$department/deptdocs.nsf/all/econ15298. (Last Accessed February 4, 2016).

Alberta Agriculture and Forestry (AAF). 2015c. Average Farm Input Prices for Alberta. http://www.agric.gov.ab.ca/app21/farminputprices. (Last Accessed February 4, 2016).

Alberta Environment and Parks (AEP). 2009. Rangeland Health Assessment for Grassland, Forest and Tame Pasture. http://esrd.alberta.ca/lands-forests/grazing-range-management/documents/RangelandHealthAssessment-Revised-2009.pdf. (Last Accessed January 27, 2016).

Alberta Environment and Parks (AEP). 2014. Environmental Quality Guidelines for Alberta Surface Waters. 2014. Water Policy Branch, Policy Division. Edmonton, Alberta.

Alberta Soil Information Centre. 2016. AGRASID: Agricultural Region of Alberta Soil Inventory Database. Version 4.0. Alberta Agriculture and Forestry. http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/sag6903. (Last Accessed March 20, 2016).

Alipour, N., G. Akbari, Z.F. Kashani, and M. Mohammadi. 2013. Evaluating the Effects of Different Planting Densities and Inoculation With Rhizobium Bacteria on Morphological Characteristics and Yield of Faba Bean (cv. Barakat). *International Journal of Biosciences,* 3(7): 42-48.

Alternative Land Use Services (ALUS). 2016. How ALUS Works. http://www.alus.ca/how-alus-works/. (Last Accessed May 20, 2016).

April, J., F. Glover, J.P. Kelly, and M. Laguna. 2003. Practical Introduction to Simulation Optimization. Proceedings of the 2003 Winter Simulation Conference.

Baron, V.S. 2015. Personal Communication with V. Baron, Research Scientist, Agriculture and Agri-Food Canada.

Barry, D.A., D. Goorahoo, and M.J. Goss. 1993. Estimation of Nitrate Concentrations in Groundwater Using a Whole Farm Nitrogen Budget. *Journal of Environmental Quality,* 22: 767-775.

Belcher, K.W., C.K. Edwards, and B. Gray. 2001. Ecological Fiscal Reform and Agricultural Landscapes – Analysis of Economic Instruments: Conservation Cover Incentive Program. National Round Table on the Environment and the Economy, Ottawa, Ontario.

Belcher, K.W., M.M. Boehm, and R.P. Zentner. 2003. The Economic Value of Soil Quality Under Alternative Management in the Canadian Prairies. *Canadian Journal of Agricultural Economics,* 51: 175-196.

Bennett, D.R., and T.E. Harms. 2011. Crop Yield and Water Requirement Relationships for Major Irrigated Crops in Southern Alberta. *Canadian Water Resources Journal,* 36(2): 159-170.

Bergstrom, K. 2009. Personal Communication with Karla Bergstrom (via Dawn Trautman), Production Crops Economist, Alberta Agriculture and Rural Development.

Biederbeck, V.O., O.T. Bouman, C.A. Campbell, L.D. Bailey, and G.E. Winkleman. 1995. Nitrogen Benefits From Four Green-Manure Legumes in Dryland Cropping Systems. *Canadian Journal of Plant Science,* 76(2): 307-315.

Black, W.N., and R.P. White. 1973. Effects of Nitrogen, Phosphorus, Potassium, and Manure Factorially Applied to Potatoes in a Long-Term Study. *Canadian Journal of Soil Science,* 53(2): 205-211.

Boehm, M., B. Junkins, R. Desjardins, S. Kulshreshtha, and W. Lindwall. 2004. Sink Potential of Canadian Agricultural Soils. *Climatic Change,* 65: 297-314.

Bolinder, M.A., H.H. Janzen, E.G. Gregorich, D.A. Angers, and A.J. VandenBygaart. 2007. An Approach For Estimating Net Primary Productivity and Annual Carbon Inputs to Soil For Common Agricultural Crops in Canada. *Agriculture, Ecosystems and Environment,* 118: 29-42.

Boring, T.J. 2005. Integration of Potato Production, Dairy Forage and Manure Management Systems: Nitrogen Dynamics and Agronomic Implications. M.Sc. Thesis. Michigan State University, East Lansing, Michigan.

Bork, E.W., T. Thomas and B. McDougall. 2001. Herbage Response to Precipitation in Central Alberta Boreal Grasslands. *Journal of Range Management* 54: 243-248.

Boxall, P., M. Weber, O. Perger, M. Cutlac, and A. Samarawickrema. 2008. Preliminary Results from the Farm Behaviour Component of the Integrated Economic-Hydrologic Model for the Watershed Evaluation of Beneficial Management Practices Program. University of Alberta, Department of Resource Economics and Environmental Sociology.

Boyd, J., and S. Banzhaf. 2007. What are Ecosystem Services? The Need For Standardized Environmental Accounting Units. *Ecological Economics,* 63(2): 616-626.

Boydston, R.A., and A. Hang. 1995. Rapeseed (Brassica napus) Green Manure Crop Suppresses Weeds in Potato (Solanum tuberosum). *Weed Technology,* 9(4): 669-675.

Brandt, S.A. 1990. Indianhead Lentil as a Green Manure Substitute for Summerfallow. Pages 222-230 in *Frontiers in Prairie Agriculture*, Soil and Crops Workshop. University of Saskatchewan, Saskatoon, Saskatchewan.

Brandt, S.A. 1999. Management Practices for Black Lentil Green Manure for the Semi-Arid Canadian Prairies. *Canadian Journal of Plant Science,* 79(1): 11-17.

Brandt, S.A., and C.H. Keys. 1982. Effect of Crop Rotations on Soil Moisture Levels. In *Proceedings of Soils and Crops Workshop,* University of Saskatchewan, Saskatoon, Saskatchewan.

Bremer, E., D.A. Rennie, and R.J. Rennie. 1988. Dinitrogen Fixation of Lentil, Field Pea and Fababean Under Dryland Conditions. *Canadian Journal of Soil Science,* 68(3): 553-562.

Bremer, E., J. Miller., B. Olson, and D. Rogness. 2008. Nitrogen and Phosphorus Balance of the WEBs Lower Little Bow Watershed in Southern Alberta. Agriculture and Agri-Food Canada, Lethbridge, Alberta.

Buczko, U., R.O. Kuchenbuch, and B. Lennartz. 2010. Assessment of the Predictive Quality of Simple Indicator Approaches for Nitrate Leaching From Agricultural Fields. *Journal of Environmental Management,* 91: 1305-1315.

Burney, J.A., S.J. Davis, D.B. Lobell. 2010. Greenhouse Gas Mitigation By Agricultural Intensification. *Proceedings of the National Academy of Sciences,* 107(26): 12052-12057.

Caldwell, W.J. 1998. Land-Use Planning, the Environment, and Siting Intensive Livestock Facilities in the 21st Century. *Journal of Soil and Water Conservation,* 53(2): 102-106.

Campbell, C.A., G.P. Lafond, R.P Zentner, and Y.W. Jame. 1990. Crop Rotation Studies on the Canadian Prairies. Agriculture Canada, Ottawa, Ontario.

Campbell, C.A., G.P. Lafond, R.P. Zentner, and Y.W. Jame. 1994. Nitrate Leaching in a Udic Haploboroll as Influenced by Fertilization and Legumes. *Journal of Environmental Quality,* 23(1): 195-201.

Campbell, C.A., R.P. Zentner, B.C. Liang, G. Roloff, E.C. Gregorich, and B. Blomert. 2000. Organic C Accumulation in Soil Over 30 Years in Semiarid Southwestern Saskatchewan – Effect of Crop Rotations and Fertilizers. *Canadian Journal of Soil Science*, 80(1): 179-192.

Campbell, C.A. A.J. VandenBygaart, B. Grant, R.P. Zentner, B.G. McConkey, R. Lemke, E.G. Gregorich, and M. Fernandez. 2007a. Quantifying Carbon Sequestration in a Conventionally Tilled Crop Rotation Study in Southwestern Saskatchewan. *Canadian Journal of Soil Science*, 87(1): 23-38.

Campbell, C.A., A.J. VandenBygaart, R.P. Zentner, B.G. McConkey, W. Smith, R. Lemke, B. Grant, and P.G. Jefferson. 2007b. Quantifying Carbon in a Minimum Tillage Crop Rotation Study in Semiarid Southwestern Saskatchewan. *Canadian Journal of Soil Science,* 87(3): 235-250.

Campbell, C.A., G.P. Lafond, A.J. VandenBygaart, R.P. Zentner, R. Lemke, W.E. May, and C.B. Holzapfel. 2011. Effect of Crop Rotation, Fertilizer and Tillage Management on Spring Wheat Grain Yield and N and P Content in a Thin Black Chernozem: A Long-Term Study. *Canadian Journal of Plant Science,* 91(3): 467-483.

Carlson, L., and J. Bauder. 2005. Sugarbeet Agronomy 101. http://waterquality.montana.edu/farm-ranch/irrigation/other_crops/sugarbeet.html. (Last Accessed February 2, 2016).

Carlsson, G., and K. Huss-Danell. 2003. Nitrogen Fixation in Perennial Forage Legumes in the Field. *Plant and Soil,* 253: 353-372.

Carson, J.S. 2003. Introduction to Modelling and Simulation. Proceedings of the 2003 Winter Simulation Conference.

Chang, C., and T. Entz. 1996. Nitrate Leaching Losses Under Repeated Cattle Feedlot Manure Applications in Southern Alberta. *Journal of Environmental Quality,* 25(1): 145-153.

Chang, C., and H.H. Janzen. 1996. Long-Term Fate of Nitrogen From Annual Feedlot Manure Applications. *Journal of Environmental Quality*, 25(4): 785-790.

Cherr, C.M., J.M. Scholberg, and R. McSorley. 2006. Green Manure Approaches to Crop Production: A Synthesis. *Agronomy Journal,* 98(2): 302-319.

Copeland, T., and J.F. Weston. 1988. Financial Theory and Corporate Policy – 3rd Edition. Boston, MA: Addison Wesley Publishing.

Copeland, T.E., J.F. Weston, and K. Shastri. 2005. Financial Theory and Corporate Policy – 3rd Edition. Boston, MA: Addison Wesley Publishing.

Dale, V., and S. Polasky. 2007. Measures of the Effects of Agricultural Practices on Ecosystem Services. *Ecological Economics,* 64(2): 286-296.

De Jong, R., C.F. Drury, J.Y. Yang, and C.A. Campbell. 2009. Risk of Water Contamination by Nitrogen in Canada as Estimated by the IROWC-N Model. *Journal of Environmental Management,* 90: 3169-3181.

De Laporte, A., A. Weersink, and W. Yang. 2010. Ecological Goals and Wetland Preservation Choice. *Canadian Journal of Agricultural Economics*, 58(1): 131-150.

Denton, M.D., D.J. Pearce, and M.B. Peoples. 2013. Nitrogen Contributions From Faba-Bean (Vicia faba L.) Reliant on Soil Rhizobia or Inoculation. *Plant and Soil,* 365(1): 363-374.

Desjardins, R.L., W. Smith, B. Grant, C. Campbell, and R. Riznek. 2005. Management Strategies to Sequester Carbon in Agricultural Soils and to Mitigate Greenhouse Gas Emissions. *Climatic Change,* 70(1-2): 283-297.

Dias, V., and K. Belcher. 2015. Value and Provision of Ecosystem Services From Prairie Wetlands: A Choice Experiment Approach. *Ecosystem Services,* 15: 35-44.

Dollevoet, B.D. 2010. Farm Wealth Implications of Ecological Goods and Services Practices and Policies. M.Sc. Thesis. Department of Resource Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta.

Douglas, L., D. Laviolette-Brown, X. Ma, B. Shapka, and Z. Yu. 2013. Alberta Fababean Producers Manual 1.0. University of Alberta, Edmonton, Alberta.

Ducks Unlimited Canada (DUC). 2006. Ecological Goods and Services Fact Sheet Series. http://www.ducks.ca/assets/2012/06/nv1_eg.pdf. (Last Accessed May 20, 2016).

Dunn, R. 2009. Personal Communication with Robb Dunn (via Dawn Trautman), Land Management Specialist, Alberta Agriculture and Rural Development.

Dunn, R. 2011. Personal Communication with Robb Dunn (via Shuoyi Xie), Land Management Specialist, Alberta Agriculture and Rural Development.

Dyck, M., D. Puurveen., and J. Robertson. 2015. Effect of Forages in Crop Rotations on Soil Carbon Levels at the University of Alberta Breton Plots. In *Grassland Carbon Workshop*, April 2015, University of Alberta, Edmonton, Alberta.

Eghball, B., and J.E. Gilley. 1994. Beef Cattle Feedlot Manure Management. *Journal of Soil and Water Conservation*, 49(2): 113.

Eilers, W., R. MacKay, L. Graham, and A. Lefebvre (eds). 2010. Environmental Sustainability of Canadian Agriculture: Agri-Environmental Indicator Report Series – Report #3. Agriculture and Agri-Food Canada, Ottawa, Ontario.

Ellert, K.M. 1995. Cropping Systems for a Sustainable Future: Long-Term Studies in Alberta. Research Report, Agriculture and Agri-Food Canada, Lethbridge, Alberta.

Emmond, G.S. and R.J. Ledingham. 1972. Effects of Crop Rotation on Some Soil-Borne Pathogens of Potato. *Canadian Journal of Plant Sciences,* 52(4): 605-611.

Entz, M.H., V.S. Baron, P.M. Carr, D.W. Meyer, S. Smith, and W.P. McCaughey. 2002. Potential of Forages to Diversify Cropping Systems in the Northern Great Plains. *Agronomy Journal,* 94(2): 240-250.

Entz, M.H., Bullied, W.J., D.A. Forster, R. Gulden, and J.K. Vessey. 2001. Extraction of Subsoil Nitrogen by Alfalfa, Alfalfa-Wheat, and Perennial Grass Systems. *Agronomy Journal,* 93(3): 495-503.

Entz, M.H., P.D. Ominski, R. Mohr, A Schoofs, D. Forster, W.J. Bullied, S. Shirtliffe, and K.C. Bamford. 1995. Forages Improve Efficiency of Prairie Cropping Systems. http://www.ssca.ca/images/pdf/1997_conference_pdf/Forages_Improve_Efficiency_of_Prairie_Cropping_Systems.pdf. (Last Accessed February 1, 2016).

Environment and Climate Change Canada (ECCC). 2009. National Inventory Report 1990-2009: Greenhouse Gas Sources and Sinks in Canada, Part 2. Pollutant Inventories and Reporting Division, Environment and Climate Change Canada.

Environment and Climate Change Canada (ECCC). 2010. Eutrophication. https://www.ec.gc.ca/air/default.asp?lang=En&n=64CD2186-1. (Last Accessed May 10, 2016).

Environment and Climate Change Canada (ECCC). 2016. Technical Update to Environment and Climate Change Canada's Social Cost of Greenhouse Gas Estimates. Ottawa, Ontario.

Environmental Utilities Department (EUD). 2016. Water Treatment. http://www.city.medicine-hat.ab.ca/index.aspx?page=574. (Last Accessed May 20, 2016).

Evans, J.R., and D.L. Olson. 2002. Introduction to Simulation and Risk Analysis. Second Edition. New Jersey: Prentice Hall.

Fitch, L., and B. Adams. 1998. Can Cows and Fish Co-Exist? *Canadian Journal of Plant Science* 78: 191-198.

Galloway, J.N., A.R. Townsend, J.W. Erisman, M. Bekunda, Z. Cai, J.R. Freney, L.A. Martinelli, S.P. Seitzinger, and M.A. Sutton. Transformation of the Nitrogen Cycle: Recent Trends, Questions, and Potential Solutions. *Science,* 320(5878): 889-892.

Gan, Y.T., C.A. Campbell, H.H. Janzen, R.L. Lemke, P. Basnyat, and C.L. McDonald. 2009. Carbon Input to Soil From Oilseed and Pulse Crops on the Canadian Prairies. *Agriculture, Ecosystems and Environment,* 132: 290-297.

Government of Alberta (GoA). 2015. Climate Leadership Plan. http://www.alberta.ca/climate-leadership-plan.cfm. (Last Accessed May 20, 2016).

Haarsma, D. 2014. Spatial Analysis of Agricultural Land Conversion and Its Associated Drivers in Alberta. M.Sc. Thesis. Department of Resource Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta.

Haarsma, D. 2015. Personal Communication with Darren Haarsma, Potato Producer, Haarwest Farms, Spruce Grove, Alberta.

Hao, X., C. Chang, H.H. Janzen, G. Clayton, and B.R. Hill. 2006. Sorption of Atmospheric Ammonia by Soil and Perennial Grass Downwind From Two Cattle Feedlots. *Journal of Environmental Quality,* 35: 1960-1965.

Harapiak, J. 2007. Giving Field Peas an Extended Crop Production Credit. *Top Crop Manager*. http://www.topcropmanager.com/content/view/915/132/. (Last Accessed January 6, 2011 via D. Trautman).

Havlin, J.L., S.L. Tisdale, W.L. Nelson, and J.D. Beaton (eds). 2014. Soil Fertility and Fertilizers: An Introduction to Nutrient Management, 8^{th} Edition. Pearson, Inc. United States of America.

Health Canada. 2013. Guidelines for Canadian Drinking Water Quality: Guideline Technical Document – Nitrate and Nitrite. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Hoyt, P.B. 1990. Residual Effects of Alfalfa and Bromegrass Cropping on Yields of Wheat Grown for 15 Subsequent Years. *Canadian Journal of Soil Science,* 70: 109-113.

Hoyt, P.B. and A. Hennig. 1971. Effect of a Forage Legume Species on Soil Moisture, Nitrogen, and Yield of Succeeding Barley Crops. *Canadian Journal of Soil Science,* 51: 177-183.

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II, and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Geneva, Switzerland.

Janzen, H.H., C.A. Campbell, R.C. Izaurralde, B.H. Ellert, N. Juma, W.B. McGill, and R.P. Zentner. 1998. Management Effects on Soil C Storage on the Canadian Prairies. *Soil & Tillage Research,* 47: 181-195.

Janzen, H.H., K.A. Beauchemin, Y. Bruinsma, C.A. Campbell, R.L. Desjardins, B.H. Ellert, and E.G. Smith. 2003. The Fate of Nitrogen in Agroecosystems: An Illustration Using Canadian Estimates. *Nutrient Cycling in Agroecosystems,* 67: 85-102.

Jarvis, P.J., C.C. Boswell, A.K. Metherell, R.M. Davison, and J.A. Murphy. 2002. A Nutrient Budget for the Meat and Wool Economic Service of a New Zealand Class 1 High-Country Farm Model. *New Zealand Journal of Agricultural Research,* 45:1-15.

Jeffrey, S., B. Cortus, B. Dollevoet, S. Koeckhoven, D. Trautman, and J. Unterschultz. 2012. Farm Level Economics of Ecosystem Service Production. Linking Environment and Agriculture Research Network, University of Alberta, Edmonton, Alberta.

Jensen, E.S., M.B. Peoples, and H. Hauggaard-Nielsen. 2010. Faba Bean in Cropping Systems. *Field Crops Research,* 115(3): 203-216.

Johnson, C.J., P.A. Bonrud, T.L. Dosch, A.W. Kilness, K.A. Senger, D.C. Busch, and M.R. Meyer. 1987. Fatal Outcome of Methemoglobinemia in an Infant. *Journal of American Medical Association,* 257(20): 27-97.

Johnson, K.A., S. Polasky, E. Nelson, 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: 71-79.

Kalischuk, A., M. Lewis, M. Kadijk, R. Bennett, D. Chanasyk, J. Miller, and B. Olson. 2008. Field Runoff and Rainfall Simulation Evaluations From a Buffer Along the Lower Little Bow River. Agriculture and Agri-Food Canada, Lethbridge, Alberta.

Karlen, D.L., T.S. Colvin, and M.D. Due. 1995. Nutrient, Labor, Energy and Economic Evaluations of Two Farming Systems in Iowa. *Journal of Production Agriculture,* 8(4): 540-546.

Kelner, D.J., J.K. Vessey, and M.H. Entz. 1997. The Nitrogen Dynamics of 1-,2- and 3-Year Stands of Alfalfa in a Cropping System. *Agriculture, Ecosystems and Environment,* 64: 1-10.

Khakbazan, M., C.A. Grant, J. Huang, E.G. Smith, J.T. O’Donovan, R.E. Blackshaw, K.N. Harker, G.P. Lafond, E.N. Johnson, Y.Gan, W.E. May, T.K. Turkington, and N.Z. Lupwayi. 2014. Economic Effects of Preceding Crops and Nitrogen Application on Canola and Subsequent Barley. *Agronomy Journal,* 106(6): 2055-2066.

Kilshreshtha, S., M. Undi, J. Zhang, M. Ghorbani, K. Wittenberg, A. Stewart, E. Salvano, E. Kebreab, and K. Ominski. 2015. Challenges and Opportinities in Estimating the Value of Goods and Services in Temperate Grasslands – A Case Study of Prairie Grasslands in Manitoba, Canada. *Agroecology*, Prof. V. Pilipavicius (Ed.), InTech.

Koeckhoven, S. 2008. Economics of Agricultural Best Management Practices in the Lower Little Bow Watershed. M.Sc. Thesis. Department of Rural Economy, University of Alberta, Edmonton, Alberta.

Koelsch, R. 2005. Evaluating Livestock System Environmental Performance with Whole-Farm Nutrient Balance. *Journal of Environmental Quality,* 34: 149-155.

Kohn, J., D.X. Soto, M. Iwanyshyn, B. Olson, A. Kalischuk, K. Lorenz, and M.J. Hendry. 2015. Groundwater Nitrate and Chloride Trends in an Agriculture-Intensive Area in

Southern Alberta, Canada. *Water Quality Research Journal of Canada*, 51(1): 47-59.

Kopke, U., and T. Nemecek. 2010. Ecological Services of Faba Bean. *Field Crops Research,* 115(3): 217-233.

Krobel, R., R. Lemke, C.A. Campbell, R. Zentner, B. McConkey, H. Steppuhn, R. De Jong, and H. Wang. 2014. Water Use Efficiency of Spring Wheat in the Semi-Arid Canadian Prairies: Effect of Legume Green Manure, Type of Spring Wheat, and Cropping Frequency. *Canadian Journal of Soil Science,* 94(2): 223-235.

Krysel, C., E.M. Boyer, C. Parson, and P. Welle. 2003. Lakeshore Property Values and Water Quality: Evidence From Property Sales in the Mississippi Headwaters Region. Report submitted to the Legislative Commission on Minnesota Resources. Mississippi Headwater Commission and Bemidji State University.

Kryzanowski, L., D. Laverty, J. Ashworth, M. Nyborg, J. Harapiak, and R. Kucey. 1988. Soil Test Recommendations for Alberta: Technical Manual. Soil Test Technical Advisory Group, Alberta Agriculture, Edmonton, Alberta.

Lafond, G.P., W.E. May, E.N. Johnson, and S.A. Brandt. 2007. Frequency of Field Pea in Rotation: the Issues and Opportunities. Presented at the Saskatchewan Soil Conservation Association Annual Meeting, January 2007. Saskatook, Saskatchewan.

Lal, R. 2004. Soil Carbon Sequestration to Mitigate Climate Change. *Geoderma,* 123: 1-22.

Lalane, R., B. Gagnon, and R.R. Simard. 2003. Papermill Biosolid and Hog Manure Compost Affect Short-Term Biological Activity and Crop Yield of a Sandy Soil. *Canadian Journal of Soil Sciences,* 83(4): 352-362.

Lamb, J.F., D.K. Barnes, M.P. Russelle, C.P. Vance, G.H. Heichel, and K.I. Henjum. 1995. Ineffectively and Effectively Nodulated Alfalfas Demonstrate Biological Nitrogen Fixation Continues With High Nitrogen Ferilization. *Crop Science,* 35(1): 153-157.

Leyshon, A.J., M.R. Kilcher, and J.D. McElgunn. 1981. Seeding rates and row spacings for three forage crops grown alone or in alternate grass-alfalfa rows in southwestern Saskatchewan. *Canadian Journal of Plant Science,* 61: 711-717.

Little, J. L., K.A. Saffron, and L. Fent. 2003. Land use and water quality relationships in the lower Little Bow River watershed, Alberta, Canada. *Water Quality Research Journal of Canada*, 38(4): 563-584.

Liu, F., C.C. Mitchell, D.T. Hill, J.W. Odom, and E.W. Rochester. 1997. Phosphorus Recovery in Surface Runoff From Swine Lagoon Effluent by Overland Flow. *Journal of Environmental Quality,* 26(4): 995-1001.

Lupwayi, N.Z., T. Lea, J.L. Beaudoin, G.W. Clayton. 2005. Soil Microbial Biomass, Functional Diversity and Crop Yields Following Application of Cattle Manure, Hog Manure and Inorganic Fertilizers. *Canadian Journal of Soil Science,* 85(2): 193-201.

Lutwick, G.W., and D.N. Graveland. 1978. A Study of the Phosphorus Adsorption Capacities of Several Southern Alberta Soils. Technical Report, Alberta Environment, Technical Development Branch, Edmonton, Alberta.

Maas, S.E., A.J. Glenn, M. Tenuta, and B.D. Amiro. 2013. Net CO_2 and N_2O Exchange During Perennial Forage Establishment in an Annual Crop Rotation in the Red River Valley, Manitoba. *Canadian Journal of Soil Science,* 93(5): 639-652.

Manitoba Agriculture, Food, and Rural Initiatives (MAFRI). 2010. The Benefits of Including Forages in Your Crop Rotation. https://www.gov.mb.ca/agriculture/crops/production/forages/benefits-of-including-forages-in-your-crop-rotation.html. (Last Accessed February 1, 2016).

Manitoba Agriculture, Food and Rural Development (MAFRD). 2012. Green Manure. http://www.gov.mb.ca/agriculture/crops/production/forages/print,green-manure.html. (Last Accessed May 20, 2016).

Manitoba Agriculture, Food and Rural Development (MAFRD). 2015. Production Economics – Cost of Production Budgets. http://www.gov.mb.ca/agriculture/business-and-economics/financial-management/cost-of-production.html#crops-and-forage. (Last Accessed February 4, 2016).

Marbek Resource Consultants Ltd (MRC). 2010. Nutrient Reduction Strategies for Wastewater Treatment Facilities for Lake Winnipeg: Cost-Benefit Analysis Using an Ecological Goods and Services Approach. Report submitted to Environment Canada.

Mapfumo, E., M.A. Naeth, V.S. Baron, A.C. Dick, and D.S. Chanasyk. 2002. Grazing Impacts on Litter and Roots: Perennial Versus Annual Grasses. *Journal of Range Management*, 55: 16-22.

McCallum, J.E., M.C. Ryan, B. Mayer, and S.J. Rodvang. 2008. Mixing-Induced Groundwater Denitrification Beneath a Manured Field in Southern Alberta, Canada. *Applied Geochemistry,* 23: 2146-2155.

McConkey, B. Personal Communication with Brian McConkey, Research Scientist, Agriculture and Agri-Food Canada.

McConkey, B., D. Angers, M. Bentham, M. Boehm, T. Brierley, D. Cerkowniak, C. Liang, P. Collas, H. de Gooijer, R. Desjardins, S. Gameda, B. Grant, T. Huffman, J.

Hutchinson, L. Hill, P. Krug, C. Monreal, T. Martin, G. Patterson, P. Rochette, W. Smith, B. Vandenbygaart, X. Verge, and D. Worth. 2014. Canadian Agricultural Greenhouse Gas Monitoring Accounting and Reporting System: Methodology and Greenhouse Gas Estimates for Agricultural Land in the LULUCF Sector for NIR 2014. Environment Canada.

McKenzie, R.H., E. Bremer, P.G. Pfiffner, A.B. Middleton, T. Dow, M. Oba, A. Efetha, and R. Hohm. 2009. Yield and Quality Responses of Irrigated Timothy to Fertilizer Application in Southern Alberta. *Canadian Journal of Plant Science,* 89(2): 247-255.

Meisinger, J.J., and G.W. Randall. 1991. Estimating Nitrogen Budgets for Soil-Crop Systems, p. 85-125, *in* R.F. Follett, D.R. Keeney and R.M. Cruse, eds. Managing Nitrogen for Groundwater Quality and Farm Profitability. Soil Science Society of America, Madison, Wisconsin.

Miller, D.A. 1996. Allelopathy in Forage Crop Systems. *Agronomy Journal,* 88(6): 854-859.

Miller, J.D. 2002. Economics of Riparian Grazing Management. M.Sc. Thesis. Department of Resource Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta.

Miller, J.J. 2014. Personal Communication with Jim Miller, Research Scientist, Agriculture and Agri-Food Canada.

Miller, J.J., N.J. Sweetland, and C. Chang. 2002. Hydrological Properties of a Clay Loam Soil After Long-Term Cattle Manure Application. *Journal of Environmental Quality,* 31(3): 989-996.

Miller, J.J., B. Beasley, D. Chanasyk, T. Curtis, C. Dyck, A. Kalischuk, D. Lastuka, M. Lewis, B. Olson, S. Petry, M. Rahbeh, D. Rogness, C. Ross, K. Schmitt, E. Smith, J. Villeneuve, and W. Willms. 2008a. Spatial Analyses of the WEBs Lower Little Bow Watershed. Agriculture and Agri-Food Canada, Lethbridge, Alberta.

Miller, J.J., D. Chanasyk, T.Curtis, A. Kalischuk, D. Lastuka, M. Lewis, B. Olson, S. Petry, M. Rahbeh, D. Rogness, C. Ross, K. Schmitt, E. Smith, J. Villeneuve, and W. Willms. 2008b. Influence of Conversion from Barley to Alfalfa on Runoff Quantity and Quality. Agriculture and Agri-Food Canada, Lethbridge, Alberta.

Miller, J.J., T.W. Curtis, E. Bremer, D.S. Chanasyk, and W.D. Willms. 2009. Soil Test Phosphorus and Nitrate Adjacent to Artificial and Natural Cattle Watering Sites in Southern Alberta. *Canadian Journal of Soil Science,* 90(2): 331-340.

Miller, J.J., D.S. Chanasyk, T.W. Curtis, and W.D. Willms. 2010. Influence of Streambank Fencing on the Environmental Quality of Cattle-Excluded Pastures. *Journal of Environmental Quality,* 39(3): 991-1000.

Miller, J.J., D.S. Chanasyk, T.W. Curtis, T. Entz, and W. Willms. 2011a. Environmental Quality of Lower Little Bow River and Riparian Zone Along an Unfenced Reach With Off-Stream Watering. *Agricultural Water Management,* 98(10): 1505-1515.

Miller, J.J., D.S. Chanasyk, T.W. Curtis, and B.M. Olson. 2011b. Phosphorus and Nitrogen in Runoff After Phosphorus- or Nitrogen-Based Manure Applications. *Journal of Environmental Quality,* 40(3): 949-958.

Miller, J.J., Curtis, T., Chanasyk, D.S., and Reedyk, S. 2015. Influence of Mowing and Narrow Grass Buffer Widths on Reductions in Sediment, Nutrients, and Bacteria in Surface Runoff. *Canadian Journal of Soil Science,* 95(2): 139-151.

Miller, J.J., Curtis, T., Chanasyk, D.S., Reedyk, S., and W. Willms. 2016. Effectiveness of Soil in Vegetated Buffers to Retain Nutrients and Sediment Transported by Concentrated Runoff Through Deep Gullies. *Canadian Journal of Soil Science,* 96(2): 154-168.

Mozumder, P., and R.P. Berrens. 2007. Inorganic Fertilizer Use and Biodiversity Risk: An Empirical Investigation. *Ecological Economics,* 62(3): 538-543.

Mun, J. 2006. Modeling Risk: Applying Monte Carlo Simulation, Real Options Analysis, Forecasting, and Optimization Techniques. Hoboken, NJ: John Wiley and Sons.

Nelson, N.M., J.B. Loomis, P.M. Jakus, M.J. Kealy, N. von Stackelburg, and J. Ostermiller. 2015. Linking Ecological Data and Economics to Estimate the Total Economic Value of Improving Water Quality By Reducing Nutrients. *Ecological Economics,* 118: 1-9.

Nitschelm, J.J., and P.J. Reginig. 2005. Effect of Composted Cattle Manure on Sugar Beet Production. *Journal of Sugar Beet Research,* 33: 52-57.

Noorbakhsh, S., J. Schoenau, B. Si, T. Zeleke, and P. Qian. 2008. Soil Properties, Yield, and Landscape Relationships in South-Central Saskatchewan Canada. *Journal of Plant Nutrition,* 31: 539-556.

Nord, E.A., and L.E. Lanyon. 2003. Managing Material Transfer and Nutrient Flow in an Agricultural Watershed. *Journal of Environmental Quality,* 32: 562-570.

Nordhaus, W.D. 2007. A Review of the "Stern Review on the Economics of Climate Change". *Journal of Economic Literature*, 45: 686-702.

O'Donovan. J.T., C.A. Grant, R.E. Blackshaw, K.N. Harker, E.N. Johnson, Y. Gan, G.P. Lafond, W.E. May, T.K. Turkington, N Z. Lupwayi, F.C. Stevenson, D.L. McLaren, M. Khakbazan, and E.G. Smith. 2014. Rotational Effects of Legumes and Non-Legumes on Hybrid Canola and Malting Barley. *Agronomy Journal,* 106(6): 1921-1932.

Oldman Watershed Council (OWC). 2010. Oldman River State of the Watershed Report 2010. Oldman Watershed Council, Lethbridge Alberta, Canada.

Olewiler, N. 2004. The Value of Natural Capital in Settled Areas of Canada. Published by Ducks Unlimited Canada and the Nature Conservancy of Canada.

Olson, B. 2015. Personal Communication with Barry Olson, Research Scientist, Alberta Agriculture and Rural Development.

Olson, B.M., D.R. Bennett, R.H. McKenzie, T.D. Ormann, and R.P. Atkins. 2009. Nitrate Leaching in Two Irrigated Soils With Different Rates of Cattle Manure. *Journal of Environmental Quality,* 38(6): 2218-2228.

Olson, B.M., E. Bremer, R.H. McKenzie, and D.R. Bennett. 2010. Phosphorus Accumulation and Leaching in Two Irrigated Soils With Incremental Rates of Cattle Manure. *Canadian Journal of Soil Science,* 90(2): 355-362.

Pannell, D.J. 2008. Public Benefits, Private Benefits, and Policy Mechanism Choice for Land-Use Change for Environmental Benefits. *Land Economics,* 84(2): 225-240.

Parton, W.J., D.S. Schimel, C.V. Cole, and D.S. Ojima. 1987. Analysis of Factors Controlling Soil Organic Matter Levels in Great Plains Grasslands. *Soil Science Society of America Journal*, 51(5): 1173-1179.

Paterson Earth and Water Consulting Ltd. (PEWC) and Alberta Agriculture and Forestry (AAF). 2014. Nutrient Beneficial Management Practices Evaluation Project. Volume 1: Summary and Recommendations. Lethbridge, Alberta.

Paustian, K.A., O. Andren, H.H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Van Noordwijk, and P.L. Woomer. 1997. *Soil Use and Management,* 13: 230-244.

Penney, D. 2015. Personal Communication with Doug Penney, Senior Agri-Coach, Agri-Trend.

PennState Extension (PSE). 2015. Inoculation of Legumes for Maximum Nitrogen Fixation. http://extension.psu.edu/plants/crops/forages/successful-forage-establishment/inoculation-of-legumes-for-maximum-nitrogen-fixation. (Last Accessed February 4, 2016).

Putnam, D., M. Russelle. S. Orloff, J. Kuhn, L. Fitzhugh, L. Godfrey, A. Kiess, and R. Long. 2001. Alfalfa, Wildlife and the Environment: The Importance and Benefits of Alfalfa in the 21st Century. California Alfalfa and Forage Association.

Rahbeh, M., D. Chanasyk, and J. Miller. 2011. Two-Way Calibration-Validation of SWAT Model for a Small Prairie Watershed With Short Observed Record. *Canadian Water Resources Journal,* 36(3): 247-270.

Rahbeh, M., D. Chanasyk, and J. Miller. (2013). Modelling the effect of irrigation on the hydrological output from a small prairie watershed. *Canadian Water Resources Journal,* 38(4): 280-295.

Rajsic, P., and A. Weersink. 2008. Do Farmers Waste Fertilizer? A Comparison of *Ex Post* Optimal Nitrogen Rates and *Ex Ante* Recommendations by Model, Site and Year. *Agricultural Systems,* 97: 56-67.

Rajsic, P., A. Weersink, and M. Gandorfer. 2009. Risk and Nitrogen Application Levels. *Canadian Journal of Agricultural Economics*, 57(2): 223-239.

Reddy, D.D., A.S. Rao, T.R. Rupa. 2000. Effects of Continuous Use of Cattle Manure and Fertilizer Phosphorus on Crop Yields and Soil Organic Phosphorus in a Vertisol. *Bioresource Technology,* 75(2): 113-118.

Rennie, R.J., and S. Dubetz. 1986. Nitrogen-15-determined Nitrogen Fixation in Field-Grown Chickpea, Lentil, Fababean, and Field Pea. *Agronomy Journal*, 78(4): 654-660.

Rivest, S. 2009. The Watershed Evaluation of Beneficial Management Practices (WEB's) The Bras d'Henri Watershed – On-Farm Economics. M.Sc. Thesis. Department of Natural Resource Sciences, McGill University, Montreal, Quebec.

Rock, L., and B. Mayer. 2004. Nitrogen Budget for the Oldman River Basin, Southern Alberta, Canada. *Nutrient Cycling in Agroecosystems,* 75(1): 147-162.

Rodvang, S.J., R. Schmidt-Bellach, and L.I. Wassenaar. 1998. Nitrate in Groundwater Below Irrigated Fields in Southern Alberta. CAESA Project RES-041-93. Alberta Agriculture and Forestry, Lethbridge, Alberta.

Rodvang, S. J., D. Mikalson, and M.C. Ryan. 2004. Changes in Ground Water Quality in an Irrigated Area of Southern Alberta. *Journal of environmental quality,* 33(2): 476-487.

Roessel, R.V. 2012. Personal Communication with R. Roessel (via Shuoyi Xie), Chair, Potato Growers of Alberta, Courtland Hill Farm Ltd.

Ross, S.A., R.W. Westerfield, J.F. Jaffe, and G.S. Roberts. 2003. Corporate Finance – 3rd Canadian Edition. Toronto, ON: McGraw-Hill Ryerson.

Russelle, M. 2004. The Environmental Impacts of N_2 Fixation by Alfalfa. 2004 National Alfalfa Symposium, San Diego, California.

Saskatchewan Agriculture, Food and Rural Revitalization (SAFRR). 2005. Soil Improvements with Legumes. https://www.saskatchewan.ca/business/agriculture-natural-resources-and-industry/agribusiness-farmers-and-ranchers/crops-and-irrigation/soils-fertility-and-nutrients/soil-improvements-with-legumes. (Last Accessed February 1, 2016).

Saskatchewan Agriculture, Food and Rural Revitalization (SAFRR). 2016. Beneficial Management Practices Guidelines. https://www.saskatchewan.ca/business/agriculture-natural-resources-and-industry/agribusiness-farmers-and-ranchers/growing-forward-2/environment-programs/farm-stewardship-program/beneficial-management-practices. (Last Accessed May 10, 2016).

Scott, J.L. 2012. Evaluating Water Quality and Biotic Indices in the Lower Little Bow River, Alberta. M.Sc. Thesis. Department of Renewable Resources, University of Alberta, Edmonton, Alberta.

Schlesinger, W.H. 1999. Carbon Sequestration in Soils. *Science,* 284(5423): 2095.

Sharpe, W.F., G.J. Alexander, J.V. Bailey, D.J. Fowler, and D.L. Domain. 2000. Investments – 3rd Edition. Scarborough, ON: Prentice Hall.

Sharpley, A., J.J. Meisinger, A. Breeuwsma, J.T. Sims, T.C. Daniel, and J.S. Schepers. 1998. Impacts of Animal Manure Management on Ground and Surface Water Quality. In Hatfield, J.L., and B.A. Stewart (Eds.), *Animal Waste Utilization: Effective Use of Manure as a Soil Resource.* CRC Press, Florida, United States.

Sharpley, A., and B. Moyer. 2000. Phosphorus Forms in Manure and Compost and Their Release During Simulated Rainfall. *Journal of Environmental Quality,* 29(5): 1462-1469.

Sims, J.T., R.R. Simard, and B.C. Joern. 1998. Phosphorus Loss in Agricultural Drainage: Historical Perspectives and Current Research. *Journal of Environmental Quality,* 27: 277-293.

Sincik, M., Z.M. Turan, and A.T. Goksoy. 2008. Responses of Potato (Solanum tuberosum L.) to Green Manure Cover Crop and Nitrogen Fertilization Rates. *American Journal of Potato Research,* 85(2): 150-158.

Smith, E.G. 2011. Personal Communication with Elwin Smith (via Shuoyi Xie), Bioeconomist, Agriculture and Agri-Food Canada.

Smith, E.G., and J.J. Miller. 2008. Economics of Livestock Manure Management BMPs in the WEBs Lower Little Bow Watershed. Agriculture and Agri-Food Canada, Lethbridge, Alberta.

Smith, W.N., R.L. Desjardins, and B. Grant. 2001. Estimated Changes in Soil Carbon Associated With Agricultural Practices in Canada. *Canadian Journal of Soil Science*, 81(2): 221-227.

Soulodre, E. 2007. Personal Communication with Etienne Soulodre (via S. Koeckhoven), Range Ecologist, Saskatchewan Watershed Authority.

Sommerfeldt, T.G., and C. Chang. 1985. Changes in Soil Properties Under Annual Applications of Feedlot Manure and Different Tillage Practices. *Soil Science Society of America Journal,* 49(4): 983-987.

Spalding, R.F., and M.E. Exner. 1993. Occurrence of Nitrate in Groundwater – A Review. *Journal of Environmental Quality,* 22(3): 392-402.

St.Luce, M., C.A. Grant, B.J. Zebarth, N. Ziadi, J.T. O'Donovan, R.E. Blackshaw, K.N. Harker, E.N. Johnson, Y. Gan, G.P. Lafond, W.E. May, M. Khakbazan, E.G. Smith. 2015. Legumes Can Reduce Economic Optimum Nitrogen Rates and Increase Yields in a Wheat-Canola Cropping Sequence in Western Canada. *Field Crops Research,* 179: 12-25.

Statistics Canada. 2011. Census of Agriculture. Statistics Canada, Ottawa, Ontario.

Stern, N. 2006. The Economics of Climate Change: The Stern Review. Cambridge, UK: Cambridge University Press.

Stevenson, F.C., and C. van Kessel. 1996. The Nitrogen and Non-Nitrogen Rotation Benefits of Pea to Succeeding Crops. *Canadian Journal of Plant Science,* 76: 735-745.

Sturgul, S.J., T.C. Daniel, D.H. Mueller. 1990. Tillage and Canopy Cover Effects on Interrill Erosion From First-Year Alfalfa. *Soil Science Society of America Journal,* 54(6): 1733-1739.

Swinton, S.M., and R.P. King. 1991. Evaluating Robust Regression Technique for Detrending Crop Yield Data with Non-Normal Error. *American Journal of Agricultural Economics,* 73(2): 446-451.

Thangaraj, R. 2015. Personal Communication with Rawlin Thangaraj, Production Crops Economist, Alberta Agriculture and Forestry.

Tol, R.S. 2009. The Economic Effects of Climate Change. *Journal of Economic Perspectives,* 23: 29-51.

Trautman, D. 2012. The Economics of Beneficial Management Practices Adoption on Representative Alberta Crop Farms. M.Sc. Thesis. Department of Resource

Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta.

Twerdoff, D.A., D.S. Chanasyk, M.A. Naeth, V.S. Baron, and E. Mapfumo. 1999. Soil Water Regimes of Rotationally Grazed Perennial and Annual Forages. *Canadian Journal of Soil Science,* 79: 627-637.

United States Department of Agriculture (USDA). 2016. Conservation Reserve Program. http://www.fsa.usda.gov/programs-and-services/conservation-programs/conservation-reserve-program/. (Last Accessed May 20, 2016).

Untershultz, J.R., J. Miller, and P.C. Boxall. 2004. The On-Range Economics of Riparian Zone Cattle Grazing Management. *Environmental Management,* 33(5): 664-676.

VandenBygaart, A.J., E.G. Gregorich, and D.A. Angers. 2003. Influence of Agricultural Management on Soil Organic Carbon: A Compendium and Assessment of Canadian Studies. *Canadian Journal of Soil Science,* 83(4): 363-380.

VandenBygaart, A.J., B.G. McConkey, D.A. Angers, W. Smith, H. De Gooijer, M. Bentham, and T. Martin. 2008. Soil Carbon Change Factors for the Canadian Agriculture National Greenhouse Gas Inventory. *Canadian Journal of Soil Science,* 88(5): 671-680.

Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and D.G. Tilman. 1997. Human Alteration of the Global Nitrogen Cycle: Sources and Consequences. *Ecological Applications,* 7(3): 737-750.

Walley, F.L., G.W. Clayton, P.R. Miller, P.M. Carr, G.P. Lafond. 2007. Nitrogen Economy of Pulse Crop Production in the Northern Great Plains. *Agronomy Journal,* 99(6): 1710-1718.

Wheeler, E.J. 1946. The Residual Effect of Crop Rotations on Potato Yield and Presence of Potato Scab. *Michigan State College Agriculture Experiment Station Quarterly Bulletin,* 28: 326-332.

Williams, C.M., J.R. King, S.M. Ross, M.A. Olson, C.F. Hoy, and K.J. Lopetinsky. 2014. Effects of Three Pulse Crops on Subsequent Barley, Canola, and Wheat. *Agronomy Journal,* 106(2): 343-350.

World Health Organization (WHO). 2011. Nitrate and Nitrite in Drinking-Water: Background Document for Development of WHO Guidelines for Drinking-Water Quality. Geneva, Switzerland.

Xie, S. 2014. Farm-Level Costs and Benefits of Beneficial Management Practices for Irrigated Crop Production Systems in Southern Alberta. M.Sc. Thesis. Department of Resource Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta.

Yang, J.Y., C.F. Drury, X.M. Yang, R. De Jong, E.C. Huffman, C.A. Campbell, and V. Kirkwood. 2010. Estimating Biological N_2 Fixation in Canadian Agricultural Land Using Legume Yields. *Agriculture, Ecosystems and Environment,* 137(1): 192-201.

Yang, J.Y., T. Huffman, C.F. Drury, X.M. Yang, R. De Jong, and C.A. Campbell. 2011. Estimating Changes of Residual Soil Nitrogen in Chernozemic Soils in Canada. *Canadian Journal of Soil Science,* 92: 481-491.

Yang, W. and A. Weersink. 2004. Cost-Effective Targeting of Riparian Buffers. *Canadian Journal of Agricultural Economics,* 52(1): 17-34.

Zebarth, B.J., and C.J. Rosen. 2007. Research Perspective on Nitrogen BMP Development for Potato. *American Journal of Potato Research,* 84(1): 3-18.

Zentner, R.P., S.A. Brandt, K.J. Kirkland, C.A. Campbell, and G.J. Sonntag. 1992. Economics of Rotation and Tillage Systems for the Dark Brown Soil Zone of the Canadian Prairies. *Soil & Tillage Research,* 24(3): 271-284.

Zentner, R.P., C.A. Campbell, V.O. Biederbeck, F. Selles, R. 1996. Indianhead Black Lentil as Green Manure for Wheat Rotations in the Brown Soil Zone. *Canadian Journal of Plant Science,* 76(3): 417-422.

Zentner, R.P., G.P. Lafond, D.A. Derksen, and C.A. Campbell. 2002. Tillage Method and Crop Diversification: Effect on Economic Returns and Riskiness of Cropping Systems in a Thin Black Chernozem of the Canadian Prairies. *Soil & Tillage Research,* 67: 9-21.

Zentner, R.P., C.A. Campbell, V.O. Biederbeck, F. Selles, R. Lemke, P.G. Jefferson, and Y.Gan. 2004. Long-Term Assessment of Management of an Annual Legume Green Manure Crop for Fallow Replacement in the Brown Soil Zone. *Canadian Journal of Plant Science,* 84(1): 11-22.

Appendix A: Field Land Use Designations and Biophysical Characteristics

Table A.1. Acreage and Land Use Category of 65 Fields in the Lower Little Bow Watershed.

Number	Acres	Irrigated / Dryland	Land Use	Riparian/Upland	Tame/Native
1	80	Dryland	Crop	-	-
2	120	Dryland	Crop	-	-
3	40	Dryland	Crop	-	-
4	40	Dryland	Crop	-	-
5	40	Dryland	Crop	-	-
6	40	Dryland	Crop	-	-
7	120	Dryland	Crop	-	-
8	160	Dryland	Crop	-	-
9	120	Dryland	Crop	-	-
10	40	Dryland	Crop	-	-
11	80	Dryland	Pasture	Upland	Native
12	120	Irrigated	Crop	-	-
13	160	Irrigated	Crop	-	-
14	120	Irrigated	Crop	-	-
15	120	Dryland	Pasture	Upland	Native
16	40	Dryland	Pasture	Upland	Native
17	120	Irrigated	Crop	-	-
18	160	Irrigated	Crop	-	-
19	160	Irrigated	Crop	-	-
20	120	Dryland	Pasture	Riparian	Native
21	40	Dryland	Pasture	Upland	Native
22	160	Dryland	Crop	-	-
23	160	Dryland	Pasture	Riparian	Native
24	120	Irrigated	Crop	-	-
25	80	Dryland	Pasture	Upland	Native
26	160	Dryland	Pasture	Upland	Native
27	160	Irrigated	Crop	-	-
28	80	Dryland	Pasture	Riparian	Native
29	120	Dryland	Pasture	Upland	Native
30	160	Dryland	Pasture	Riparian	Native
31	80	Irrigated	Crop	-	-
32	120	Dryland	Pasture	Upland	Tame
33	40	Dryland	Pasture	Upland	Tame
34	80	Irrigated	Pasture	Riparian	Tame
35	160	Dryland	Pasture	Riparian	Native
36	160	Dryland	Pasture	Riparian	Native
37	40	Irrigated	Pasture	Upland	Tame

38	160	Irrigated	Pasture	Upland	Tame
39	40	Dryland	Pasture	Upland	Native
40	80	Dryland	Pasture	Upland	Native
41	160	Dryland	Pasture	Upland	Native
42	160	Dryland	Pasture	Upland	Native
43	160	Irrigated	Crop	-	-
44	120	Dryland	Crop	-	-
45	160	Irrigated	Crop	-	-
46	160	Irrigated	Crop	-	-
47	80	Dryland	Crop	-	-
48	80	Irrigated	Crop	-	-
49	120	Irrigated	Crop	-	-
50	80	Dryland	Crop	-	-
51	80	Irrigated	Crop	-	-
52	160	Irrigated	Crop	-	-
53	160	Irrigated	Crop	-	-
54	80	Dryland	Pasture	Upland	Native
55	40	Irrigated	Crop	-	-
56	120	Irrigated	Crop	-	-
57	80	Dryland	Crop	-	-
58	40	Irrigated	Crop	-	-
59	80	Irrigated	Crop	Upland	Tame
60	40	Irrigated	Pasture	Upland	Tame
61	120	Dryland	Pasture	-	-
62	40	Irrigated	Crop		
63	80	Irrigated	Crop	-	-
64	40	Irrigated	Crop	-	-
65	40	Irrigated	Pasture	Upland	Tame

Table A.2. Biophysical Characteristics of 65 Fields in the Lower Little Bow Watershed.

Number	**Soil Texture** [a]	**Textural Category**	**LSS** [b]	**Soil Type** [c]
1	Sandy Loam	Coarse	4MT	ODBC
2	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
3	Loam	Medium	5M	ODBC w/ Regosolic Profiles
4	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
5	Loam	Medium	4MT	ODBC
6	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
7	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
8	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
9	Loamy Sand	Coarse	5M	ODBC w/ Regosolic Profiles
10	Sandy Loam	Coarse	4MT	ODBC
11	Sandy Loam	Coarse	4MT	ODBC
12	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
13	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles

14	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
15	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
16	Sandy Loam	Coarse	5M	ODBC w/ Regosolic Profiles
17	Sandy Loam	Coarse	4MT	ODBC
18	Sandy Loam	Coarse	4MT	ODBC
19	Loamy Sand	Coarse	5MT	ODBC w/ Regosolic Profiles
20	Sandy Loam	Coarse	5MT	ODBC w/ Regosolic Profiles
21	Sandy Loam	Coarse	4MT	ODBC
22	Sandy Loam	Coarse	5MT	ODBC w/ Regosolic Profiles
23	Sandy Loam	Coarse	5MT	ODBC w/ Regosolic Profiles
24	Sandy Loam	Coarse	4M	Orthic Regosol
25	Loamy Sand	Coarse	5MT	ODBC w/ Regosolic Profiles
26	Sandy Loam	Coarse	4M (5) - 5MTP(5)	ODBC
27	Loam	Medium	4M (5) - 5MTP(5)	ODBC
28	Loam	Medium	5TM	Orthic Regosol
29	Sandy Loam	Coarse	4M (5) - 5MTP(5)	ODBC
30	Sandy Loam	Coarse	4M (5) - 5MTP(5)	ODBC
31	Loam	Medium	4M	Orthic Regosol
32	Sandy Loam	Coarse	5MT	Orthic Regosol
33	Sandy Loam	Coarse	5MT	ODBC w/ Regosolic Profiles
34	Sandy Loam	Coarse	5TM	Orthic Regosol
35	Loam	Medium	4M	Orthic Regosol
36	Loam	Medium	4M	Orthic Regosol
37	Loam	Medium	4M	ODBC
38	Sandy Loam	Coarse	4M	ODBC
39	Sandy Loam	Coarse	4MT	ODBC
40	Loam	Medium	5TM	Orthic Regosol
41	Loam	Medium	5TM	Orthic Regosol
42	Loam	Medium	4M	ODBC
43	Loam	Medium	4MT	ODBC
44	Loam	Medium	4M	ODBC
45	Sandy Loam	Coarse	4M	ODBC
46	Loam	Medium	4M	ODBC
47	Loam	Medium	4MT	ODBC
48	Loam	Medium	4MT	ODBC
49	Loam	Medium	4MT	ODBC
50	Sandy Clay Loam	Medium	4MT	ODBC
51	Sandy Loam	Coarse	4M	ODBC
52	Loam	Medium	4M	ODBC
53	Loam	Medium	4MT	ODBC
54	Sandy Loam	Coarse	4MT	ODBC
55	Loam	Medium	4MT	ODBC

56	Sandy Loam	Coarse	4MT	ODBC
57	Loam	Medium	4MT	ODBC
58	Sandy Loam	Coarse	4MT	ODBC
59	Loam	Medium	4M	Orthic Regosol
60	Sandy Loam	Coarse	4M	ODBC
61	Loam	Medium	5TM	Orthic Regosol
62	Loam	Medium	5TM	Orthic Regosol
63	Sandy Loam	Coarse	4M	ODBC
64	Loam	Medium	4MT	ODBC
65	Sandy Clay Loam	Medium	4MT	ODBC

[a] Determined based on the results of soil testing done in July of 2006 as part of the WEBs Lower Little Bow project.
[b] Land Suitability Rating System rating; Fields with a 'T' (slope) limitation are classified as marginal for purposes of analysis in Scenario 4c.
[c] ODBC = Orthic Dark Brown Chernozem.

Appendix B: Crop Rotations and Field Allocation Strategy

Table B.1. Crop Acronyms Used in Appendix B.

Acronym	Crop
SW	Spring Wheat
DW	Durum Wheat
C	Canola
B	Barley
SF	Summerfallow
DB	Dry Beans
P	Potato
SB	Sugar Beet
F	Fababean Green Manure
RC	Red Clover Green Manure
FP	Field Peas
AH	Alfalfa Hay
AGM	Alfalfa / Grass Mix
T	Timothy Hay

B.1. Baseline Scenarios and Scenario 3

The 39 fields assigned to annual cropping are each assigned to a group representing a starting point in their respective rotation (Dryland, Irrigated 1, Irrigated 2). In the Baseline and BAU Baseline Scenarios, each of the base crop rotations is four years in length. Hence, fields are sorted into four different groups. Tables B.2-B.4 display the crop each group of fields is planted to in each of the 20 years of the modeling timeframe for the three base rotations (Dryland, Irrigated 1, Irrigated 2). Table B.5 displays the acreage and field-specific group assignment for each rotation. Table B.6 summarizes the number of fields and total acreage assignment to each group.

Table B.2. Baseline Dryland Crop Rotation.

	Group			
Year	**1**	**2**	**3**	**4**
1	SW	C	B	SF
2	C	B	SF	SW
3	B	SF	SW	C
4	SF	SW	C	B
5	SW	C	B	SF
6	C	B	SF	SW
7	B	SF	SW	C
8	SF	SW	C	B
9	SW	C	B	SF
10	C	B	SF	SW
11	B	SF	SW	C
12	SF	SW	C	B
13	SW	C	B	SF
14	C	B	SF	SW
15	B	SF	SW	C
16	SF	SW	C	B
17	SW	C	B	SF
18	C	B	SF	SW
19	B	SF	SW	C
20	SF	SW	C	B

Table B.3. Baseline Irrigated 1 Crop Rotation.

	Group			
Year	**1**	**2**	**3**	**4**
1	SW	C	DW	DB
2	C	DW	DB	SW
3	DW	DB	SW	C
4	DB	SW	C	DW
5	SW	C	DW	DB
6	C	DW	DB	SW
7	DW	DB	SW	C
8	DB	SW	C	DW
9	SW	C	DW	DB
10	C	DW	DB	SW
11	DW	DB	SW	C
12	DB	SW	C	DW

13	SW	C	DW	DB
14	C	DW	DB	SW
15	DW	DB	SW	C
16	DB	SW	C	DW
17	SW	C	DW	DB
18	C	DW	DB	SW
19	DW	DB	SW	C
20	DB	SW	C	DW

Table B.4. Baseline Irrigated 2 Crop Rotation.

	Group			
Year	**1**	**2**	**3**	**4**
1	P	SW	SB	B
2	SW	SB	B	P
3	SB	B	P	SW
4	B	P	SW	SB
5	P	SW	SB	B
6	SW	SB	B	P
7	SB	B	P	SW
8	B	P	SW	SB
9	P	SW	SB	B
10	SW	SB	B	P
11	SB	B	P	SW
12	B	P	SW	SB
13	P	SW	SB	B
14	SW	SB	B	P
15	SB	B	P	SW
16	B	P	SW	SB
17	P	SW	SB	B
18	SW	SB	B	P
19	SB	B	P	SW
20	B	P	SW	SB

Table B.5. Field Specific Assignment to Rotation Groups, Baseline Scenario.[a]

Field Number	Acres	Dryland Group	Irrigated 1 Group	Irrigated 2 Group
1	80	1	-	-
2	120	2	-	-
3	40	3	-	-

4	40	4	-	-
5	40	1	-	-
6	40	2	-	-
7	120	3	-	-
8	160	4	-	-
9	120	1	-	-
10	40	2	-	-
12	120	-	-	2
13	160	-	-	3
14	120	-	-	4
17	120	-	-	1
18	160	-	-	1
19	160	-	-	2
22	160	3	-	-
24	120	-	-	3
27	160	-	1	-
31	80	-	2	-
44	160	-	3	-
45	120	4	-	-
46	160	-	-	4
47	160	-	4	-
48	80	1	-	-
49	80	-	1	-
50	120	-	2	-
51	80	2	-	-
52	80	-	-	1
53	160	-	3	-
54	160	-	4	-
56	40	-	1	-
57	120	-	-	2
58	80	3	-	-
59	40	-	-	3
60	80	-	2	-
63	40	-	3	-
64	80	-	-	4
65	40	-	4	-

[a] Includes only fields assigned to cropping activities (both dryland and irrigated).

Table B.6. Summary of Field Assignment and Acreage to Each Group, Baseline Scenarios and Scenario 3.

Rotation	**Group**	**Number of Fields**	**Acres**
Dryland	Group 1	4	320
	Group 2	4	280
	Group 3	4	400

	Group 4	3	320
	Total	*15*	*1320*
Irrigated 1	Group 1	3	280
	Group 2	3	280
	Group 3	3	360
	Group 4	3	360
	Total	*12*	*1280*
Irrigated 2	Group 1	3	360
	Group 2	3	400
	Group 3	3	320
	Group 4	3	360
	Total	*12*	*1440*
Total (All Cropland)		**39**	**4040**

B.2. Scenarios 5a and 5c

The identical base crop rotations are used on pasture fields in Scenarios 5a and 5c. Table B.7 details the allocation of these fields to the groupings used for the original cropped fields.

Table B.7. Field Specific Assignment to Rotation Groups, Scenarios 5a and 5c.[a]

Field Number	Acres	Dryland Group	Irrigated 1 Group	Irrigated 2 Group
11	80	2	-	-
15	120	1	-	-
16	40	2	-	-
20	120	3	-	-
21	40	4	-	-
23	160	1	-	-
25	80	2	-	-
26	160	1	-	-
28	80	3	-	-
29	120	4	-	-
30	160	1	-	-
32	120	2	-	-
33	40	3	-	-
34	80	-	-	1

35	160	3	-	-
36	160	4	-	-
37	40	-	1	-
38	160	-	-	2
39	40	4	-	-
40	80	1	-	-
41	160	2	-	-
42	160	1	-	-
54	80	3	-	-
60	40	-	-	3
61	120	4	-	-
65	40	-	2	-

[a] Includes only fields originally assigned to pasture (per Table A.1) and converted to cropland.

B.2. Scenarios 1a, 1b, and 5b

Alfalfa is introduced to annual crop rotations in Scenarios 1a, 1b, and 5b. The following tables outline the field allocation strategy in these scenarios. Cropped fields are assigned to one of seven groups in these scenarios, reflecting the longer rotation length; each group of fields begins the modeling timeframe planted to a different starting crop (or year of perennial crop stand).

Table B.8. Alfalfa BMP Dryland Rotation, Scenarios 1a, 1b, and 5b.

	Group						
Year	**1**	**2**	**3**	**4**	**5**	**6**	**7**
1	AGM	AGM	AGM	SW	C	B	SF
2	AGM	AGM	SW	C	B	SF	AGM
3	AGM	SW	C	B	SF	AGM	AGM
4	SW	C	B	SF	AGM	AGM	AGM
5	C	B	SF	AGM	AGM	AGM	SW
6	B	SF	AGM	AGM	AGM	SW	C
7	SF	AGM	AGM	AGM	SW	C	B
8	AGM	AGM	AGM	SW	C	B	SF
9	AGM	AGM	SW	C	B	SF	AGM
10	AGM	SW	C	B	SF	AGM	AGM
11	SW	C	B	SF	AGM	AGM	AGM
12	C	B	SF	AGM	AGM	AGM	SW
13	B	SF	AGM	AGM	AGM	SW	C

14	SF	AGM	AGM	AGM	SW	C	B
15	AGM	AGM	AGM	SW	C	B	SF
16	AGM	AGM	SW	C	B	SF	AGM
17	AGM	SW	C	B	SF	AGM	AGM
18	SW	C	B	SF	AGM	AGM	AGM
19	C	B	SF	AGM	AGM	AGM	SW
20	B	SF	AGM	AGM	AGM	SW	C

Table B.9. Alfalfa BMP Irrigated 1 Rotation, Scenarios 1a and 5b.

	Group						
Year	**1**	**2**	**3**	**4**	**5**	**6**	**7**
1	SW	C	DW	DB	AH	AH	AH
2	C	DW	DB	AH	AH	AH	SW
3	DW	DB	AH	AH	AH	SW	C
4	DB	AH	AH	AH	SW	C	DW
5	AH	AH	AH	SW	C	DW	DB
6	AH	AH	SW	C	DW	DB	AH
7	AH	SW	C	DW	DB	AH	AH
8	SW	C	DW	DB	AH	AH	AH
9	C	DW	DB	AH	AH	AH	SW
10	DW	DB	AH	AH	AH	SW	C
11	DB	AH	AH	AH	SW	C	DW
12	AH	AH	AH	SW	C	DW	DB
13	AH	AH	SW	C	DW	DB	AH
14	AH	SW	C	DW	DB	AH	AH
15	SW	C	DW	DB	AH	AH	AH
16	C	DW	DB	AH	AH	AH	SW
17	DW	DB	AH	AH	AH	SW	C
18	DB	AH	AH	AH	SW	C	DW
19	AH	AH	AH	SW	C	DW	DB
20	AH	AH	SW	C	DW	DB	AH

Table B.10. Alfalfa BMP Irrigated 2 Rotation, Scenarios 1a and 5b.

	Group						
Year	**1**	**2**	**3**	**4**	**5**	**6**	**7**
1	P	SW	SB	DB	AH	AH	AH
2	SW	SB	DB	AH	AH	AH	DW
3	SB	DB	AH	AH	AH	DW	P
4	DB	AH	AH	AH	DW	P	SW

5	AH	AH	AH	DW	P	SW	SB
6	AH	AH	DW	P	SW	SB	DB
7	AH	DW	P	SW	SB	DB	AH
8	DW	P	SW	SB	DB	AH	AH
9	P	SW	SB	DB	AH	AH	AH
10	SW	SB	DB	AH	AH	AH	DW
11	SB	DB	AH	AH	AH	DW	P
12	DB	AH	AH	AH	DW	P	SW
13	AH	AH	AH	DW	P	SW	SB
14	AH	AH	DW	P	SW	SB	DB
15	AH	DW	P	SW	SB	DB	AH
16	DW	P	SW	SB	DB	AH	AH
17	P	SW	SB	DB	AH	AH	AH
18	SW	SB	DB	AH	AH	AH	DW
19	SB	DB	AH	AH	AH	DW	P
20	DB	AH	AH	AH	DW	P	SW

Table B.11. Field Specific Assignment to Rotation Groups, Scenarios 1a, 1b, and 5b.[a]

Field Number	**Acres**	**BMP Dryland Group**	**BMP Irrigated 1 Group**	**BMP Irrigated 2 Group**
1	80	1	-	-
2	120	2	-	-
3	40	3	-	-
4	40	4	-	-
5	40	5	-	-
6	40	6	-	-
7	120	7	-	-
8	160	1	-	-
9	120	2	-	-
10	40	3	-	-
12	120	-	-	2
13	160	-	-	3
14	120	-	-	4
17	120	-	-	1
18	160	-	-	5
19	160	-	-	6
22	160	4	-	-
24	120	-	-	7
27	160	-	1	-
31	80	-	2	-
44	160	-	3	-
45	120	5	-	-
46	160	-	-	1
47	160	-	4	-

48	80	6	-	-
49	80	-	5	-
50	120	-	6	-
51	80	7	-	-
52	80	-	-	2
53	160	-	7	-
54	160	-	1	-
56	40	-	2	-
57	120	-	-	3
58	80	1	-	-
59	40	-	-	4
60	80	-	3	-
63	40	-	4	-
64	80	-	-	5
65	40	-	5	-

[a] Includes only fields assigned to cropping activities (both dryland and irrigated).

Table B.12. Summary of Field Assignment and Acreage to Each Group, Scenarios 1a, 1b, and 5b.

Rotation	**Group**	**Number of Fields**	**Acres**
BMP Dryland	Group 1	3	320
	Group 2	2	240
	Group 3	2	80
	Group 4	2	200
	Group 5	2	160
	Group 6	2	120
	Group 7	2	200
	Total	*15*	*1320*
BMP Irrigated 1	Group 1	2	320
	Group 2	2	120
	Group 3	2	240
	Group 4	2	200
	Group 5	2	120
	Group 6	1	120
	Group 7	1	160
	Total	*12*	*1280*
BMP Irrigated 2	Group 1	2	280
	Group 2	2	200
	Group 3	2	280
	Group 4	2	160
	Group 5	2	240

Group 6	1	160
Group 7	1	120
Total	*12*	*1440*
Total (All Cropland)	**39**	**4040**

B.3. Scenarios 2a, 2b, 2c

Table B.13. Legume Green Manure (Fababean) BMP Irrigated 1 Rotation, Scenarios 2a, 2b, and 2c.

	Group				
Year	**1**	**2**	**3**	**4**	**5**
1	SW	F	C	DW	DB
2	F	C	DW	DB	SW
3	C	DW	DB	SW	F
4	DW	DB	SW	F	C
5	DB	SW	F	C	DW
6	SW	F	C	DW	DB
7	F	C	DW	DB	SW
8	C	DW	DB	SW	F
9	DW	DB	SW	F	C
10	DB	SW	F	C	DW
11	SW	F	C	DW	DB
12	F	C	DW	DB	SW
13	C	DW	DB	SW	F
14	DW	DB	SW	F	C
15	DB	SW	F	C	DW
16	SW	F	C	DW	DB
17	F	C	DW	DB	SW
18	C	DW	DB	SW	F
19	DW	DB	SW	F	C
20	DB	SW	F	C	DW

Table B.14. Legume Green Manure (Fababean) BMP Irrigated 2 Rotation, Scenarios 2a, 2b, and 2c.

	Group				
Year	**1**	**2**	**3**	**4**	**5**
1	P	SW	SB	DB	F
2	SW	SB	DB	F	P

3	SB	DB	F	P	SW
4	DB	F	P	SW	SB
5	F	P	SW	SB	DB
6	P	SW	SB	DB	F
7	SW	SB	DB	F	P
8	SB	DB	F	P	SW
9	DB	F	P	SW	SB
10	F	P	SW	SB	DB
11	P	SW	SB	DB	F
12	SW	SB	DB	F	P
13	SB	DB	F	P	SW
14	DB	F	P	SW	SB
15	F	P	SW	SB	DB
16	P	SW	SB	DB	F
17	SW	SB	DB	F	P
18	SB	DB	F	P	SW
19	DB	F	P	SW	SB
20	F	P	SW	SB	DB

Table B.15. Field Pea BMP Dryland Rotation, Scenario 2a.

			Group			
Year	**1**	**2**	**3**	**4**	**5**	**6**
1	SW	SF	SW	FP	B	C
2	C	SW	SF	SW	FP	B
3	B	C	SW	SF	SW	FP
4	FP	B	C	SW	SF	SW
5	SW	FP	B	C	SW	SF
6	SF	SW	FP	B	C	SW
7	SW	SF	SW	FP	B	C
8	C	SW	SF	SW	FP	B
9	B	C	SW	SF	SW	FP
10	FP	B	C	SW	SF	SW
11	SW	FP	B	C	SW	SF
12	SF	SW	FP	B	C	SW
13	SW	SF	SW	FP	B	C
14	C	SW	SF	SW	FP	B
15	B	C	SW	SF	SW	FP
16	FP	B	C	SW	SF	SW
17	SW	FP	B	C	SW	SF
18	SF	SW	FP	B	C	SW
19	SW	SF	SW	FP	B	C
20	C	SW	SF	SW	FP	B

Table B.16. Legume Green Manure (Red Clover) BMP Dryland Rotation, Scenario 2b.

	Group			
Year	**1**	**2**	**3**	**4**
1	SW	C	B	LGM
2	C	B	LGM	SW
3	B	LGM	SW	C
4	LGM	SW	C	B
5	SW	C	B	LGM
6	C	B	LGM	SW
7	B	LGM	SW	C
8	LGM	SW	C	B
9	SW	C	B	LGM
10	C	B	LGM	SW
11	B	LGM	SW	C
12	LGM	SW	C	B
13	SW	C	B	LGM
14	C	B	LGM	SW
15	B	LGM	SW	C
16	LGM	SW	C	B
17	SW	C	B	LGM
18	C	B	LGM	SW
19	B	LGM	SW	C
20	LGM	SW	C	B

Table B.17. Field Pea / Legume Green Manure BMP Dryland Rotation, Scenario 2c.

	Group							
Year	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
1	SW	C	B	LGM	SW	FP	SW	C
2	C	B	LGM	SW	FP	SW	C	SW
3	B	LGM	SW	FP	SW	C	SW	C
4	LGM	SW	FP	SW	C	SW	C	B
5	SW	FP	SW	C	SW	C	B	LGM
6	FP	SW	C	SW	C	B	LGM	SW
7	SW	C	SW	C	B	LGM	SW	FP
8	C	SW	C	B	LGM	SW	FP	SW
9	SW	C	B	LGM	SW	FP	SW	C
10	C	B	LGM	SW	FP	SW	C	SW

11	B	LGM	SW	FP	SW	C	SW	C
12	LGM	SW	FP	SW	C	SW	C	B
13	SW	FP	SW	C	SW	C	B	LGM
14	FP	SW	C	SW	C	B	LGM	SW
15	SW	C	SW	C	B	LGM	SW	FP
16	C	SW	C	B	LGM	SW	FP	SW
17	SW	C	B	LGM	SW	FP	SW	C
18	C	B	LGM	SW	FP	SW	C	SW
19	B	LGM	SW	FP	SW	C	SW	C
20	LGM	SW	FP	SW	C	SW	C	B

Table B.18. Field Specific Assignment to Rotation Groups, Scenarios 2a, 2b, and 2c.[a]

Field Number	**Acres**	**Scen.2a Dryland Group**	**Scen.2b Dryland Group**	**Scen.2c Dryland Group**	**Irrigated 1 Group**[b]	**Irrigated 2 Group**[b]
1	80	1	1	1	0	0
2	120	2	2	2	0	0
3	40	3	3	3	0	0
4	40	4	4	4	0	0
5	40	5	1	5	0	0
6	40	6	2	6	0	0
7	120	1	3	7	0	0
8	160	2	4	8	0	0
9	120	3	1	1	0	0
10	40	4	2	2	0	0
12	120	0	0	0	0	2
13	160	0	0	0	0	3
14	120	0	0	0	0	4
17	120	0	0	0	0	1
18	160	0	0	0	0	5
19	160	0	0	0	0	1
22	160	5	3	3	0	0
24	120	0	0	0	0	2
27	160	0	0	0	1	0
31	80	0	0	0	2	0
44	160	0	0	0	3	0
45	120	6	4	4	0	0
46	160	0	0	0	0	3
47	160	0	0	0	4	0
48	80	1	1	5	0	0
49	80	0	0	0	5	0
50	120	0	0	0	1	0
51	80	2	2	6	0	0

52	80	0	0	0	0	4
53	160	0	0	0	2	0
54	160	0	0	0	3	0
56	40	0	0	0	4	0
57	120	0	0	0	0	5
58	80	3	3	7	0	0
59	40	0	0	0	0	1
60	80	0	0	0	5	0
63	40	0	0	0	1	0
64	80	0	0	0	0	2
65	40	0	0	0	2	0

[a] Includes only fields assigned to cropping activities (both dryland and irrigated).
[b] Scenarios 2a, 2b, and 2c share the same two irrigated BMP rotations.

Appendix C: Yield Trend Regression Results

Table C.1. Yield Trend Regression Results.[a]

Crop	Irrigated/Dryland	Intercept Coefficient	Year Coefficient	T-Stat	p-value
Barley	Dryland	659.9	20.02	3.75	0.001
Canola	Dryland	359.37	10.1	3.27	0.002
Field Peas	Dryland	-169.6	41.12	2.53	0.024
Durum Wheat	Dryland	539.3	19.35	4.33	0.000
Spring Wheat	Dryland	505.3	17.81	4.65	0.000
Barley	Irrigated	1538.8	18.6	3.82	0.001
Canola	Irrigated	625.2	15.2	6.45	0.000
Dry Beans	Irrigated	1081	-4.1	-0.41	0.689
Potatoes	Irrigated	503.65	53.73	2.88	0.016
Sugar Beets	Irrigated	1365.8	274.5	2.21	0.040
Durum Wheat	Irrigated	1001.9	35.08	7.17	0.000
Spring Wheat	Irrigated	866.08	32.2	7.54	0.000

[a] Based on the regression $Y_t = \alpha + \beta t + \varepsilon_t$, where Y is crop yield and t is time in years.

Appendix D: Detailed Scenario Results

Table D.1. BAU Baseline Scenario Results.

	Mean Net Cashflow	**Watershed N Balance** *(kg)*	**Average N Balance** *(kg acre^{-1})*	**Cumulative N Balance** *(kg)*	**Watershed P Balance** *(kg)*	**Average P Balance** *(kg acre^{-1})*	**Cumulative P Balance** *(kg)*	**Net SOC Change** *(tonnes)*
Year 1	$1,552,437	175,243	26.23	175,243	40,639	6.08	40,639	n/a
Year 2	$1,616,286	104,281	15.61	279,524	21,584	3.23	62,223	n/a
Year 3	$1,519,046	117,618	17.61	397,142	25,840	3.87	88,063	n/a
Year 4	$1,705,874	102,546	15.35	499,688	20,619	3.09	108,682	n/a
Year 5	$1,552,437	175,243	26.23	674,931	38,095	5.70	146,777	n/a
Year 6	$1,616,286	104,281	15.61	779,212	20,776	3.11	167,553	n/a
Year 7	$1,519,046	117,618	17.61	896,830	24,578	3.68	192,131	n/a
Year 8	$1,705,874	102,546	15.35	999,376	20,618	3.09	212,749	n/a
Year 9	$1,552,437	175,243	26.23	1,174,619	38,096	5.70	250,845	n/a
Year 10	$1,616,286	104,281	15.61	1,278,900	20,776	3.11	271,620	n/a
Year 11	$1,519,046	117,618	17.61	1,396,518	24,577	3.68	296,197	n/a
Year 12	$1,705,874	102,546	15.35	1,499,064	20,618	3.09	316,816	n/a
Year 13	$1,552,437	175,243	26.23	1,674,307	38,096	5.70	354,912	n/a
Year 14	$1,616,286	104,281	15.61	1,778,588	20,776	3.11	375,687	n/a
Year 15	$1,519,046	117,618	17.61	1,896,206	24,577	3.68	400,264	n/a
Year 16	$1,705,874	102,546	15.35	1,998,752	20,618	3.09	420,883	n/a
Year 17	$1,552,437	175,243	26.23	2,173,994	38,096	5.70	458,978	n/a
Year 18	$1,616,286	104,281	15.61	2,278,276	20,776	3.11	479,754	n/a
Year 19	$1,519,046	117,618	17.61	2,395,894	24,577	3.68	504,331	n/a
Year 20	$1,705,874	102,546	15.35	2,498,440	20,618	3.09	524,949	n/a

Table D.2. Baseline Scenario Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$1,554,103	66,038	9.89	66,038	13,661	2.05	13,661	n/a
Year 2	$1,574,422	66,636	9.98	132,674	14,325	2.14	27,985	n/a
Year 3	$1,491,392	68,612	10.27	201,285	16,327	2.44	44,312	n/a
Year 4	$1,658,448	68,810	10.30	270,095	13,804	2.07	58,116	n/a
Year 5	$1,554,103	66,038	9.89	336,133	13,661	2.05	71,777	n/a
Year 6	$1,574,422	66,636	9.98	402,769	14,325	2.14	86,101	n/a
Year 7	$1,491,392	68,612	10.27	471,381	13,486	2.02	99,587	n/a
Year 8	$1,658,448	68,810	10.30	540,190	13,804	2.07	113,391	n/a
Year 9	$1,554,103	66,038	9.89	606,228	13,661	2.05	127,051	n/a
Year 10	$1,574,422	66,636	9.98	672,864	14,325	2.14	141,376	n/a
Year 11	$1,491,392	68,612	10.27	741,476	13,486	2.02	154,862	n/a
Year 12	$1,658,448	68,810	10.30	810,286	13,804	2.07	168,665	n/a
Year 13	$1,554,103	66,038	9.89	876,323	13,661	2.05	182,326	n/a
Year 14	$1,574,422	66,636	9.98	942,959	14,325	2.14	196,651	n/a
Year 15	$1,491,392	68,612	10.27	1,011,571	13,486	2.02	210,136	n/a
Year 16	$1,658,448	68,810	10.30	1,080,381	13,804	2.07	223,940	n/a
Year 17	$1,554,103	66,038	9.89	1,146,418	13,661	2.05	237,601	n/a
Year 18	$1,574,422	66,636	9.98	1,213,054	14,325	2.14	251,925	n/a
Year 19	$1,491,392	68,612	10.27	1,281,666	13,486	2.02	265,411	n/a
Year 20	$1,658,448	68,810	10.30	1,350,476	13,804	2.07	279,215	n/a

Table D.3. Scenario 1a Results.

	Mean Net Cashflow	**Watershed N Balance** *(kg)*	**Average N Balance** *(kg acre^{-1})*	**Cumulative N Balance** *(kg)*	**Watershed P Balance** *(kg)*	**Average P Balance** *(kg acre^{-1})*	**Cumulative P Balance** *(kg)*	**Net SOC Change** *(tonnes)*
Year 1	$1,255,801	75521	11.31	75,521	11,950	1.79	11,950	463
Year 2	$719,419	54912	8.22	130,433	8,115	1.21	20,065	449
Year 3	$1,194,138	29262	4.38	159,695	7,430	1.11	27,495	442
Year 4	$1,320,832	-6077	-0.91	153,618	4,109	0.62	31,604	435
Year 5	$1,595,354	6736	1.01	160,355	5,512	0.83	37,117	429
Year 6	$1,418,186	-11473	-1.72	148,882	3,330	0.50	40,446	422
Year 7	$1,633,815	12777	1.91	161,659	7,720	1.16	48,166	416
Year 8	$1,584,432	-15909	-2.38	145,750	7,289	1.09	55,455	410
Year 9	$1,755,791	39612	5.93	185,362	8,199	1.23	63,654	404
Year 10	$871,701	20745	3.11	206,107	4,757	0.71	68,411	398
Year 11	$1,243,683	24308	3.64	230,415	7,556	1.13	75,967	392
Year 12	$1,317,157	4584	0.69	234,999	3,826	0.57	79,793	387
Year 13	$1,593,068	21774	3.26	256,773	4,335	0.65	84,129	381
Year 14	$1,295,462	-2624	-0.39	254,150	3,119	0.47	87,247	376
Year 15	$1,766,206	6057	0.91	260,206	9,055	1.36	96,302	370
Year 16	$1,568,143	9798	1.47	270,004	6,630	0.99	102,932	365
Year 17	$1,813,796	37438	5.60	307,442	8,339	1.25	111,272	360
Year 18	$864,650	3558	0.53	311,000	6,341	0.95	117,613	355
Year 19	$1,243,385	32090	4.80	343,090	6,795	1.02	124,408	350
Year 20	$1,332,655	20277	3.04	363,367	2,489	0.37	126,897	345

Table D.4. Scenario 1b Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$1,511,441	70,273	10.52	70,273	15,398	2.31	15,398	160
Year 2	$1,736,207	71,053	10.64	141,327	16,261	2.43	31,659	155
Year 3	$1,568,418	66,805	10.00	208,131	17,653	2.64	49,312	153
Year 4	$1,714,126	58,678	8.78	266,809	14,441	2.16	63,753	150
Year 5	$1,804,621	58,953	8.83	325,763	14,457	2.16	78,210	148
Year 6	$1,845,616	59,249	8.87	385,011	14,657	2.19	92,866	146
Year 7	$1,560,132	62,101	9.30	447,112	14,094	2.11	106,960	143
Year 8	$1,669,668	63,872	9.56	510,984	15,369	2.30	122,329	141
Year 9	$1,665,352	65,910	9.87	576,894	14,933	2.24	137,262	139
Year 10	$1,755,232	61,199	9.16	638,093	15,638	2.34	152,900	137
Year 11	$1,622,107	60,446	9.05	698,538	13,593	2.03	166,493	135
Year 12	$1,833,124	60,720	9.09	759,258	14,046	2.10	180,539	133
Year 13	$1,740,544	61,508	9.21	820,766	14,237	2.13	194,775	131
Year 14	$1,733,641	59,638	8.93	880,404	15,350	2.30	210,125	129
Year 15	$1,576,475	63,615	9.52	944,019	14,700	2.20	224,825	127
Year 16	$1,686,917	65,654	9.83	1,009,673	15,306	2.29	240,131	125
Year 17	$1,650,354	62,075	9.29	1,071,748	14,589	2.18	254,721	124
Year 18	$1,790,610	58,718	8.79	1,130,466	14,795	2.21	269,516	122
Year 19	$1,738,718	60,855	9.11	1,191,321	13,599	2.04	283,115	120
Year 20	$1,768,042	62,519	9.36	1,253,840	14,392	2.15	297,506	118

Table D.5. Scenario 2a Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$1,218,896	105,494	15.79	105,494	15,518	2.32	15,518	18
Year 2	$1,375,334	89,992	13.47	195,486	17,747	2.66	33,265	17
Year 3	$1,209,430	86,059	12.88	281,545	18,605	2.79	51,871	17
Year 4	$1,451,676	87,061	13.03	368,606	18,381	2.75	70,252	17
Year 5	$1,377,836	92,677	13.87	461,282	20,216	3.03	90,468	16
Year 6	$1,452,191	85,241	12.76	546,523	18,177	2.72	108,645	16
Year 7	$1,390,304	82,735	12.39	629,258	17,369	2.60	126,013	16
Year 8	$1,229,919	81,203	12.16	710,461	17,158	2.57	143,171	16
Year 9	$1,465,011	83,427	12.49	793,888	18,141	2.72	161,312	15
Year 10	$1,389,807	92,263	13.81	886,151	18,544	2.78	179,856	15
Year 11	$1,469,582	81,076	12.14	967,226	18,184	2.72	198,040	15
Year 12	$1,369,194	76,164	11.40	1,043,390	17,093	2.56	215,133	15
Year 13	$1,184,211	73,760	11.04	1,117,150	16,611	2.49	231,744	14
Year 14	$1,464,419	80,809	12.10	1,197,959	17,834	2.67	249,578	14
Year 15	$1,407,839	89,133	13.34	1,287,092	18,557	2.78	268,135	14
Year 16	$1,482,381	81,405	12.19	1,368,498	18,095	2.71	286,230	14
Year 17	$1,403,434	74,120	11.10	1,442,618	17,952	2.69	304,182	14
Year 18	$1,199,679	70,556	10.56	1,513,174	16,509	2.47	320,691	14
Year 19	$1,437,222	79,433	11.89	1,592,606	17,703	2.65	338,394	13
Year 20	$1,376,216	89,915	13.46	1,682,521	18,651	2.79	357,045	13

Table D.6. Scenarios 2b Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$1,227,307	117,279	17.56	117,279	18,235	2.73	18,235	53
Year 2	$1,337,937	104,348	15.62	221,627	20,876	3.13	39,111	51
Year 3	$1,187,679	97,949	14.66	319,575	20,866	3.12	59,978	50
Year 4	$1,434,082	104,283	15.61	423,859	20,774	3.11	80,752	50
Year 5	$1,346,107	110,757	16.58	534,615	22,386	3.35	103,137	49
Year 6	$1,423,023	108,283	16.21	642,898	20,622	3.09	123,760	48
Year 7	$1,370,179	100,592	15.06	743,490	19,390	2.90	143,150	47
Year 8	$1,193,833	96,919	14.51	840,409	19,093	2.86	162,243	47
Year 9	$1,441,832	98,713	14.78	939,122	20,765	3.11	183,008	46
Year 10	$1,342,793	107,601	16.11	1,046,722	22,097	3.31	205,105	45
Year 11	$1,450,696	99,219	14.85	1,145,942	20,197	3.02	225,303	45
Year 12	$1,375,187	93,642	14.02	1,239,584	19,693	2.95	244,995	44
Year 13	$1,196,427	87,791	13.14	1,327,374	19,487	2.92	264,483	43
Year 14	$1,427,582	94,711	14.18	1,422,085	21,152	3.17	285,635	43
Year 15	$1,367,638	99,967	14.97	1,522,052	21,039	3.15	306,673	42
Year 16	$1,455,257	92,982	13.92	1,615,034	20,123	3.01	326,797	42
Year 17	$1,375,179	90,458	13.54	1,705,492	19,949	2.99	346,746	41
Year 18	$1,180,922	92,432	13.84	1,797,924	20,434	3.06	367,180	41
Year 19	$1,454,102	89,358	13.38	1,887,282	20,257	3.03	387,437	40
Year 20	$1,368,563	99,236	14.86	1,986,518	21,180	3.17	408,617	39

Table D.7. Scenario 2c Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$1,246,785	111,555	16.70	111,555	17,196	2.57	17,196	53
Year 2	$1,374,491	96,787	14.49	208,341	19,462	2.91	36,657	51
Year 3	$1,204,839	90,378	13.53	298,720	20,020	3.00	56,678	50
Year 4	$1,453,148	94,731	14.18	393,450	19,999	2.99	76,676	50
Year 5	$1,386,366	100,712	15.08	494,162	21,290	3.19	97,966	49
Year 6	$1,450,582	98,534	14.75	592,696	19,291	2.89	117,258	48
Year 7	$1,387,145	92,114	13.79	684,810	18,671	2.80	135,929	47
Year 8	$1,228,516	86,553	12.96	771,363	17,851	2.67	153,780	47
Year 9	$1,465,659	89,092	13.34	860,455	19,458	2.91	173,239	46
Year 10	$1,381,154	101,875	15.25	962,330	20,544	3.08	193,783	45
Year 11	$1,469,939	92,408	13.83	1,054,739	19,087	2.86	212,869	45
Year 12	$1,384,636	87,233	13.06	1,141,972	19,169	2.87	232,039	44
Year 13	$1,224,610	80,734	12.09	1,222,706	18,170	2.72	250,209	43
Year 14	$1,467,159	84,490	12.65	1,307,196	19,580	2.93	269,789	43
Year 15	$1,398,221	88,887	13.31	1,396,084	20,308	3.04	290,097	42
Year 16	$1,494,918	85,283	12.77	1,481,367	18,920	2.83	309,016	42
Year 17	$1,399,020	82,976	12.42	1,564,342	18,398	2.75	327,414	41
Year 18	$1,218,138	85,488	12.80	1,649,830	18,451	2.76	345,866	41
Year 19	$1,473,909	83,576	12.51	1,733,406	19,329	2.89	365,195	40
Year 20	$1,389,143	94,186	14.10	1,827,593	20,485	3.07	385,680	39

Table D.8. Scenario 3 Results.

	Mean Net Cashflow	Watershed N Balance (kg)	Average N Balance (kg $acre^{-1}$)	Cumulative N Balance (kg)	Watershed P Balance (kg)	Average P Balance (kg $acre^{-1}$)	Cumulative P Balance (kg)	Net SOC Change (tonnes)
Year 1	$1,678,453	165,662	24.80	165,662	35,509	5.32	35,509	n/a
Year 2	$1,774,364	157,033	23.51	322,695	25,883	3.87	61,392	n/a
Year 3	$1,591,114	154,635	23.15	477,330	18,940	2.84	80,332	n/a
Year 4	$1,706,799	131,818	19.73	609,148	8,002	1.20	88,333	n/a
Year 5	$1,680,692	142,875	21.39	752,023	13,738	2.06	102,072	n/a
Year 6	$1,778,369	148,697	22.26	900,720	15,279	2.29	117,351	n/a
Year 7	$1,589,340	152,167	22.78	1,052,887	14,994	2.24	132,345	n/a
Year 8	$1,708,229	131,723	19.72	1,184,610	8,088	1.21	140,433	n/a
Year 9	$1,680,351	143,384	21.46	1,327,994	13,542	2.03	153,974	n/a
Year 10	$1,774,000	148,295	22.20	1,476,289	15,365	2.30	169,339	n/a
Year 11	$1,588,386	153,047	22.91	1,629,336	14,787	2.21	184,126	n/a
Year 12	$1,706,529	129,699	19.42	1,759,035	8,000	1.20	192,126	n/a
Year 13	$1,680,894	141,779	21.22	1,900,814	13,792	2.06	205,918	n/a
Year 14	$1,772,296	146,999	22.01	2,047,813	15,412	2.31	221,331	n/a
Year 15	$1,591,942	153,069	22.91	2,200,882	14,742	2.21	236,073	n/a
Year 16	$1,704,853	129,296	19.36	2,330,178	8,071	1.21	244,144	n/a
Year 17	$1,679,288	142,248	21.29	2,472,426	13,771	2.06	257,914	n/a
Year 18	$1,775,376	148,398	22.22	2,620,825	15,544	2.33	273,458	n/a
Year 19	$1,590,579	152,812	22.88	2,773,637	14,972	2.24	288,430	n/a
Year 20	$1,705,244	130,586	19.55	2,904,223	8,152	1.22	296,581	n/a

Table D.9. Scenario 4a Results.

	Mean Net Cashflow	**Watershed N Balance** *(kg)*	**Average N Balance** *(kg acre^{-1})*	**Cumulative N Balance** *(kg)*	**Watershed P Balance** *(kg)*	**Average P Balance** *(kg acre^{-1})*	**Cumulative P Balance** *(kg)*	**Net SOC Change** *(tonnes)*
Year 1	$749,208	66191	9.91	66,191	6,151	0.92	6,151	1081
Year 2	$684,221	101176	15.15	167,367	1,746	0.26	7,897	1048
Year 3	$633,395	43325	6.49	210,692	(278)	-0.04	7,619	1032
Year 4	$556,997	41818	6.26	252,510	3,896	0.58	11,516	1016
Year 5	$383,377	66296	9.92	318,806	5,556	0.83	17,072	1001
Year 6	$375,360	42833	6.41	361,639	5,264	0.79	22,336	986
Year 7	$270,363	-3087	-0.46	358,552	6,793	1.02	29,129	971
Year 8	$290,649	-25656	-3.84	332,896	5,095	0.76	34,224	957
Year 9	$281,863	-17497	-2.62	315,399	4,379	0.66	38,603	943
Year 10	$378,055	17642	2.64	333,041	231	0.03	38,834	929
Year 11	$408,165	35983	5.39	369,024	689	0.10	39,524	915
Year 12	$386,799	62607	9.37	431,631	2,286	0.34	41,809	902
Year 13	$412,823	72373	10.83	504,004	2,889	0.43	44,698	889
Year 14	$354,592	68763	10.29	572,767	4,875	0.73	49,573	876
Year 15	$309,936	39292	5.88	612,059	7,158	1.07	56,731	831
Year 16	$328,389	33496	5.01	645,555	5,758	0.86	62,488	805
Year 17	$311,120	-5706	-0.85	639,849	5,775	0.86	68,263	780
Year 18	$369,778	-6253	-0.94	633,596	3,391	0.51	71,654	763
Year 19	$347,923	-21229	-3.18	612,368	3,262	0.49	74,916	755
Year 20	$400,117	3827	0.57	616,195	695	0.10	75,611	751

Table D.10. Scenario 4b Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$1,469,365	65,985	9.88	65,985	14,664	2.20	14,664	373
Year 2	$1,566,834	72,369	10.83	138,354	14,705	2.20	29,369	362
Year 3	$1,340,806	71,961	10.77	210,315	17,034	2.55	46,404	356
Year 4	$1,411,994	71,797	10.75	282,112	14,850	2.22	61,253	351
Year 5	$1,356,332	68,093	10.19	350,205	14,708	2.20	75,961	345
Year 6	$1,435,725	68,600	10.27	418,805	16,118	2.41	92,079	340
Year 7	$1,243,871	69,562	10.41	488,367	15,612	2.34	107,691	335
Year 8	$1,352,297	70,286	10.52	558,652	16,554	2.48	124,245	330
Year 9	$1,344,345	68,045	10.19	626,697	15,825	2.37	140,070	325
Year 10	$1,501,871	71,580	10.72	698,277	15,654	2.34	155,724	320
Year 11	$1,297,993	71,871	10.76	770,148	14,332	2.15	170,056	315
Year 12	$1,345,848	70,556	10.56	840,704	14,952	2.24	185,008	311
Year 13	$1,302,210	66,860	10.01	907,564	15,332	2.30	200,339	306
Year 14	$1,442,174	67,504	10.11	975,068	16,673	2.50	217,012	301
Year 15	$1,286,006	70,184	10.51	1,045,252	16,036	2.40	233,048	297
Year 16	$1,411,994	69,807	10.45	1,115,058	16,069	2.41	249,117	293
Year 17	$1,356,332	69,573	10.42	1,184,632	14,868	2.23	263,984	289
Year 18	$1,435,725	68,899	10.31	1,253,531	15,392	2.30	279,377	284
Year 19	$1,243,871	70,347	10.53	1,323,878	14,434	2.16	293,811	280
Year 20	$1,352,297	69,482	10.40	1,393,360	15,575	2.33	309,386	276

Table D.11. Scenario 4c Results.

	Mean Net Cashflow	**Watershed N Balance** *(kg)*	**Average N Balance** *(kg acre^{-1})*	**Cumulative N Balance** *(kg)*	**Watershed P Balance** *(kg)*	**Average P Balance** *(kg acre^{-1})*	**Cumulative P Balance** *(kg)*	**Net SOC Change** *(tonnes)*
Year 1	$1,558,161	65,999	9.88	65,999	13,805	2.07	13,805	165
Year 2	$1,649,214	68,295	10.22	134,294	14,717	2.20	28,522	160
Year 3	$1,428,766	70,917	10.62	205,211	16,536	2.48	45,058	158
Year 4	$1,553,200	69,149	10.35	274,360	14,256	2.13	59,314	155
Year 5	$1,511,382	66,440	9.95	340,800	13,765	2.06	73,080	153
Year 6	$1,591,919	66,546	9.96	407,347	15,457	2.31	88,537	151
Year 7	$1,394,822	68,794	10.30	476,140	14,383	2.15	102,920	148
Year 8	$1,529,305	69,739	10.44	545,880	15,089	2.26	118,009	146
Year 9	$1,529,435	66,266	9.92	612,145	14,001	2.10	132,010	144
Year 10	$1,632,009	68,375	10.24	680,520	14,929	2.23	146,939	142
Year 11	$1,410,713	70,125	10.50	750,645	13,582	2.03	160,521	140
Year 12	$1,473,726	69,099	10.34	819,743	14,386	2.15	174,907	138
Year 13	$1,592,395	66,557	9.96	886,300	14,277	2.14	189,184	136
Year 14	$1,419,632	65,755	9.84	952,055	15,617	2.34	204,801	134
Year 15	$1,559,728	70,202	10.51	1,022,256	14,415	2.16	219,216	132
Year 16	$1,513,816	69,097	10.34	1,091,353	14,621	2.19	233,837	130
Year 17	$1,608,286	67,176	10.06	1,158,529	13,748	2.06	247,585	128
Year 18	$1,403,438	67,339	10.08	1,225,868	14,816	2.22	262,402	126
Year 19	$1,525,784	69,685	10.43	1,295,553	13,712	2.05	276,113	125
Year 20	$1,529,305	68,738	10.29	1,364,290	14,898	2.23	291,011	123

Table D.12. Scenario 5a Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$2,129,240	78,232	11.71	78,232	16,360	2.45	16,360	-741
Year 2	$2,317,303	80,970	12.12	159,202	17,070	2.56	33,430	-718
Year 3	$1,928,867	81,250	12.16	240,452	19,279	2.89	52,709	-707
Year 4	$1,934,796	81,166	12.15	321,618	16,473	2.47	69,182	-696
Year 5	$2,129,240	78,232	11.71	399,851	16,360	2.45	85,542	-685
Year 6	$2,317,303	80,970	12.12	480,820	17,070	2.56	102,612	-675
Year 7	$1,928,867	81,250	12.16	562,071	16,438	2.46	119,050	-665
Year 8	$1,934,796	81,166	12.15	643,236	16,473	2.47	135,523	-655
Year 9	$2,129,240	78,232	11.71	721,469	16,360	2.45	151,883	-645
Year 10	$2,317,303	80,970	12.12	802,439	17,070	2.56	168,953	-635
Year 11	$1,950,090	81,250	12.16	883,689	16,438	2.46	185,392	-626
Year 12	$2,025,872	81,166	12.15	964,855	16,473	2.47	201,865	-617
Year 13	$2,133,769	78,232	11.71	1,043,087	16,360	2.45	218,224	-608
Year 14	$2,234,812	80,970	12.12	1,124,057	17,070	2.56	235,295	-599
Year 15	$1,950,090	81,250	12.16	1,205,307	16,438	2.46	251,733	-590
Year 16	$2,025,872	81,166	12.15	1,286,473	16,473	2.47	268,206	-582
Year 17	$2,133,769	78,232	11.71	1,364,705	16,360	2.45	284,566	-573
Year 18	$2,234,812	80,970	12.12	1,445,675	17,070	2.56	301,636	-565
Year 19	$1,950,090	81,250	12.16	1,526,925	16,438	2.46	318,074	-557
Year 20	$2,025,872	81,166	12.15	1,608,091	16,473	2.47	334,547	-550

Table D.13. Scenario 5b Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$1,664,505	87060	13.03	87,060	18,117	2.71	18,117	40
Year 2	$1,042,435	57020	8.54	144,080	14,417	2.16	32,533	39
Year 3	$1,505,242	27026	4.05	171,106	12,523	1.87	45,056	38
Year 4	$1,685,796	9719	1.45	180,825	9,762	1.46	54,817	38
Year 5	$1,915,886	-14497	-2.17	166,329	8,351	1.25	63,169	37
Year 6	$1,703,259	-9540	-1.43	156,789	5,932	0.89	69,101	37
Year 7	$2,137,245	-43368	-6.49	113,421	8,863	1.33	77,964	36
Year 8	$2,423,502	-24884	-3.73	88,537	10,447	1.56	88,411	36
Year 9	$2,365,662	8198	1.23	96,734	12,712	1.90	101,122	35
Year 10	$1,245,874	20310	3.04	117,044	8,532	1.28	109,654	35
Year 11	$1,625,572	44143	6.61	161,187	11,984	1.79	121,639	35
Year 12	$1,667,955	3071	0.46	164,258	6,006	0.90	127,645	34
Year 13	$1,905,385	-20665	-3.09	143,593	6,325	0.95	133,969	34
Year 14	$1,614,550	-23756	-3.56	119,838	6,653	1.00	140,622	33
Year 15	$2,198,426	2896	0.43	122,734	11,387	1.70	152,009	33
Year 16	$2,422,534	-10766	-1.61	111,968	12,611	1.89	164,620	32
Year 17	$2,377,403	15421	2.31	127,389	12,665	1.90	177,285	32
Year 18	$1,282,980	20782	3.11	148,171	10,030	1.50	187,315	32
Year 19	$1,611,161	6106	0.91	154,277	8,699	1.30	196,014	31
Year 20	$1,651,649	-11240	-1.68	143,037	4,837	0.72	200,851	31

Table D.14. Scenario 5c Results.

	Mean Net Cashflow	Watershed N Balance *(kg)*	Average N Balance *(kg acre^{-1})*	Cumulative N Balance *(kg)*	Watershed P Balance *(kg)*	Average P Balance *(kg acre^{-1})*	Cumulative P Balance *(kg)*	Net SOC Change *(tonnes)*
Year 1	$2,132,103	187,436	28.06	187,436	43,257	6.48	43,257	-741
Year 2	$2,273,250	118,614	17.76	306,050	24,141	3.61	67,398	-718
Year 3	$1,986,026	130,255	19.50	436,306	28,793	4.31	96,190	-707
Year 4	$2,074,163	114,901	17.20	551,207	23,024	3.45	119,214	-696
Year 5	$2,132,103	187,436	28.06	738,642	40,713	6.09	159,927	-685
Year 6	$2,273,250	118,614	17.76	857,257	23,333	3.49	183,260	-675
Year 7	$1,986,026	130,255	19.50	987,512	27,530	4.12	210,790	-665
Year 8	$2,074,163	114,901	17.20	1,102,413	23,023	3.45	233,813	-655
Year 9	$2,132,103	187,436	28.06	1,289,849	40,714	6.09	274,526	-645
Year 10	$2,273,250	118,614	17.76	1,408,463	23,333	3.49	297,859	-635
Year 11	$1,986,026	130,255	19.50	1,538,719	27,529	4.12	325,389	-626
Year 12	$2,074,163	114,901	17.20	1,653,620	23,023	3.45	348,412	-617
Year 13	$2,132,103	187,436	28.06	1,841,056	40,714	6.09	389,125	-608
Year 14	$2,273,250	118,614	17.76	1,959,670	23,333	3.49	412,458	-599
Year 15	$1,986,026	130,255	19.50	2,089,925	27,529	4.12	439,987	-590
Year 16	$2,074,163	114,901	17.20	2,204,826	23,023	3.45	463,010	-582
Year 17	$2,132,103	187,436	28.06	2,392,262	40,714	6.09	503,724	-573
Year 18	$2,273,250	118,614	17.76	2,510,876	23,333	3.49	527,057	-565
Year 19	$1,986,026	130,255	19.50	2,641,132	27,529	4.12	554,586	-557
Year 20	$2,074,163	114,901	17.20	2,756,033	23,023	3.45	577,609	-550

Appendix E: Results of Sensitivity Analysis

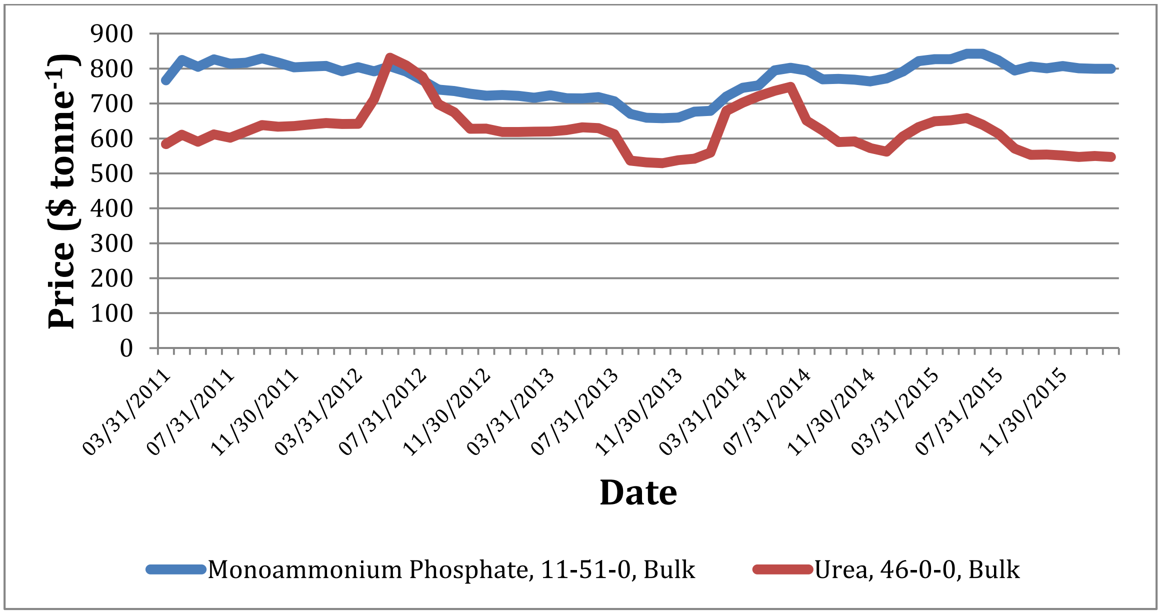
E.1. Fertilizer Price Sensitivity Results

Fertilizer prices are an important part of this analysis not only because nutrient costs are a significant share of crop production costs, but because each of the scenarios modeled involves tradeoffs between obtaining nutrients from different sources. There are different costs associated with the various nutrient sources, whether that be manure, biological fixation, or chemical fertilizer. For instance, N from biological fixation may be obtained from the atmosphere for free, but a significant opportunity cost exists when a field is seeded to a legume crop instead of a higher-value non-legume crop. Conversely, manure and chemical fertilizer each involve a price that the crop producer must pay. In this analysis, the difference between the economic returns generated in each scenario will be impacted by the relative tradeoffs between these nutrient sources. For example, if the price of urea (a common N fertilizer) is high, it may be more cost-effective to obtain N from planting legume crops, which enhances the attractiveness of the alfalfa BMP. However, if the price is low, a producer may choose to purchase the fertilizer and forgo the planting of legume crops.

Figure G.1 displays the average monthly price over the last five years in Alberta of two commonly used chemical fertilizers: urea and monoammonium phosphate. The price of urea is used to estimate to cost of N on a per kg basis in this analysis, and the price of monoammonium phosphate is used to estimate the cost of P (see section 5.5.4). As Figure E.1 shows, prices can be highly variable depending both on time of year and other external factors. These factors include production costs, market demand, and competition, as well as currency exchange rates and government policies (AAFC, 2013). For instance, the cost of N fertilizer production has been historically well-correlated with that of fossil fuel prices (particularly natural gas) because the production process is energy-intensive (AAFC, 2013). Additionally, because fertilizers are internationally traded commodities, their prices are influenced by global supply and demand factors.

Higher crop prices in recent years has led to increased fertilizer usage among crop producers, which has in turn led to increased fertilizer prices (AAFC, 2013).

Figure E.1. Average Monthly Farm Fertilizer Prices in Alberta, 2011-2015.



Source: AAF (2016).

Prices in April of 2015 were used in the initial evaluation of the baseline and BMP scenarios. The bulk prices of urea and monoammonium phosphate were $652 and $801 per tonne, respectively, which works out to $1.41 per kg of N and $1.27 per kg of P. For the sensitivity analysis, prices of the two fertilizers at the same time of year each of the past three years were used: April 2014, April 2013, and April 2012. In April 2014 the price of urea was significantly higher ($721 per tonne), but the price of monoammonium phosphate lower ($751). This offset in prices between the two somewhat obscures an evaluation of a change in price of either individually, although total N inputs are substantially higher and make up a much larger share of total nutrient costs. In April 2013, the price of both urea and monoammonium phosphate is lower than in April 2015, allowing for a consistent evaluation of changes to the tradeoffs in obtaining nutrients from the various sources.

April 2012 is also a useful price point for the sensitivity analysis, as the price of urea peaked ($792 per tonne, $142 more than in April 2015) while the price of monoammonium phosphate was relatively the same as in April 2015. It should be noted that the corresponding selection of price for monoammonium phosphate was March 2012 instead of April in order to hold P prices as constant as possible relative to April 2015. Additionally, the prices at a non-seeding period of the season, October 2015, were used as well. In this case, the price of urea was significantly lower ($554 per tonne) and the price of monoammonium phosphate nearly identical to that of April 2015. These last two price points are beneficial to the sensitivity analysis because they both allow for the isolation of N input price impacts. Table G.1 details the price of both urea and monoammonium phosphate at each of the dates listed, and the corresponding breakdown of the price of a kg of N and P.

Table E.1. Fertilizer Prices at Select Dates For Sensitivity Analysis.

Date	**Price of Urea** ($ tonne^{1})	**Price of N** ($ kg^{-1})	**Price of MP** [a] ($ tonne^{-1})	**Price of P** ($ kg^{-1})
Apr-15	650	1.41	801	1.27
Apr-14	721	1.57	751	1.13
Apr-13	624	1.36	715	1.11
Apr-12	792	1.72	809[b]	1.21
Oct-15	554	1.20	801	1.31

Source: AAF (2016).
[a] Monoammonium phosphate, bulk.
[b] Price in March, 2012.

The mean NPV of agricultural activity in each land use scenario, including both baseline scenarios, is recalculated using each of the four alternate fertilizer price points. Table 6.29 lists the annualized NPV on a per acre basis for each scenario under each of the different price points. The percent difference of each BMP scenarios from the Baseline Scenario (or the BAU Baseline Scenario in the case of Scenario 5c) is calculated. Then, the difference between each BMP result relative to that of the April 2015 price point (the original) is calculated and also presented in Table E.2.

Broadly speaking, several conclusions can be drawn. First, none of the changes to fertilizer prices impacted the overall direction of mean NPV change due to the implementation of a BMP. In other words, whether a gain or loss in NPV was found initially, this overall result was insensitive to fertilizer price changes. A scenario involving a decrease in NPV, such as Scenario 1a, remains a wealth-decreasing land use change regardless of fertilizer price.

Second, as expected, changes to the price of fertilizer (particularly urea), did impact the magnitude of NPV changes relative to baseline scenarios. As the cost of obtaining N and P from chemical fertilizer increases, so does the attractiveness of alternative nutrient sources. Conversely, when fertilizer is less expensive, the benefits of using other sources are diminished. For instance, urea is cheapest in October, 2015. Using this N price ($1.20 per kg), the inclusion of alfalfa in annual crop rotations (Scenario 1a) incurs the highest loss relative to the baseline (16.33%). However, when urea was most expensive (April, 2012), Scenario 1a becomes a more attractive option vis-à-vis the Baseline Scenario, posting a decrease in NPV of only 13.18%. This corresponds to an annualized loss of only $32.59 per ace, compared to $38.84 under April 2015 prices. If compared to October 2015, when price of urea is at its lowest, the decrease in annualized NPV per acre is highest at $43.32. The difference between the two extremes (when comparing Scenario 1a and the Baseline) is $10.73 an acre annually.

Another scenario in which nutrient source tradeoffs are highlighted by changing fertilizer prices is Scenario 3. Similar to Scenario 1a (and a less extent Scenario 1b and the three versions of Scenario 2) with biological fixation, manure becomes a more attractive nutrient source as the price of chemical fertilizers increase. Conversely, the benefits of manure application are reduced when chemical fertilizer becomes a less expensive option.

Lastly, the magnitude of the difference in NPV changes is generally small. In general, the main findings of this analysis are no sensitive to the choice of fertilizer price levels.

Table E.2. Fertilizer Price Sensitivity Analysis Results ($ $acre^{-1}$ $year^{-1}$).

	Apr-15	Apr-14	Apr-13	Apr-12	Oct-15
Baseline	**$258.04**	**$253.31**	**$261.06**	**$247.26**	**$265.26**
Scenario 1a	**$219.20**	$217.91	$221.14	$214.67	$221.94
% diff. relative to Baseline	***-15.05%***	*-13.98%*	*-15.29%*	*-13.18%*	*-16.33%*
diff. from Apr-15 result		*1.08%*	*-0.24%*	*1.88%*	*-1.28%*
Scenario 1b	**$277.32**	$273.31	$280.09	$267.97	$283.55
% diff. relative to Baseline	***7.47%***	*7.90%*	*7.29%*	*8.38%*	*6.89%*
diff. from Apr-15 result		*0.43%*	*-0.18%*	*0.91%*	*-0.58%*
Scenario 2a	**$223.43**	$219.82	$225.76	$214.72	$228.71
% diff. relative to Baseline	***-13.41%***	*-13.22%*	*-13.52%*	*-13.16%*	*-13.78%*
diff. from Apr-15 result		*0.19%*	*-0.11%*	*0.26%*	*-0.37%*
Scenario 2b	**$219.56**	$216.14	$222.31	$210.86	$225.41
% diff. relative to Baseline	***-14.91%***	*-14.67%*	*-14.84%*	*-14.72%*	*-15.02%*
diff. from Apr-15 result		*0.24%*	*0.07%*	*0.19%*	*-0.11%*
Scenario 2c	**$224.38**	$220.49	$226.89	$215.04	$230.10
% diff. relative to Baseline	***-13.04%***	*-12.96%*	*-13.09%*	*-13.03%*	*-13.25%*
diff. from Apr-15 result		*0.09%*	*-0.04%*	*0.02%*	*-0.21%*
Scenario 3	**$278.16**	$274.58	$279.59	$270.78	$282.92
% diff. relative to Baseline	***7.80%***	*8.39%*	*7.10%*	*9.51%*	*6.66%*
diff. from Apr-15 result		*0.60%*	*-0.70%*	*1.72%*	*-1.14%*
Scenario 4a	**$76.64**	$74.86	$78.66	$71.68	$79.94
% diff. relative to Baseline	***-70.30%***	*-70.45%*	*-69.87%*	*-71.01%*	*-69.86%*
diff. from Apr-15 result		*-0.15%*	*0.43%*	*-0.71%*	*0.43%*
Scenario 4b	**$229.40**	$225.34	$232.36	$219.75	$235.85
% diff. relative to Baseline	***-11.10%***	*-11.04%*	*-10.99%*	*-11.13%*	*-11.09%*
diff. from Apr-15 result		*0.05%*	*0.11%*	*-0.03%*	*0.01%*
Scenario 4c	**$251.59**	$247.06	$254.56	$241.17	$258.56
% diff. relative to Baseline	***-2.50%***	*-2.47%*	*-2.49%*	*-2.46%*	*-2.53%*
diff. from Apr-15 result		*0.03%*	*0.01%*	*0.04%*	*-0.03%*
Scenario 5a	**$344.71**	$338.26	$348.85	$329.97	$354.57
% diff. relative to Baseline	***33.59%***	*33.53%*	*33.63%*	*33.45%*	*33.67%*
diff. from Apr-15 result		*-0.05%*	*0.04%*	*-0.13%*	*0.08%*
Scenario 5b	**$287.33**	$285.54	$289.84	$281.35	$290.89
% diff. relative to Baseline	***11.35%***	*12.72%*	*11.02%*	*13.79%*	*9.66%*
diff. from Apr-15 result		*1.37%*	*-0.32%*	*2.44%*	*-1.69%*
BAU Baseline	**$262.52**	$259.67	$264.37	$255.99	$266.89
Scenario 5c	**$349.40**	$344.83	$352.37	$338.91	$356.41
% diff. relative to BAU	***33.09%***	*31.35%*	*34.22%*	*29.10%*	*35.76%*
diff. from Apr-15 result		*-1.74%*	*1.13%*	*-3.99%*	*2.67%*

E.2. Crop Price Sensitivity Results

The sensitivity of final results to choice of average crop price was also investigated. Initially, the average of crop prices between the years 2004-2013 were chosen as representative for the analysis, corresponding to the last ten years of available data. However, the rise of crop commodity prices in more recent years lends credence to the possibility that a representative price should include only the past five years. Also, changes in relative value of the different crops over time may impact NPV results due to the change in crop acreage in various scenarios. For example, the five-year average barley price is lower than the ten-year average. This trend is opposite for most other crops. Table E.3 summarizes the five, ten, and twenty-year average crop prices for the annual crops modeled in this analysis (adjusted for inflation). Mean NPV of each of the land use scenarios is recalculated using the two variants in crop prices (the five and twenty year averages) and compared to the original results (using the ten year average) in Table E.4.

Table E.3. Summary of 5-, 10-, and 20-Year Average Crop Price Data ($ $tonne^{-1}$) (2014 Canadian Dollars).

Crop	5-Year [a] Average	10-Year [b] Average	20-Year [c] Average
Spring Wheat	$272.49	$254.63	$248.45
Durum Wheat	$263.65	$260.25	$266.95
Barley	$209.77	$216.34	$189.91
Canola	$474.89	$436.21	$442.38
Field Peas	$277.02	$246.87	$239.26
Dry Beans	$828.94	$732.63	$750.66
Potato	$259.38	$243.35	$247.76
Sugar Beets	$53.61	$52.19	$53.40
Alfalfa	$87.49	$85.07	$100.89

[a] Average of 2009-2013;
[b] Average of 2004-2013;
[c] Average of 1994-2013.

Table E.4. Crop Price Sensitivity Analysis Results ($ acre^{-1} year^{-1}).

	10-Year	5-Year	20-Year
Baseline	**$258.04**	**$296.91**	**$265.19**
Scenario 1a	**$219.20**	$247.59	$237.11
% diff. relative to Baseline	***-15.05%***	*-16.61%*	*-10.59%*
diff. from 10-Year result		*-1.56%*	*4.47%*
Scenario 1b	**$277.32**	$315.13	$285.48
% diff. relative to Baseline	***7.47%***	*6.14%*	*7.65%*
diff. from 10-Year result		*-1.33%*	*0.18%*
Scenario 2a	**$223.43**	$254.87	$229.74
% diff. relative to Baseline	***-13.41%***	*-14.16%*	*-13.37%*
diff. from 10-Year result		*-0.75%*	*0.04%*
Scenario 2b	**$219.56**	$252.12	$225.65
% diff. relative to Baseline	***-14.91%***	*-15.09%*	*-14.91%*
diff. from 10-Year result		*-0.17%*	*0.00%*
Scenario 2c	**$224.38**	$255.39	$231.32
% diff. relative to Baseline	***-13.04%***	*-13.99%*	*-12.77%*
diff. from 10-Year result		*-0.94%*	*0.27%*
Scenario 3	**$278.16**	$319.58	$285.44
% diff. relative to Baseline	***7.80%***	*7.64%*	*7.64%*
diff. from 10-Year result		*-0.16%*	*-0.16%*
Scenario 4a	**$76.64**	$82.11	$94.14
% diff. relative to Baseline	***-70.30%***	*-72.35%*	*-64.50%*
diff. from 10-Year result		*-2.05%*	*5.80%*
Scenario 4b	**$229.40**	$255.55	$241.11
% diff. relative to Baseline	***-11.10%***	*-13.93%*	*-9.08%*
diff. from 10-Year result		*-2.83%*	*2.02%*
Scenario 4c	**$251.59**	$287.98	$259.32
% diff. relative to Baseline	***-2.50%***	*-3.01%*	*-2.21%*
diff. from 10-Year result		*-0.51%*	*0.29%*
Scenario 5a	**$344.71**	$399.69	$352.23
% diff. relative to Baseline	***33.59%***	*34.62%*	*32.82%*
diff. from 10-Year result		*1.03%*	*-0.77%*
Scenario 5b	**$287.33**	$325.57	$306.12
% diff. relative to Baseline	***11.35%***	*9.65%*	*15.43%*
diff. from 10-Year result		*-1.70%*	*4.08%*
BAU Baseline	**$262.52**	$301.39	$269.67
Scenario 5c	**$349.40**	$404.38	$356.91
% diff. relative to BAU	***33.09%***	*34.17%*	*32.35%*
diff. from 10-Year result		*1.08%*	*-0.74%*

The overall results proved to be insensitive to the selection of representative crop prices. The direction of BMP scenario impacts in terms of NPV remained consistent between the use of five, ten, and twenty-year average crop prices. As expected, using the five-year averages increases the calculated net returns to producers, as crop prices have generally increased over the last five years. Because of this, the opportunity cost of certain BMP scenarios increases relative to the Baseline Scenario. For instance, although alfalfa (and other perennial crop) prices also increased, it was not enough to offset the increased opportunity cost from substituting away from higher-value crops such as potatoes and canola. Conversely, using twenty-year average crop prices lessened the opportunity costs associated with several of the BMP scenarios. In particular, the three versions of Scenario 4 all featured absolute increases in NPV compared to NPV calculated under ten-year average crop prices, as well as proportional increases relative to the newly calculated Baseline Scenario. However, none of the changes induced by the use of either the five or twenty-year average prices were of enough magnitude to change the overall interpretation of BMP scenario results.

E.3. Crop Yield Sensitivity Results

Lastly, the sensitivity of final results to crop yield was investigated by using the five-year and twenty-year averages to compare with the previously chosen ten-year average. Although the crop data was detrended, small variations in average yield still exist between the different timeframes. The effect of changing crop yields on both economic returns and nutrient balance is examined, as crop yield directly impacts the rate of N and P removal from the agro-environmental system. Table E.5 summarizes the five, ten, and twenty-year average crop yield for each annual crop. Variation in the yield of alfalfa or alfalfa/grass is not analyzed here due to lack of historical data.

Table E.5. Summary of Detrended 5-, 10-, and 20-Year Average Crop Yield Data (kg acre^{-1}).

	Crop	5-Year [a] Average	10-Year [b] Average	20-Year [c] Average
Dryland	Barley	1,548	1,511	1,391
	Canola	875	812	705
	Field Peas	1,379	1,327	1,311
	Durum Wheat	1,321	1,299	1,249
	Red Spring Wheat	1,266	1,235	1,161
Irrigated	Barley	1,961	2,118	2,256
	Canola	1,192	1,219	1,169
	Dry Beans	867	950	968
	Potatoes	16,103	16,772	16,282
	Sugar Beets	23,046	23,804	23,559
	Durum Wheat	2,263	2,293	2,272
	Red Spring Wheat	1,956	2,028	2,053

[a] Average of 2009-2013;
[b] Average of 2004-2013;
[c] Average of 1994-2013.

Table E.6 displays the results of the crop yield sensitivity analysis on NPV. Annualized NPV per acre is re-calculated for each BMP scenario under the different crop yield averages. To evaluate the changes to results, the corresponding percent change in annualized NPV relative to the Baseline Scenario for each of the five and twenty-year average yields is presented, along with the difference with respect to the ten-year average result. Like the crop and fertilizer price sensitivity results, no major changes to the effect on producer returns of the BMP scenarios were uncovered by the crop yield sensitivity analysis. In general, use of the five-year average crop yields decreased the absolute level of producer returns generated when compared to ten-year result, but increased returns relative to the newly calculated Baseline Scenario. For instance, the annualized NPV per acre decreased from $219 to $209 in Scenario 1a, yet the percent difference relative to the corresponding Baseline Scenario increased from -15% to -13%, an increase of 2%. These results were largely driven by decreases in average yield for irrigated crops from the ten-year average to the five-year average. This effect was mitigated somewhat when using the twenty-year averages, as the absolute decrease in annualized NPV relative to the ten-year result was not as large for most BMP scenarios.

Tables E.7 and E.8 show the results of the crop yield sensitivity analysis on the watershed balance of N and P, respectively. Again, the magnitude of change in the results was generally low, and the direction of impact on nutrient balance of each BMP scenario remained consistent with initial results (using ten-year averages). In each BMP scenario, and for both N and P, the absolute level of residual nutrients increased when either the five or twenty year averages were used. This was because irrigated crop yields are generally highest when averaged over the last ten years (2004-2013) instead of last five or twenty. Using the later two averages resulted in an absolute decrease in nutrient export from the LLB watershed. However, the relative change with respect to the Baseline Scenario differed between the two average yield alternatives. This is most clearly illustrated in Scenario 5a, where, in the case of the five-year average result, both N and P balance increased in absolute terms compared to the ten-year corresponding BMP scenario result, but decreased relative to the newly calculated Baseline Scenario. On the other hand, use of the twenty-year yield averages increased nutrient balances both in absolute (compared to the ten-year BMP scenario result) and relative (compared to the twenty-year Baseline Scenario result) terms.

Table E.6. Crop Yield Sensitivity Analysis Results, NPV ($ acre^{-1} year^{-1}).

	10-year	5-year	20-year
Baseline	**$258.04**	**$240.28**	**$247.10**
Scenario 1a	**$219.20**	$208.96	$212.01
% diff. relative to Baseline	***-15.05%***	*-13.03%*	*-14.20%*
diff. from 10-Year result		*2.02%*	*0.85%*
Scenario 1b	**$277.32**	$259.74	$266.13
% diff. relative to Baseline	***7.47%***	*8.10%*	*7.70%*
diff. from 10-Year result		*0.63%*	*0.23%*
Scenario 2a	**$223.43**	$208.50	$213.44
% diff. relative to Baseline	***-13.41%***	*-13.23%*	*-13.62%*
diff. from 10-Year result		*0.19%*	*-0.21%*
Scenario 2b	**$219.56**	$205.02	$209.10
% diff. relative to Baseline	***-14.91%***	*-14.67%*	*-15.38%*
diff. from 10-Year result		*0.24%*	*-0.47%*
Scenario 2c	**$224.38**	$209.84	$213.68
% diff. relative to Baseline	***-13.04%***	*-12.67%*	*-13.53%*
diff. from 10-Year result		*0.38%*	*-0.48%*
Scenario 3	**$278.16**	$259.76	$266.71
% diff. relative to Baseline	***7.80%***	*8.11%*	*7.93%*
diff. from 10-Year result		*0.31%*	*0.14%*
Scenario 4a	**$76.64**	$75.50	$75.31
% diff. relative to Baseline	***-70.30%***	*-68.58%*	*-69.52%*
diff. from 10-Year result		*1.72%*	*0.78%*
Scenario 4b	**$229.40**	$211.84	$220.48
% diff. relative to Baseline	***-11.10%***	*-11.84%*	*-10.77%*
diff. from 10-Year result		*-0.74%*	*0.33%*
Scenario 4c	**$251.59**	$233.03	$242.30
% diff. relative to Baseline	***-2.50%***	*-3.02%*	*-1.94%*
diff. from 10-Year result		*-0.52%*	*0.56%*
Scenario 5a	**$344.71**	$327.20	$323.85
% diff. relative to Baseline	***33.59%***	*36.18%*	*31.06%*
diff. from 10-Year result		*2.59%*	*-2.53%*
Scenario 5b	**$287.33**	$277.83	$273.83
% diff. relative to Baseline	***11.35%***	*15.63%*	*10.82%*
diff. from 10-Year result		*4.28%*	*-0.53%*
BAU Baseline	**$262.52**	$244.76	$251.58
Scenario 5c	**$349.40**	$331.89	$328.53
% diff. relative to BAU	***33.09%***	*35.60%*	*30.59%*
diff. from 10-Year result		*2.51%*	*-2.51%*

Table E.7. Crop Yield Sensitivity Analysis Results, Nitrogen Balance (kg acre^{-1}).

	10 Year	5 Year	20 Year
Baseline	**10.11**	**10.86**	**10.20**
Scenario 1a	**2.87**	3.09	3.18
% diff. relative to Baseline	***-71.61%***	*-71.51%*	*-68.87%*
diff. from 10-Year result		*0.10%*	*2.74%*
Scenario 1b	**9.34**	10.00	9.60
% diff. relative to Baseline	***-7.60%***	*-7.90%*	*-5.85%*
diff. from 10-Year result		*-0.30%*	*1.75%*
Scenario 2a	**12.63**	13.12	12.98
% diff. relative to Baseline	***24.95%***	*20.81%*	*27.24%*
diff. from 10-Year result		*-4.14%*	*2.29%*
Scenario 2b	**14.80**	15.26	15.22
% diff. relative to Baseline	***46.41%***	*40.56%*	*49.22%*
diff. from 10-Year result		*-5.85%*	*2.80%*
Scenario 2c	**13.64**	14.12	13.99
% diff. relative to Baseline	***34.94%***	*30.01%*	*37.20%*
diff. from 10-Year result		*-4.93%*	*2.26%*
Scenario 4a	**4.71**	4.72	4.77
% diff. relative to Baseline	***-53.41%***	*-56.51%*	*-53.24%*
diff. from 10-Year result		*-3.09%*	*0.17%*
Scenario 4b	**10.42**	11.20	10.44
% diff. relative to Baseline	***3.07%***	*3.13%*	*2.35%*
diff. from 10-Year result		*0.06%*	*-0.72%*
Scenario 4c	**10.22**	10.91	10.42
% diff. relative to Baseline	***1.08%***	*0.50%*	*2.13%*
diff. from 10-Year result		*-0.58%*	*1.05%*
Scenario 5a	**12.04**	12.46	13.03
% diff. relative to Baseline	***19.08%***	*14.75%*	*27.75%*
diff. from 10-Year result		*-4.33%*	*8.67%*
Scenario 5b	**1.40**	1.41	2.16
% diff. relative to Baseline	***-86.15%***	*-87.00%*	*-78.79%*
diff. from 10-Year result		*-0.85%*	*7.36%*
BAU Baseline	**18.70**	19.32	19.06
Scenario 3	**21.73**	22.34	22.12
% diff. relative to BAU	***16.20%***	*15.60%*	*16.03%*
diff. from 10-Year result		*-0.59%*	*-0.17%*
Scenario 5c	**20.63**	21.05	21.63
% diff. relative to BAU	***10.31%***	*8.97%*	*13.46%*
diff. from 10-Year result		*-1.34%*	*3.15%*

Table E.8. Crop Yield Sensitivity Analysis Results, Phosphorus Balance (kg acre^{-1}).

	10 Year	5 Year	20 Year
Baseline	**2.09**	**2.18**	**2.17**
Scenario 1a	**0.94**	0.99	1.00
% diff. relative to Baseline	***-55.02%***	*-54.53%*	*-53.82%*
diff. from 10-Year result		*0.49%*	*1.20%*
Scenario 1b	**2.22**	2.33	2.28
% diff. relative to Baseline	***6.22%***	*6.54%*	*4.97%*
diff. from 10-Year result		*0.32%*	*-1.26%*
Scenario 2a	**2.67**	2.74	2.74
% diff. relative to Baseline	***27.76%***	*25.55%*	*26.35%*
diff. from 10-Year result		*-2.20%*	*-1.40%*
Scenario 2b	**3.06**	3.13	3.15
% diff. relative to Baseline	***46.42%***	*43.29%*	*44.96%*
diff. from 10-Year result		*-3.13%*	*-1.46%*
Scenario 2c	**2.90**	2.97	2.99
% diff. relative to Baseline	***38.76%***	*35.86%*	*37.73%*
diff. from 10-Year result		*-2.90%*	*-1.03%*
Scenario 4a	**0.57**	0.57	0.58
% diff. relative to Baseline	***-72.92%***	*-73.76%*	*-73.32%*
diff. from 10-Year result		*-0.84%*	*-0.40%*
Scenario 4b	**2.32**	2.45	2.32
% diff. relative to Baseline	***10.81%***	*12.07%*	*7.05%*
diff. from 10-Year result		*1.26%*	*-3.76%*
Scenario 4c	**2.18**	2.29	2.23
% diff. relative to Baseline	***4.31%***	*4.80%*	*2.71%*
diff. from 10-Year result		*0.49%*	*-1.60%*
Scenario 5a	**2.50**	2.54	2.73
% diff. relative to Baseline	***19.82%***	*16.50%*	*25.81%*
diff. from 10-Year result		*-3.32%*	*5.99%*
Scenario 5b	**1.51**	1.53	1.67
% diff. relative to Baseline	***-27.75%***	*-30.11%*	*-22.82%*
diff. from 10-Year result		*-2.36%*	*4.93%*
BAU Baseline	**3.93**	4.02	4.01
Scenario 3	**2.21**	2.31	2.30
% diff. relative to BAU	***-43.77%***	*-42.51%*	*-42.62%*
diff. from 10-Year result		*1.26%*	*1.15%*
Scenario 5c	**4.32**	4.37	4.54
% diff. relative to BAU	***9.92%***	*8.59%*	*13.22%*
diff. from 10-Year result		*-1.33%*	*3.29%*