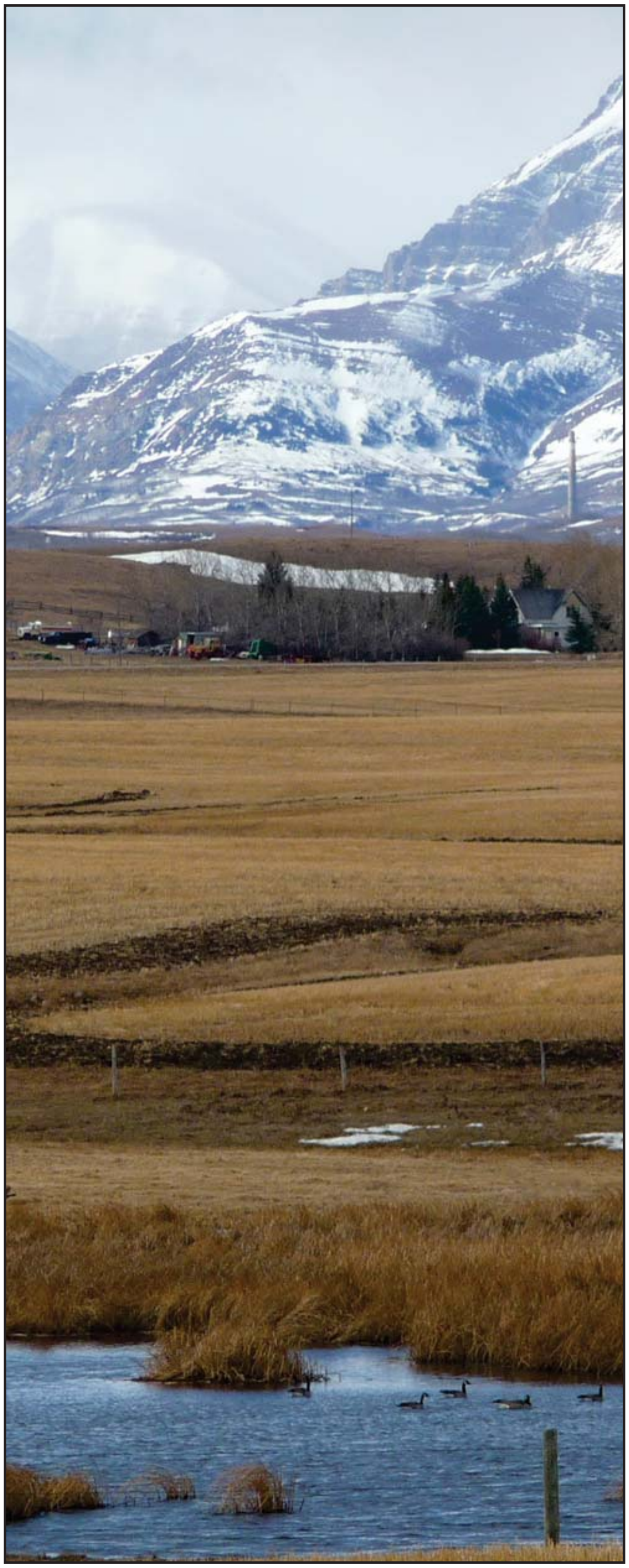
1 INTRODUCTION

1.1 Background



Complex relationships exist between agriculture and the environment. In recent years, environmental protection from agricultural activities has received much attention from scientists, academics, government policy makers, industry, the public, and agricultural producers. As such, the momentum towards sustainable and practical solutions has grown in many jurisdictions, including Alberta.

Next to the energy sector, the agri-food industry is an important driver of Alberta's economy. In 2011, agriculture in Alberta represented 21% of Canada's primary agricultural production with $10.5 billion in total farm cash receipts (Government of Alberta 2012). Approximately 31% of the total area of Alberta, 20.4 million ha, was used for crop and livestock production in 2011. In 2012, irrigated land was reported to be more than 625,000 ha (Government of Alberta 2013), or about 65% of total irrigated land in Canada (Statistics Canada 2012).

In Alberta, crop and livestock producers face challenges every day with increasing input costs, market competition, and continued pressure to improve environmental stewardship. Producers seek proven and practical beneficial management practices (BMPs) that will maintain efficient and viable farm operations while protecting the environment. The risks to the environment from agriculture are many, including risks to water quality. Agricultural practices have the potential to impair surface water quality and the surrounding environment. Inversely, environmental events and poor surface water quality can negatively impact agricultural production (Council of Canadian Academies 2013). In recent years, the impact of agriculture on the environment has focused on livestock production, and in particular the intensive livestock industry and manure management. Manure is recognized as a beneficial source of nutrients and as a soil conditioner that can be used to decrease input costs. However, if not managed properly, manure application can lead to excess accumulation of nutrients and introduction of bacteria into the soil, which can then enter ground or surface water (Smith et al. 2010).

Numerous BMPs have been developed and promoted to minimize the impacts of agriculture on the environment and increase the sustainability of the agricultural industry. Beneficial management practices are defined as conservation practices, management techniques, or social actions that minimize negative effects on the environment while being practical for producers to meet or exceed legal requirements and production targets (AFRD 2004; Sharpley et al. 2006). In this study, the evaluation of BMPs will focus mainly on water quality improvement.

The concept of soil and water conservation has been around since the late 1800s. Economic loss as a result of drought conditions and poor cropland practices of the infamous 1930s in Canada and the United States catalyzed governments to invest in and encourage conservation practices on private land. The push for scientific knowledge about soil and erosion prevention initiated the soil and water conservation movement (Johnson 1987). This paved the way for the advancement of technology and practices for what are today called BMPs.

Beneficial management practices are developed to protect water quality by managing nutrient inputs at the source and minimize or prevent nutrient losses. Examples include nutrient management plans; preventing wind and water soil erosion; irrigation management; timing of manure application to avoid saturated, snow-covered, and frozen soils; injection of liquid manure; incorporation of surface applied manure; the maintenance of healthy riparian and pasture areas; and livestock management. Nutrient management fulfills crop nutrient requirements and minimizes the potential for nutrients to become diffuse sources of contaminants (Oenema and Pietrzak 2002).

The effectiveness of BMPs under Alberta conditions is not well known. This is, in part, because many BMPs were developed in other parts of North America or at a research plot scale. Individual BMPs have rarely been evaluated under Alberta conditions (Wuite and Chanasyk 2003; AAFC 2007) and recent studies have recommended further research, especially with respect to phosphorus (P) management (Paterson et al. 2006). In addition, producers are requesting site-specific, risk-based analytical tools to assist them in deciding which management practices would yield the greatest impact for their financial investment. Needed is science-based evidence that these practices reduce risks to the environment and producers, gain economic and environmental advantages, and provide options for producers to meet or exceed regulatory requirements in Alberta.

Two key components that are needed to provide producers in Alberta with additional decision-making information are economic analysis and nutrient utilization. Producers and policy makers require information on the least-cost alternatives for reducing environmental impacts, and this requires an economic analysis of costs and benefits to the producer. Without information on the impact of management practices on nutrient utilization, prediction tools cannot provide the degree of accuracy producers need to make management decisions.

Environmental sustainability and BMPs within Alberta's agriculture industry are promoted and supported through several programs and organizations. For example, the Alberta Environmentally Sustainable Agriculture (AESA) soil quality project, established in 1997, responded to increased awareness of human activity on soil. It was a follow-up to recommendations made from the Canada-Alberta Environmentally Sustainable Agriculture (CAESA) soil quality project (Palliser Environmental Services Ltd. and Alberta Agriculture and Rural Development 2008). The AESA Project was aimed at determining the state of soil quality and its relation to environmental sustainability. As well, the Alberta Stewardship Network, established in 2004, supports the need of stewardship groups to conserve the environment. This includes funding opportunities, recruitment programs, and access to other resources (Land Stewardship Centre 2013). Additionally, Growing Forward (2008 to 2013) and Growing Forward 2 (2013 to 2018) are partnerships between Canada's federal and provincial governments to support the development of a competitive, yet environmentally aware, agriculture and agri-food sector (ARD 2013c). Other programs that support and promote BMP adoption include the Alberta Environmental Farm Plan Program, and in the past, the Canada-Alberta Farm Stewardship Program. These programs lend themselves as sources of information and support for sustainable agriculture.

It is largely assumed that BMPs elicit a positive effect on the environment. However, despite this and the aforementioned sources of information and support, there exist several barriers to BMP adoption. Producers may be reluctant to adopt BMPs due to time and monetary costs associated with implementation and ongoing maintenance and other perceived risks (Sharpley et al. 2006; Greiner et al. 2009). Further, BMPs are often not assessed or evaluated at larger scales (i.e., watersheds) because fewer factors can be controlled, replication is less feasible, and larger studies are expensive. There is also limited research showing cumulative effects of BMPs on the environment, specifically their impacts on the health of watersheds. The failure to assess or evaluate BMP effectiveness in a robust manner has contributed to a general lack of knowledge and understanding about BMPs. Within Alberta, additional BMP study sites are especially needed because of the diverse agro-climatic regions in the province. This BMP study evaluates the effectiveness of nutrient BMPs on water quality improvement in prime agricultural regions in Alberta in order to provide viable options for sustainable agriculture and water quality protection.

1.2 Surface Runoff

Climate, soils, field management, and landform all combine to predispose an area for surface runoff (Kleinman et al. 2006). Runoff is usually generated when rainfall intensity exceeds the infiltration capacity of the soil (Horton 1933, 1940) or when the water storage capacity of the soil is exceeded, resulting in a saturated soil condition (Hursh 1944; Dunne 1970). The latter can be promoted by subsurface features that cause a temporarily perched water table such as a fragipan, argillic horizon (Blanco-Canqui et al. 2002; Gburek et al. 2006), or a frozen ground layer during snowmelt (Hayashi et al. 2003). Infiltration rates vary depending on the total soil moisture (ice and water) conditions near the ground surface (Kane and Stein 1983). Henninger et al. (1976) found that high surface moisture in areas near a stream contributed more to surface runoff for a short time than areas distant from the stream channel and thus, these areas could become major sources of nutrient export to surface waters (Weld et al. 2001). These sources of nutrient export are called critical source areas (CSAs), which occur where a contaminant source coincides with hydrologic transport mechanisms (Gburek and Sharpley 1998; Meals et al. 2012). Critical source areas can also occur as a result of a combination of characteristics that makes an area vulnerable to nutrient loss such as soil type, land use, slope, and proximity to streams and other water bodies (White et al. 2009). This results in the potential for a small portion of the watershed to contribute the majority of exported material. Critical source areas typically vary with event. For example, a one-in-five year precipitation event may have a smaller CSA than a one-in-one hundred year precipitation event because hydrologic connectivity and subsequently transport potential are increased with greater precipitation. Gburek and Sharpley (1998) suggest that P export be managed by focusing on control of P levels in hydrologically active zones that are most likely to produce surface runoff. Areas with high runoff potential occur at different scales and exhibit similar export characteristics.

Agricultural watersheds are typically small in scale, occur in different ecoregions, and have been shown to produce surface runoff. Studies on several agricultural watersheds in Alberta have shown that the total yearly surface runoff from small agricultural watersheds is dominated by snowmelt (Gill et al. 1998; Wuite and Chanasyk 2003; Ontkean et al. 2005). A 3-yr study of eight, field-scale (92 to 248 ha) watersheds in Alberta demonstrated that spring runoff (primarily snowmelt) and summer runoff (primarily rainfall) varied geographically within the province (Little et al. 2006). In their study, Little et al. (2006) demonstrated that an average of 91% of the total runoff volume was from summer runoff (rainfall plus irrigation) at the Lower Little Bow site. Similarly, in one year, the Grande Prairie site had 71% of total runoff occur as summer runoff. In contrast, at three other sites in Alberta (Renwick Creek, Three Hills Creek, and Ponoka) 82% of the average total runoff volume was spring snowmelt runoff. In spite of some regional differences, Little et al. (2006) found that on average about 90% of the runoff came from spring snowmelt among the eight sites.

At different scales, concentrations of nutrients in the soil, such as P, can be linked with runoff potential, giving an indication of what concentrations may be in surface runoff. Jedrych et al. (2006) developed a method to calculate site-specific soil-test phosphorus (STP) limits for agricultural land in Alberta and tested the method on six watersheds and seven microwatersheds in Alberta. They found that STP variance was related to runoff potential among soil polygons within each watershed and microwatershed. Specifically, they found that minimum STP values related to soil polygons with high runoff potential, while soil polygons with low runoff potential had higher calculated STP values (Jedrych et al. 2006).

Stream flow and runoff volumes at the large scale vary across Canada depending on physiographic and climatic patterns (Cole 2013). At the larger scale, an appropriate measure of runoff is the annual unit runoff, which is a measure of runoff volume per square kilometre, and has been calculated for much of Canada in order to understand runoff patterns throughout the country (Cole 2013). These values have been used for purposes such as on-farm water storage structure design and determining water availability for project licensing. Because it gives an idea of runoff patterns and surface water storage, the annual unit runoff for Canada is also an indicator for runoff potential and thus contaminant transport. According to the Annual Unit Runoff Report, expected runoff volumes in an average year (i.e., 50% probability of exceedence) in southern Alberta can range from 2 to 600 dam^3 km^{-2} (Cole 2013). The report predicts higher values in Alberta's boreal forest, foothills, and mountain regions, and low values in the plains and prairie regions (Cole 2013).

1.3 Agricultural Impacts on Surface Water Quality

1.3.1 Nutrients

Eutrophication is the over-enrichment of receiving waters with nutrients, resulting in excessive production of algae and other aquatic vegetation (National Academy of Sciences 1993; Daniel et al. 1994; Correll 1998) (Figure 1.1). Eutrophication is a natural process but it can be accelerated by anthropogenic activities, producing adverse effects. The excessive production of algae and aquatic vegetation is accompanied by high respiration rates when the plants decay, leading to hypoxia or anoxia in lakes and streams, and the release of materials normally bound to bottom sediments including various forms of P (Kim et al. 2003; Ajmone-Marsan et al. 2006). Eutrophication is frequently associated with fish kills; loss of biodiversity; loss of aquatic plant beds and coral reefs; overall degradation of aquatic ecosystems; and the impairment of water quality for drinking, recreation, irrigation, and other purposes (Carpenter et al. 1998).

The combination of increased levels of dissolved inorganic nitrogen (DIN) and P (DIP), the ratio of DIN:DIP, and the influence of climatic factors such as light and temperature, have been associated with the eutrophication of surface waters (Isermann 1990). In freshwaters, P is the main limiting nutrient and nitrogen (N) is the second limiting nutrient, while in coastal waters the reverse is true (Schindler 1977; Cullen and Forsberg 1988; Isermann 1990; Blomqvist et al. 2004). Thus, most freshwater studies focus on P, and its control is paramount in minimizing the accelerated eutrophication of freshwaters (Schindler 1974; Sharpley et al. 1987).

Nutrients, such as total N (TN) and total P (TP), can be divided into dissolved and particulate forms, and each form can be further subdivided into inorganic and organic fractions (Gburek et al. 2005). Dissolved reactive P (DRP) is the fraction of P that reacts with molybdate during the Murphy-Riley analytical procedure (Murphy and Riley 1962) and this fraction consists of orthophosphate ($H_2PO_4^-$ or HPO_4^{2-}), other inorganic P forms, and some organic P (Gburek et al. 2005). The principal forms of N exported through runoff are ammonia (NH_3), ammonium ion (NH_4^+), and nitrate (NO_3^-) (Marston 1989). Ammonium and P are relatively immobile in the soil, while NO_3^- is mobile and can easily leach into groundwater (Chang and Entz 1996). Nitrogen and P as surface water contaminants often originate on land and are transported to surface water through overland routes such as runoff and erosion, and subsurface routes such as leaching and groundwater flow (Nash and Halliwell 2000; Haygarth et al. 2005).

Simply stated, transport involves the separation of nutrients from their sources. Nutrients are lost or transported by attaching to eroded sediment in surface runoff, dissolved in water in surface runoff, or dissolved in leaching water (Baker et al. 2008). Soil erosion by water can result in the direct transport of soil particles or sediment and associated nutrients into nearby surface water (Haygarth et al. 2005). Surface runoff as saturated overland flow can also carry nutrients to surface water in dissolved forms (Nash and Halliwell 2000). Infiltration or leaching involves the movement of water from the surface through the soil matrix either by preferential flow through macropores, or flow through soil that has not reached its infiltration capacity (Baker et al. 2008), settling in shallow groundwater and eventually making its way to surface water (Cooke et al. 2005).

Figure 1.1. Example of excessive plant and algae growth in water due to eutrophication.

The contribution of agriculture to accelerated eutrophication of surface waters (Sims et al. 2000; Smil 2000; Bennett et al. 2001) is well recognized, especially in the United States (Sharpley et al. 1987, 1999; Carpenter et al. 1998) and Europe (Isermann 1990; Smith et al. 2001a; Smith et al. 2001b). Agricultural inputs of N and P into soil in the form of inorganic fertilizers or livestock manure are essential for profitable crop production. However, excessive application of nutrients beyond crop utilization and removal can lead to nutrient accumulation in the soil, and this can increase the risk of nutrient loss in surface runoff. Runoff from agricultural land is one of the major sources of non-point source pollutants, particularly bioavailable P, which can impair water quality in lakes and streams (Sharpley 1993; Daniel et al. 1994).

Soil-test P level is an estimate of plant-available P. In many areas of the world, long-term trends in STP values have shown that soil P is now greater than crop requirements (Sims et al. 2000). In areas of intensive crop and livestock production in Europe (Barberis et al. 1995; Hooda et al. 1997), the United States (Daniel et al. 1993; Sharpley et al. 1996; McDowell et al. 2002), and Canada (Campbell et al. 1986; Simard et al. 1995), P has accumulated in soils to levels that are of concern for long-term eutrophication risk rather than an agronomic concern (Sharpley and Withers 1994; Sharpley 1995; Carpenter et al. 1998; Hooda et al. 2001; Sharpley et al. 2001). Intensive livestock production and concentrated animal feeding operations in many parts of the world have led to problems with the disposal of manures and wastewaters. There is a need to improve nutrient management to avoid problems associated with surplus nutrients entering the environment (Hooda et al. 2000; Sims et al. 2000).

In general, most of the STP values in Alberta are less than or equal to 25 mg kg^{-1} (Manunta et al. 2000), which is less than the agronomic threshold of 60 mg kg^{-1} (Howard 2006). However, there are fields in Alberta that have been measured with excessive STP (>200 mg kg^{-1}) as a result of repeated manure application (Svederus et al. 2006; Little et al. 2007). Several research studies in Alberta have shown the effects of manure application on the accumulation of N and P in soil (Chang and Janzen 1996; Chang et al. 2005; Olson and Papworth 2006; Olson et al. 2009, 2010a,b). Additionally, surveys have shown that 64% of Alberta farmers apply manure to their crop land (Brethour et al. 2007). This compares to reported rates in the other Canadian prairie provinces (Saskatchewan, 43%; Manitoba, 65%) and eastern Canadian provinces (Ontario, 75%; Quebec, 78%) (Brethour et al. 2007). The survey also revealed that in Alberta in particular, producers who use manure only applied manure to 22% of their land and only 34% of the producers that apply manure in Alberta used a formal manure management plan.

An initial water quality survey was conducted in Alberta in 1995 and 1996 on 27 streams and 25 lakes in runoff-prone agricultural areas (CAESA 1998). The associated watersheds were further classified as low, moderate, or high agricultural intensity based on livestock numbers, pesticides sales, and fertilizer sales. The study found that 99% of the high, 88% of the moderate, and 89% of the low intensity agricultural streams exceeded the Surface Water Quality Guidelines for TP in Alberta (0.05 mg L^{-1}) (Alberta Environment 1999). The corresponding values for TN (surface water guideline of 1.0 mg L^{-1}) were 87%, 65%, and 32% exceedance, respectively. Similarly, 96% of lakes in high intensity and 38% of lakes in low intensity areas did not comply with the TP guideline. Note that these two guidelines have been withdrawn in Alberta and replaced with narrative statements (ESRD 2014).

Following the CAESA study, the AESA long-term study continued monitoring 23 small agricultural watersheds in Alberta (Lorenz et al. 2008). This study found that in general, agricultural intensity influenced N and P concentrations such that higher agricultural intensity watersheds yielded higher N and P concentrations, while low intensity watersheds yielded lower N and P concentrations. High agricultural intensity watersheds from 1999 to 2006 had the lowest compliance with Alberta's TP and TN guidelines for the protection of aquatic life in use at that time. Although compliance in low and moderate intensity agricultural watersheds was better than high intensity agricultural watersheds in each year, overall compliance was still low for each year.

A 3-yr study of eight small (2 to 248 ha) agricultural watersheds in Alberta by Little et al. (2007) examined STP values in soil and P in runoff water. Average STP in the 0- to 15-cm soil layer ranged from 3 to 512 mg kg^{-1} among the eight watersheds. Little et al. (2007) found a direct linear relationship between STP and runoff P concentration. They also found that DRP, TP, and degree of soil P saturation were greater in manured cultivated fields than in non-manured cultivated fields and at an ungrazed grassland site. In the same study reported by Little et al. (2007), N in soil and runoff water was also measured, as summarized in Casson et al. (2008). Nitrogen applied to crops in the form of commercial fertilizers and livestock manures was found to be susceptible to losses through the same processes as P, such as surface runoff and leaching (Casson et al. 2008). Casson et al. (2008) found relationships between nitrate N (NO_3-N) in soil and TN in runoff (r^2 = 0.65 to 0.72) and NO_3-N in runoff (r^2 = 0.62 to 0.69). The authors also observed that NO_3-N concentrations in the three soil layer depths (0 to 2.5 cm, 0 to 5.0 cm, 0 to 15.0 cm) were similar, and that soil N concentrations from each depth predicted runoff N equally well.

Nutrient contributions to streams are often a function of soil nutrient concentrations and magnitude of spring and summer runoff events. When characterizing how soil affects surface water quality, it is also helpful to examine the role of sediment.

1.3.2 Sediment

Sediment loss and transport from land to surface waters is often dependent on the intensity of rainfall, physical and chemical attachment between various solid components, and the amounts and velocity of runoff waters (Guy 1970). Sediment and nutrient loads often occur co-dependently, but sediment itself is also a physical parameter of water quality. The amount of suspended sediment in surface water has implications for turbidity, light penetration, and temperature.

Turbidity is a measure of water clarity and is caused by suspended matter such as clay, silt, and fine organic and inorganic matter (USEPA 1999). In addition to being aesthetically unappealing, high turbidity from suspended sediments can negatively impact the depth of light penetration in the water. Light is required for photosynthesis by algae and macrophytes, which are found in the euphotic zone or wherever sufficient light penetrates to allow for photosynthesis (Chapman 1996; Gallagher and MacMillan 2007). If turbidity is too high, such photosynthetic processes may be impaired.

Rainfall and snowmelt events drive water erosion of soil, which in turn affects surface water quality in Alberta. Jedrych and Martin (2013) developed a water erosion potential map for agricultural land in Alberta where erosion rates were calculated as functions of area-specific information relating to climate, soil, landscape conditions and a uniform land-use scenario. Predicted erosion rates ranged from 0 to 783 Mg ha^{-1} yr^{-1}. As expected, Jedrych and Martin (2013) found that highest erosion rates were on hill slopes adjacent to river valleys, and the lowest rates were on flat land.

1.3.3 Microorganisms

Livestock manure, particularly untreated slurry and feces of grazing animals, can carry a variety of bacterial, protozoan, and viral microbes from diseased and carrier animals (Mawdsley et al. 1995; Hooda et al. 2000). Microbial contamination of water supplies may occur as a consequence of leakage from manure in buildings or storage facilities, application of manure to land, direct access of livestock to streams, and feces deposited on pasture by grazing animals (Figure 1.2). Mawdsley et al. (1995) listed 11 bacteria, three viruses, and four protozoa (parasites) from livestock waste that may cause human diseases. Wildlife may also play a role in microbial contamination of waters (Niemi and Niemi 1991). Since microbes can survive for long periods in the environment, it is a matter of concern not only for livestock health but also for human health, which can be affected through contact with contaminated water.

Microbes can be transported in surface runoff (Tyrrel and Quinton 2003) in addition to nutrients and a host of other contaminants. Microbe contaminated runoff has been reported from manure-applied fields (Patni et al. 1985; Thornley and Bos 1985), grazed pastures (Doran and Linn 1979; Doran et al. 1981; Howell et al. 1995), barnyards and manure piles (Thornley and Bos 1985), and feedlots (Young et al. 1980). Assessment of runoff bacterial contamination is generally achieved through the use of indicators such as total coliforms, fecal coliforms, fecal streptococci, or enterococci. *Escherichia coli* (*E. coli*) are also used as an indicator organism for detecting environmental fecal pollution (Mawdsley et al. 1995). Lorenz et al. (2008) found a strong

Figure 1.2. Direct cattle access to a river adjacent to a pasture.

relationship between total suspended sediment and median annual concentrations of fecal coliforms (r = 0.775, $P<0.005$) and *E. coli* (r = 0.782, $P<0.005$) for 23 agricultural streams in Alberta, suggesting that streams with higher suspended sediment are more likely to have higher bacteria concentrations. Further, Lorenz et al. (2008) found that 37 to 96% of the samples among the 23 agricultural streams were in compliance for irrigation fecal coliform guideline (100 cfu 100 mL^{-1}) and 22 to 100% of samples were in compliance for *E. coli* recreation guideline (200 cfu 100 mL^{-1}). They also reported that fecal coliform counts, unlike nutrients, did not increase with agricultural intensity, suggesting that source and transport mechanisms may not be the same between fecal bacteria and nutrients.

It is also understood that fecal coliforms can exist in the environment for long periods of time. High concentration of fecal coliforms in runoff can persist for more than 1 yr after cattle are removed from a grazed area (Jawson et al. 1982). Bacteria persisted in the soil for at least 2 yr after application of dairy manure slurry on a grassland was stopped (Bittman et al. 2005), for 143 d for *Salmonella* after application of liquid pig manure (Gessel et al. 2004), and for 60 d for *E. coli* after cattle were removed from a grassland (Oliver et al. 2005). Even when obvious known sources (manures) of fecal microbes in a watershed are identified, studies to date have not found statistically significant relationships between bacterial concentrations in water and confined feeding operations (Johnson et al. 2003; Little et al. 2003).

Prolonged survival of bacteria in sediments suggests increased risk of exposure to microorganisms when sediments are re-suspended (Craig et al. 2002; Alm et al. 2003). Significant *E. coli* loading in stream and canal segments in southern Alberta suggests that certain practices may be disproportionately contributing to bacterial contamination of local surface waters, or that re-suspension of bacteria in sediments is occurring between sites. Weak correlations found between discharge and *E. coli* concentrations in the Lower Little Bow River suggest that the re-suspension of sediments was not the only source of *E. coli* peaks in the watershed (Gannon et al. 2005). The authors attributed this to direct access of cattle between upstream and downstream sites (Gannon et al. 2005). The researchers also found increased concentrations of *E. coli*, Streptococcus, and fecal coliform concentrations in downstream versus upstream water samples throughout the Lethbridge Northern Irrigation District, suggesting that open channel irrigation canals may promote field-to-stream linkages by increasing the drainage density of the landscape. Gannon et al. (2005) suggested the use of constructed wetlands near termini of irrigation returns, storm sewer outlets in urban areas, and in industrial effluent channels to remove biological pollutants before they reach important drinking sources.

Escherichia coli have been reported to peak in surface water during the warmest months of the year (Hyland et al. 2003; Johnson et al. 2003) and the "first flush" phenomenon occurs when there is a rise in bacteria at the onset of a precipitation event following a period of dry weather (Tong and Chen 2002). Gannon et al. (2005) determined that *E. coli* in southern Alberta reservoirs did not originate from in-stream sediments, but rather from non-point sources. However, sedimentation of bacteria in local reservoirs may pose a potential human health risk as bacterial concentrations can be up to 1000 times greater in sediments compared to the water column (Hendricks and Morrison 1967; Gannon et al. 2005).

1.4 BMP Types and Options

Several different BMP options are available to producers to minimize the environmental impacts of their farm operations (Figure 1.3). Different operations require different BMPs, and much of the literature has grouped BMPs into several different categories. For instance, Sharpley et al. (2006) differentiate between source and transport BMPs and argue that the goal of source BMP implementation is to reduce nutrient loss at the source by minimizing buildup of nutrients in the soil, ensuring that nutrient concentrations in the soil do not exceed levels needed for a given activity (i.e., optimum crop growth). Source BMPs include practices such as livestock relocation and improved manure storage. Arguably, it may be more desirable to implement source BMPs as it is less costly to prevent nutrient loss than to treat the effects of excess soil nutrients on water quality. Transport BMPs are practices that control the movement of nutrients from soils to waterways by limiting runoff, erosion, and leaching (Sharpley et al. 2006). Examples of transport BMPs include conservation tillage, irrigation management, and stream slope stabilization (bioengineering). Further, Rao et al. (2012) categorized BMPs into structural and management groups. As the name would suggest, structural BMPs include physical structures and buildings that incur one-time construction costs and subsequent maintenance costs, such as manure storage buildings or detention ponds. Management, or non-structural, BMPs are strategies that reduce the quantity of contaminants without a structural facility and are implemented on a continuous basis, such as changing manure application practices (Chang et al. 2007). Table 1.1 provides an overview of the different BMP options.

While several BMPs have been developed for managing manure from the livestock industry, and nutrient management in general, it is unlikely that a single BMP will effectively reduce or eliminate negative environmental impacts. Often, it is a combination of BMPs that will result in reduced environmental impacts (Bishop et al. 2005; Sharpley et al. 2006; Chaubey et al. 2010). Li et al. (2011) monitored two individual BMPs and a suite of three BMPs for effects on water quality within an agricultural watershed in south-central Manitoba. The two individual BMPs included a

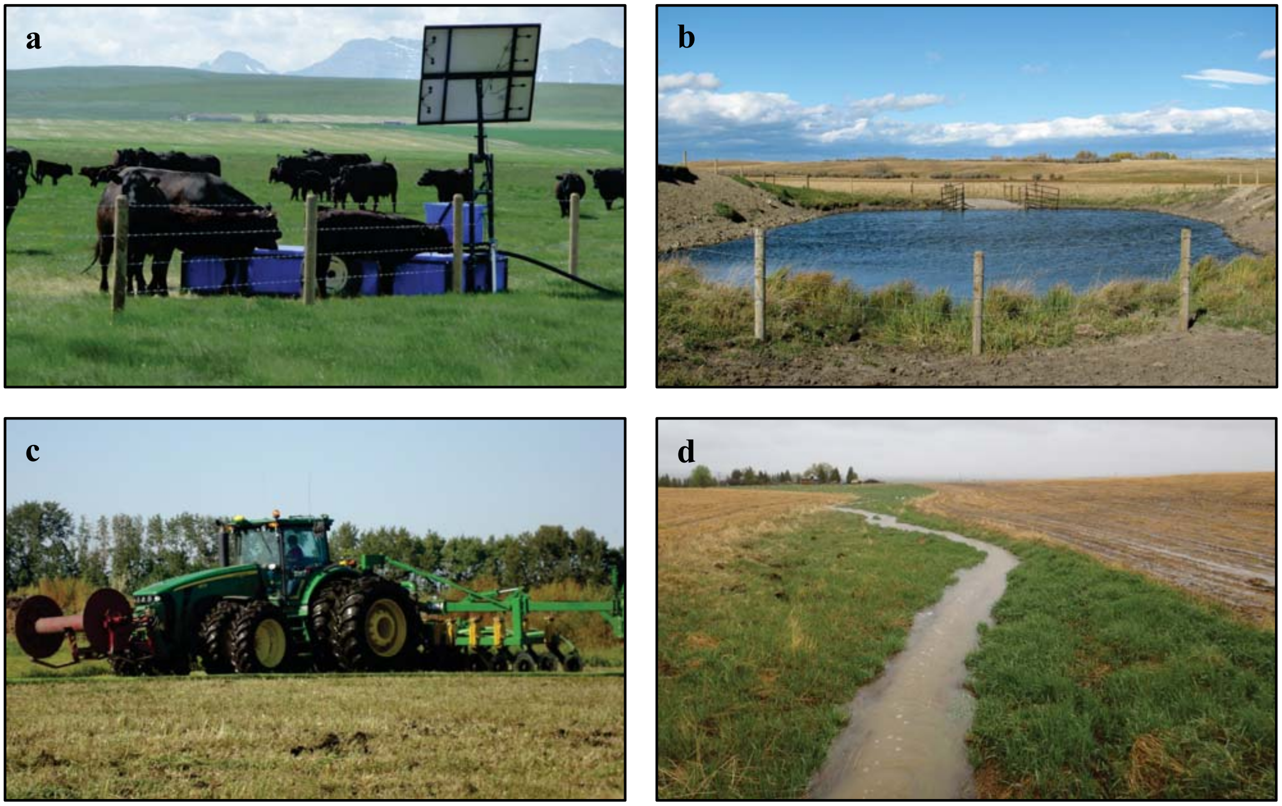


Figure 1.3. Examples of beneficial management practices: (a) off-stream watering for cattle, (b) control of cattle access to water using fencing, (c) injection of liquid manure, and (d) a grass channel within a cultivated field.

holding pond downstream of a confined feedlot operation and the conversion of annual cropland to forage, while the suite of three BMPs included grassed waterway management, grazing management, and nutrient management. It was concluded that the combination of all five BMPs was effective at reducing nutrient loss, but the effects of individual BMPs were difficult to discern due to varying factors. Arguably, the main decision facing producers is to decide which combination of BMPs is best suited to their operations (Sharpley et al. 2006).

1.5 Methods to Evaluate BMP Effectiveness

The effectiveness of BMPs on the environment can be measured by several parameters such as water quality, surface flow, species richness, soil conditions, and riparian health. Within Canada and the United States, several studies have examined the effectiveness of BMPs at field and watershed scales, and some have conducted economic feasibility analyses. Examples of two major research projects are the Watershed Evaluation of Beneficial Management Practices (WEBs) project in Canada (AAFC 2007) and the Conservation Effects Assessment Project (CEAP) in the United States (USDA 2008). Primarily focused on water quality, the WEBs project was initiated at seven small watersheds across Canada in order to assess the environmental and economic performance of different BMPs and to inform policy and decision making. At each watershed, a

Table 1.1. A summary of different beneficial management practice (BMP) options available. Type denotes if the BMP is source (S), transport (T), structural (St), management (M), crop (C), or livestock (L).

BMP	Type	Description and purpose	Evaluated in this study	Supporting literature
Grazing management	S, M, L	• decreases soil loss and direct transfer of fecal matter and nutrients by separating livestock and streams, and maintaining riparian vegetative cover	Yes	• Owens et al. 1996 • Sheffield et al. 1997 • Schwarte et al. 2011
Manure storage areas	S, St	• prevents manure from leaching into the ground or ending up as runoff	No	• Fullhage 1997 • Inamdar et al. 2002
Stream bank stabilization (bioengineering)	T, St	• stabilizes steep stream slopes to prevent water contamination by erosion with use of biological, mechanical, and ecological concepts	Yes	• Meals 2001 • Barret et al. 2006
Irrigation management	T, M, C	• efficient use of water and minimal erosion and runoff by determining and controlling the rate, amount, and timing of irrigation water application, and use of water-efficient equipment	Yes	• AFRD 2004 • Sharpley et al. 2006
Manure composting	S, St	• increased retention of nutrient and coliform bacteria, makes nutrient properties more uniform allowing for even application of nutrients	No	• Fitzpatrick et al. 2005 • Larney et al. 2006
Conservation tillage	T, M, C	• reduces transport of sediment and sediment-bound nutrients	No	• Ziemen et al. 2006 • Tiessen et al. 2010
Riparian buffer zones	T	• reduce nutrient loading to adjacent streams by retaining and transforming nutrients	Yes	• Duchemin and Hogue 2009 • Hoffman et al. 2009
Grassed waterways	T	• reduce runoff and erosion, filters sediment	Yes	• Chow et al. 1999 • Inamdar et al. 2002
Artificial wetlands, lagoons, and sediment basins	T, St	• reduces nutrient transport by capturing nutrient-enriched runoff	No	• Cronk 1996 • Li et al. 2011
Manure/fertilizer application management	S, M	• minimizes nutrient loss in runoff based on the rate, method, and timing of applications	Yes	• Srinivasan et al. 2006 • Easton et al. 2008

suite of BMPs were applied, which were selected to match the conditions of each individual watershed (Stuart et al. 2010). It was found that more than half of the BMPs had the potential to reduce contaminant loading to surface waters (Stuart et al. 2010).

One of the main goals of the CEAP was to establish scientific understanding and quantification of the effects of conservation practices at the watershed scale in 13 sites throughout the United States (Duriancik et al. 2008). The different BMPs used in the study included irrigation management practices, conservation buffers, nutrient management, and tillage management (Duriancik et al. 2008). Some of the major findings were that constructed wetlands reduced the movement of nitrate from tile-drained fields to streams, while riparian buffers were effective in mitigating the loss of nutrients and bacteria in runoff, and fertilizer management techniques reduced nitrate losses from fields (Richardson et al. 2008). The WEBs and CEAP studies were able to show the ability of BMPs to improve surface water quality as well as the health of nearby soils and habitat.

1.5.1 Experimental Designs

Common approaches to evaluate the effectiveness of agricultural BMPs on water quality include three experimental monitoring designs: paired watershed design, before-after monitoring, and upstream-downstream designs (Spooner et al. 1985; Sheffield et al. 1997; Grabow et al. 1998; Mostaghimi et al. 2001). In some instances, it may be appropriate to combine several of these approaches into the same monitoring program.

The paired watershed approach consists of two or more watersheds where at least one watershed experiences BMP implementation (i.e., treatment watershed) and at least one watershed remains unchanged (i.e., reference or control watershed) (Spooner et al. 1985). It is assumed that two or more nearby watersheds with similar physical properties (i.e., soils, land use, climate) will respond in predictable manners. Both watersheds experience the same monitoring regime and the temporal trends of response variables are compared between the control and treatment watersheds (Lemke et al. 2011). Paired watershed designs often involve use of the before-after approach, as discussed below.

In the before-after approach, water quality data are collected from a location downstream from the BMP, usually the outlet, for a period of time before and after BMP implementation (Grabow et al. 1998). The before-after approach for a single watershed does not use a control watershed (Mostaghimi et al. 2001). However, when before-after designs are combined with a paired watershed approach, water quality data are collected from a control watershed and a treatment watershed at time periods before and after BMP implementation. Any differences identified, namely improvements in water quality, may be indicative of the effect of implementing the BMP. A before-after single watershed design was used in our current study.

Tiessen et al. (2010) used a paired watershed, before-after approach to compare N and P losses from tillage BMPs in southern Manitoba. During the pre-conversion period, both watersheds were managed by using conventional tillage practices. After some time, the conservation tillage BMP was implemented in the treatment watershed and both watersheds experienced the same sampling and monitoring regime. The authors found that the conversion to conservation tillage reduced exports of total suspended solids and TN in the treatment watershed, suggesting improved water quality in runoff from the treatment watershed. Inamdar et al. (2001) implemented a before-after approach in their study aimed at determining the impact of cropland BMPs on surface and ground water quality in a single agricultural watershed in Virginia, United States. The authors observed reductions in N and P loads up to seven years after conservation tillage, nutrient management, and vegetative buffer strip BMPs were implemented.

The upstream-downstream approach is typically only used with single-watershed studies (Grabow et al. 1998). As the name suggests, monitoring is performed upstream and downstream of a BMP site. If a larger water body is used, this design requires that the water from a BMP site directly enter the stream or river being monitored, thus allowing for the differentiation between water upstream and downstream of the BMP site (Mostaghimi et al. 2001). It is assumed that changes in the response variables are due to BMP implementation (Miller et al. 2010).

Monitoring frequency, baseline sampling, and event-based sampling must take into account the experimental design chosen. For example, if sampling is too frequent, autocorrelation of data may occur. If sampling is too infrequent, critical information may be missed (Mostaghimi et al. 2001). A monitoring program should be long enough to capture variations in watershed hydrology response to weather (i.e., storm or rainfall-runoff) events (Easton et al. 2008; Duchemin and Hogue 2009). However, it should also include event-based monitoring because nutrient loss from a watershed is largely a result of rainfall-runoff events (Sharpley et al. 2008).

There are several more considerations to account for in designing a BMP evaluation study, such as determining sampling locations, the degree of precision needed for the study, lag effects between BMP implementation and observing a change, and point and non-point contaminant sources (Mostaghimi et al. 2001; Bishop et al. 2005; Maguire et al. 2009; Sharpley et al. 2009). Perhaps the most challenging and widely discussed issue in a study design is that of scale. Agricultural activities have the capacity to affect adjacent waterways at the edge-of-field and watershed scales. Accordingly, BMP study designs must take into consideration what scale a BMP or suite of BMPs is expected to mitigate landscape effects in order to make appropriate monitoring decisions. If the monitored area is too large, efforts may be futile in that small fluxes may not be observed, but if the scale is too small, basin-wide water quality standards may not be met (Gove et al. 2001). The effectiveness of BMPs at the small scale is generally better understood than at larger watershed scales (Inamdar et al. 2002; Jackson-Smith et al. 2010). Larger-scale research can be challenging due to high financial cost and because replication is less feasible. Additionally, at the larger scale there is a wider range of climatic, environmental, and landscape interactions and variability. This variability can include flow pathways, nutrient control mechanisms (Sharpley et al. 2009), and topography making it difficult to implement control measures. Although a comprehensive discussion of spatial scale is beyond the focus of this study, site-specific scale factors and detailed knowledge and understanding of the area (including point and non-point sources) should drive BMP study experimental design.

1.5.2 Statistical Techniques

Before BMPs can be evaluated for their effectiveness on improving water quality or other environmental parameters, there are several steps that must be taken to prepare the data for analysis. Exploratory or descriptive data analysis must be carried out in order to identify data discrepancies such as autocorrelation, outliers, errors, missing data, and censored data (Helsel and Hirsch 1992). Descriptive statistics are essential for understanding the nature of the data, such as the distribution, cycles, skewness, and allow for visual interpretation (Mostaghimi et al 2001; Meals 2011). Once exploratory data analysis has been conducted and statistical testing deemed appropriate, tests for normality must be performed to determine if the raw data can be used in parametric tests, or if the data should first be transformed (e.g., log transform, rank). If a data transformation does not yield a normal distribution, non-parametric statistical tests should then be used. Quite often in water quality studies, multivariate statistical analyses are utilized (Little et al. 2003; Bishop et al. 2005), such as when several water quality response variables are tested against several BMPs or agricultural practices. There are some instances where one response variable may be examined, in which univariate techniques may suffice.

Once the initial data exploration has been performed, decisions can be made for which statistical techniques are appropriate for the data. Several of the standard statistical techniques for evaluating BMP effectiveness on water quality are further discussed in the following sub-sections.

1.5.3 Considerations for Water Quality Trend Detection Analysis

Trend detection methods are important for determining if water quality in a stream is improving, deteriorating, or remaining static with time (Westbrook and MacEachern 2002). Identifying the factors that influence water quality fluctuations with time allows for appropriate decisions to be made for improving water quality. Several statistical methods exist for detecting trends. However, the tests chosen must be suited to the nature of the data, study design, and objectives.

Monotonic trend versus step trend. Monotonic trends are described as gradual changes in a water quality parameter in one consistent direction with time, such as changes in the ambient concentration of a nutrient in a stream (Westbrook and MacEachern 2002). Monotonic trend tests are used to detect changes in water quality with time and are appropriate only if there is no prior assumption of a discrete change (Helsel and Hirsch 1992). The Mann-Kendall test determines the linear dependence of a variable on time (Westbrook and MacEachern 2002) and is used if parameters do not exhibit significant seasonality. Other techniques include regression, multiple regression, and the seasonal Mann-Kendall test (Table 1.2).

Alternatively, it is assumed in step trend analysis that data collected before and after a specific time are from different populations. Step trend analysis is used to analyse an abrupt shift at a specific point in time, or where a known discrete event has occurred (Walker 1994). Because this study is aimed at assessing the impact of BMP implementation on water quality concentrations, it is suited for step trend analysis. Common step trend analyses include two sample *t* tests, analysis of variance (ANOVA), analysis of covariance (ANCOVA), and Mann-Whitney (Table 1.2).

Table 1.2. A summary of commonly used statistical techniques in the literature for determining water quality trends and impacts of beneficial management practices on surface water quality.

Test	Monotonic (M) or step trend (ST)	Parametric (P) or non-parametric (NP)	Purpose	Supporting literature
Linear regression	M	P	• Shows the relationship between a dependent variable and an independent variable	• Udawatta et al. 2004 • Lemke et al. 2011
Multiple linear regression	M	P	• Shows the relationship between a dependent variable and more than one dependent variables	• Little et al. 2003 • Schwarte et al. 2011
Mann-Kendall/ Wilcoxon rank sum	M	NP	• Determines the linear dependence of a variable on time	• Glozier et al. 2006 • Inamdar et al. 2002
Seasonal Mann-Kendall	M	NP	• Removes the effects of seasonality on water quality variables to determine linear dependence of a variable on time	• Tian and Fernandez 2000 • Glozier et al. 2006
LOWESS	M	NP	• Removes the effects of discharge on water quality variables	• Esterby 1996 • Thomas and Pool 2006
t-test	ST	P	• Compares the means of two samples with time	• Withers et al. 2009 • Miller et al. 2010
ANOVA	ST	P	• Compares if two or more means are equal with time	• Chaubey et al. 2010 • Tiessen et al. 2010
ANCOVA	ST	P	• Compares two or more independent variables for a linear relationship	• Bishop et al. 2005 • Flores-López et al. 2010
Mann-Whitney	ST	NP	• A non-parametric test to compare the means of two samples with time, analogous to the parametric *t*-test	• Arheimer and Lidén 2000 • White et al. 2009
Kruskal-Wallis	M	NP	• Tests for the presence of seasonality	• Ontkean et al. 2003 • Jeffries et al. 2010

Monotonic and step trend analyses help to identify increasing and decreasing patterns in water quality and estimate rate of change. However, establishing causality of an observed change requires different statistical techniques.

Removal of variance from exogenous factors (data reduction). Much of the variance in water quality parameter concentrations is a function of stream flow (Hirsch et al. 1991; Meals 2011). Dilution conditions occur when the input of a contaminant remains the same with increasing flow. However, the introduction of increased contaminant concentrations to a stream typically occurs with overland flow processes, which are often observed with increased flow conditions (Hirsch et al. 1991). Flow adjustment is commonly used to remove this source of variance from the data, making subsequent trend analyses more powerful. The locally weighted regression and smoothing scatterplot (LOWESS) (Cleveland and Grosse 1991) technique is a commonly used data smoothing technique for describing trends in water quality data (Westbrook and MacEachern 2002) (Table 1.2). It is a non-parametric smoothing procedure that removes the effect of flow on a water quality variable (Esterby 1996).

Throughout the year, natural changes in water chemistry within a stream follow patterns that respond to changes in hydrologic regime, such as variations in precipitation, temperature, and evapotranspiration (Helsel and Hirsch 1992; Glozier et al. 2006). These seasonal fluctuations in stream nutrient concentrations can be magnified by anthropogenic activities (e.g., agriculture) on the surrounding landscape, which may also have a seasonal component (Jensen et al. 2011). Consequently, seasonal effects must be accounted for in order to better identify true concentration trends with time. The Seasonal Mann-Kendall test (Table 1.2) accounts for seasonality by computing the Mann-Kendall test on each of the seasons separately, and then combining the results, thereby, removing the effect of seasonality prior to running the significance test (Tian and Fernandez 2000; Glozier et al. 2006).

It should also be noted that autocorrelation, which is the tendency for samples that are taken close together to have similar values, must also be corrected before trend analysis is performed in order to prevent the interpretation of a false trend in a data set. Exploratory data analysis techniques such as normality testing and graphical visualisation during the data exploration phase of a study are also critical to correct for data errors and choosing data-appropriate statistical tests (Westbrook and MacEachern 2002; Meals 2011). These techniques are summarized in Table 1.2.

1.5.4 Other Methods to Evaluate Water Quality Data

In some instances, it may not be possible or necessary to conduct robust statistical testing on data (Esterby 1996; Owens et al. 1996). For example, it would be difficult to conduct further analysis on trends of N concentrations in surface water from different countries due to the lack of consistency in sampling and analytical techniques (Esterby 1996). Other ways to assess water quality are available. Graphical representation, an exploratory method, is commonly used as a simple and effective alternative to examine, interpret, and represent data. It is useful for the immediate identification of the data distribution, skewness, and temporal trends. Common graphs used to represent water quality data include histograms, scatterplots, time-series plots, and box and whisker plots (Figure 1.4) (Esterby 1996; Arheimer and Lidén 2000; Cooke et al. 2005; Meals 2011). As well, comparisons can be used as a way to identify water quality properties. Comparing water quality measures to pre-determined standards, such as water quality guidelines or environmental thresholds, is effective and frequently utilized in BMP studies (Cooke et al. 2005; Saffran 2005; Inamdar et al. 2002; Chambers et al. 2012). Surface water quality guidelines exist for aquatic, agricultural, recreational, and aesthetic protection. Comparing water quality data against these guidelines can help determine if concentrations are representative of good quality or of poor quality. Similarly, comparisons can be made among different time periods or different watersheds. Finding the difference in values between pre- and post-BMP implementation, for example, can indicate the direction and magnitude of change in water quality parameters (Sheffield et al. 1997).

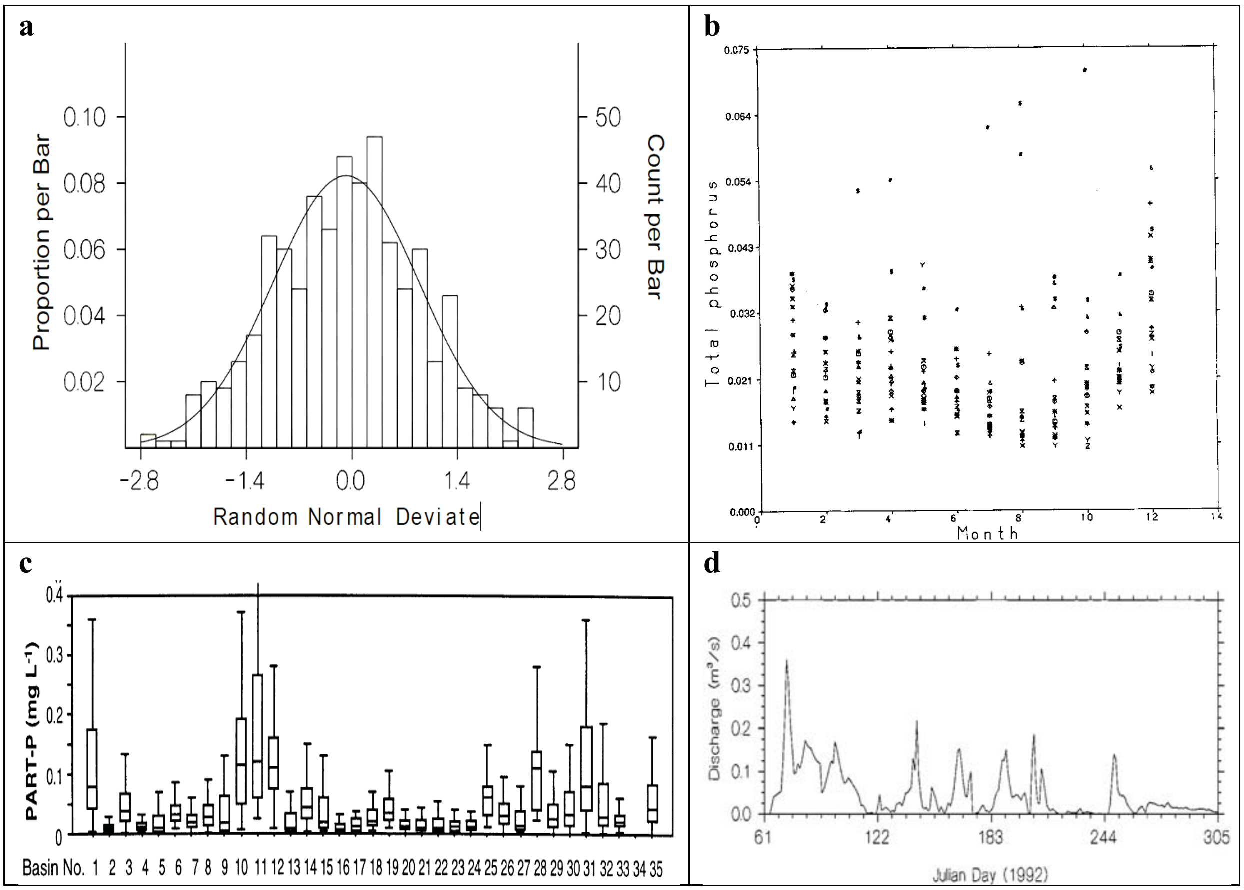


Figure 1.4. Examples of common graphs used in exploratory data analysis: (a) histogram showing normally distributed data (Cooke et al. 2005), (b) scatterplot showing annual variability (Esterby 1996), (c) box and whisker plot showing concentration range between basins (Arheimer and Lidén 2000), and (d) a time series graph (Cooke et al. 2005).

Whether used in statistical testing or other non-statistical evaluation methods, water quality data can be represented in several different ways. An important consideration when analyzing water quality data is the influence of hydrologic conditions. Stream flow can vary greatly between high- and low-flow periods, and fluctuate seasonally and meteorologically (Brannan et al. 2000; Cooke et al. 2005; Saffran 2005), potentially effecting water quality responses. This variability is especially the case in recent years due to increasing climatological uncertainty (Council of Canadian Academies 2013). Concentrations of contaminants of interest (e.g., mg L^{-1}) are appropriate for determining ambient water quality in a stream or if there is concern for exposure of populations that reside or depend on the stream (Hirsch et al. 1991). However, it is understood that concentration alone is usually not sufficient for addressing all water quality concerns within a watershed (Glozier et al. 2006). Ideally, when adequate stream flow measurements are available, concentrations are combined with stream flow in calculations of mass loads, export coefficients, and flow-weighted mean concentrations, which all provide a more robust interpretation of water quality in a given water body (Cooke et al. 2005; Quilbé et al. 2006).

A mass load is commonly used in studies that aim to evaluate water quality (Duchemin and Hogue 2009; Miller et al. 2010; Tiessen et al. 2010). It is the total mass of a solute or contaminant passing a certain point on a stream or river for a length of time (e.g., kg d^{-1}) and is the product of concentration (mg L^{-1}) and flow (m^3 s^{-1}) (Saffran 2005). Contaminant concentrations tend to fluctuate with low and high flows. Because mass loading is a flow-proportionate measure, it provides a better idea of the total mass of a contaminant in the stream than concentration alone (Cooke et al. 2005).

An export coefficient is determined by dividing the mass load by the area of the watershed, and gives an estimate of the amount of material loss per unit area of the watershed per unit time (e.g., kg ha^{-1} yr^{-1}). This allows for comparisons of contaminant export from watersheds of different sizes. A high export coefficient value would indicate a potential source of contamination in that more material is entering the water than would normally be expected (Saffran 2005). For example, fertilizer lost from a farming operation could be estimated based on export coefficients of nutrient loss from a field (Cooke et al. 2005). Salvano and Flaten (2009) calculated export coefficients of P to evaluate relationships between water quality data in agricultural watersheds in southern Manitoba.

Flow-weighted mean concentration (FWMC) is another commonly used water quality measure in BMP studies (Sheffield et al. 1997; Little et al. 2003; Sharpley et al. 2008; Tiessen et al. 2010). It is the concentration adjusted for variability in stream flow with time and is calculated by dividing the mass load by the total volume of water during a specified period (e.g., daily or annual). The FWMC allows for comparison among streams with different flows, or flows of one stream among different years with different flows. In their study to evaluate the impact of water troughs as a BMP on soil, nutrient, and bacterial loss from pasture lands, Sheffield et al. (1997) used FWMC as a measure of water quality. They calculated percent differences in FWMC between pre- and post-BMP implementation periods and found reductions in total suspended solids, TN, NH_4-N, and TP.

1.5.5 Models

Watershed and hydrologic models are excellent tools to simulate water quality response in streams. Such models are valuable because they can evaluate BMP effectiveness in agricultural watersheds and results can be used to inform management decisions (Easton et al. 2008; Chaubey et al. 2010). A popular and extensively developed tool for hydrology studies is the Soil and Water Assessment Tool (SWAT), which has been widely used in studies examining the impact of land-use activities on quality of surface water (Santhi et al. 2001; Secchi et al. 2007; Yang et al. 2007; Chaubey et al. 2010; Jha et al. 2010). The SWAT model is a small watershed to river basin-scale model that simulates the quality and quantity of surface and ground waters as a result of land management decisions, and is used in assessing non-point source pollution management in watersheds (USDA 2013).

Several studies suggest using watershed models to simultaneously assess economic costs and environmental benefits associated with BMP implementation (Secchi et al. 2007; Yang et al. 2007). Yang et al. (2007) highlighted several challenges associated with evaluating the effectiveness of BMPs or conservation programs, such as unknown costs and adoption rates of BMPs, accounting for complex contaminant transport processes, and understanding trade-offs between economic and environmental effects. In response, the authors provided an integrated economic-hydrologic modelling framework to evaluate the economic and environmental performances of BMP implementation. The framework includes an on-farm economic model, a farmer adoption behaviour model, a watershed modelling tool box, and a non-market valuation model (Yang et al. 2007).

In our current study, the value of models to evaluate BMPs and associated economic considerations has been recognized as a key component. The Comprehensive Economic and Environmental Optimization Tool (CEEOT) was chosen for this study. The CEEOT model is capable of evaluating a wide range of BMPs at the watershed scale, and also includes economic and policy modules. The application and description of the CEEOT model in this study are detailed in Jedrych et al. (2014).

1.6 Project Objectives

Sustainable agriculture combines optimum agricultural productivity and profitability with no or minimum damage to the environment, especially soil and water resources. Beneficial management practices are practical control measures (including technological, economical, and institutional considerations) that have been demonstrated to effectively minimize environmental degradation (Ice 2003). For a number of years, BMPs have been promoted to, and implemented by, producers. However, these have not been scientifically and widely evaluated under Alberta conditions for their environmental effectiveness and economic benefit to agricultural producers. Numerous BMPs have been suggested and promoted to improve all aspects of farm operations. A common recommendation is that these BMPs not be implemented individually at individual sites, but rather as 'suites' that include several BMPs relevant to the intended outcome and characteristics of the site. This study followed this recommendation and implemented different suites of BMPs at each field site. The BMPs were assessed to determine if they improved surface-water quality, and other indicators, such as riparian and rangeland health, were assessed at some sites. Nutrient BMPs were examined in this study, with a focus on livestock production systems. In this study, three main BMP types were evaluated: manure nutrient management, livestock management, and surface-water management. The specific project objectives were to:

- Evaluate the effectiveness of nutrient BMPs in reducing the agricultural impacts on the environment at the farm scale.
- Assess the effects of BMPs on water quality in specific reaches of a watershed stream.
- Predict the cumulative effects of BMPs on the overall quality of a watershed stream using models (refer to Volume 3; Jedrych et al. 2014a,b).
- Evaluate nutrient BMPs for effective use of manure in crop production.
- Assess economic costs and benefits associated with BMPs implemented in this study.