FALLOW REPLACEMENT AND ALTERNATIVE FERTILIZER PRACTICES: EFFECTS ON NITRATE LEACHING, GRAIN YIELD AND PROTEIN, AND NET REVENUE IN A SEMIARID REGION

by

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DEDICATION

This thesis is dedicated to my wife, Katie, whose unwavering love and encouragement helped me through the process. I would not be who I am today without her in my life.

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ABSTRACT

High nitrate concentrations in groundwater have been observed in agricultural regions worldwide. In the Judith River Watershed of central Montana, groundwater nitrate concentrations have increased from 10 to 23 mg L^{-1} over the span of 20 years. Nitrate leaching from agricultural fields is a major concern for growers and stakeholders in the region. Little research has been conducted in dryland semiarid regions on the effects of agricultural practices on nitrate leaching.

We conducted a 2-yr study comparing three alternative management practices (pea rotation, controlled-release urea, split nitrogen application) to grower standard practices (fallow, conventional urea, spring broadcast urea) on grain yield, grain protein, net revenue, and the amount of nitrate leached. Eight field treatment interfaces were established across three farms and each treatment was in duplicate per year. Ten soil and biomass sampling locations were designated on both sides of the interface. Net revenue was calculated by enterprise budgets constructed from local and state data. Nitrate leaching was calculated using a nitrogen mass balance equation.

Replacing pea with fallow decreased winter wheat grain yield and protein yet had no effect on net revenue during the first year of the study (2013). In the second year, pea-winter wheat earned \$83 ha^{-1} more ($P<0.1$) than fallow-winter wheat. Neither fertilizer alternative management practice had an effect on net revenue. In the 2013 treatment year, wheat after pea leached less nitrate (20 kg N ha^{-1}) than wheat after fallow (56 kg N ha^{-1}), indicating more deep percolation of nitrate with fallow practice. In the 2014 treatment year, a greater amount of nitrate leached ($P<0.1$) while using controlled-release urea than conventional urea, possibly in part because the controlled release urea was applied earlier than conventional urea.

The results of our study revealed that replacing fallow with pea can decrease the amount of nitrate that leaches out of the root zone. Also, this practice either increased or had no effect on net revenue, revealing its ability to be economically feasible for a grower to implement. Based on our findings, future research should likely focus on practices that decrease rates of deep percolation.

CHAPTER ONE

INTRODUCTION

Background

Nitrate (NO_3^-) leaching potentially has negative health, environmental, and economic impacts. Nitrate leaching occurs when water percolates downward through the soil profile and out of the root zone carrying the highly mobile nutrient (NO_3^-) with it (Lehmann and Schroth 2003). This issue has been well studied in certain regions, especially in irrigated systems. A common finding is that increased nitrogen (N) input to agroecosystems results in increased groundwater NO_3^- concentrations (Power and Schepers 1989; Nolan and Stoner 2000; Burow et al. 2010; Puckett et al. 2011).

With the advent of the Haber-Bosch process, synthetic fertilizer application has increased dramatically to where approximately 120 Tg or 75 % of anthropogenically fixed N is added in the form of fertilizer to agroecosystems each year (Galloway et al. 2003). In the United States (U.S.) alone, annual fertilizer use increased from 0 to approximately 13 billion kg between 1940 and 2003. Concurrently, NO_3-N concentrations in shallow unconfined drinking water wells across the United States have increased from 2 to 15 mg L^{-1} (Puckett et al. 2011).

The rise of NO_3-N concentrations in groundwater has led to government regulation, including a nationwide drinking water standard of 10 mg L^{-1} set by the U.S. Environmental Protection Agency (EPA) in 1976 (U. S. Environmental Protection Agency 2001). However, even with these standards, NO_3^- contamination in groundwater continues to be a problem. A recent study conducted by the United States Geological Survey (USGS) showed that out of more than 1,400 wells sampled in various agricultural locations throughout the United States, 20 % had NO_3-N levels above the drinking water standard (Burow et al. 2010). The high NO_3-N concentrations in water across the United States are a problem for human and livestock health and environmental quality, and they reflect loss of potential economic return given costs of fertilizer application.

Groundwater Nitrate and Health Implications

The consumption of drinking water containing NO_3-N concentrations above the standard has been linked to methemoglobinemia ("blue baby syndrome") in infants and young livestock. Ingestion of high NO_3-N levels can cause the conversion of hemoglobin to methemoglobin, which does not carry oxygen through the bloodstream efficiently (Knobeloch et al. 2000). Affected infants and young livestock turn blue due to the lack of oxygen, possibly causing death or brain damage (Self et al. 2008). The first documented case of methemoglobinemia in humans after NO_3^- contaminated drinking water was consumed by infants was in 1945, resulting in a recommendation that NO_3-N concentrations above 10 mg L^{-1} were not safe for consumption (Comley 1945). The study also found that this was especially true in infants with particular kinds of gastrointestinal infection. Shortly after, the American Public Health Association (APHA) conducted a nationwide survey that did not find any cases of methemoglobinemia from wells that had lower NO_3-N concentrations than 10 mg L^{-1} (Walter 1951). This finding eventually led to today's drinking water standard. The Comley (1945) finding that high NO_3-N concentrations in drinking water can lead to methemoglobinemia when paired with health dispositions has been substantially validated (Murray and Christie 1993; Avery 1999; Knobeloch et al. 2000; Fewtrell 2014).

High drinking water NO_3-N concentrations can also indirectly increase cancer risk. When ingested, NO_3^- can react with different body functional groups to form carcinogenic N-nitroso compounds (Bouchard et al. 1992; Cantor 1997; Ward et al. 2005). Research has shown some association between long term drinking water NO_3-N concentration exposure and both non-Hodgkin's lymphoma (Ward et al. 1996) and colon cancer (De Roos et al. 2003).

Nitrate Leaching and Environmental Impacts

The movement of NO_3^- into groundwater can lead to detrimental environmental impacts. Nitrate leaching into groundwater can eventually enter surface water bodies. High N concentrations can cause eutrophication in surface water bodies (Galloway et al. 2003; Joyce 2000) and when advanced, eutrophication can lead to hypoxic zones, or areas with depleted oxygen concentrations (Nijboer et al. 2003; Wu and Tanaka 2005). Eutrophication is characterized by unsustainable algal growth and can limit recreational activities as well as lead to fish kills (Sparks 2003). In the United States alone, half of the coastal estuaries are plagued by periodic hypoxia (Joyce 2000).

A study assessing the inputs and outputs of nutrients through the Mississippi-Atchafalaya River Basin (MARB; Goolsby et al. 1999), estimated that approximately 90% of N entering the Gulf of Mexico is from non-point sources. Of that total, the combination of fertilizer N and soil N accounted for 50±9 % within outputs from the MARB as a whole, which totaled 1.6 x 10^9 kg yr^{-1} during 1980-1996 and were three-fold higher than in 1955-1970 (Goolsby et al. 1999; Goolsby and Battaglin 2001). For the Missouri River Basin, cultivated agriculture accounts for 59 % of N in local waters with strong potential for beneficial effects of conservation practices (Santhi et al. 2014). Though agriculture is not the only source of N loading in surface waterbodies, it is clear that agricultural producers' role in improving this issue is crucial.

Nitrate Leaching and Economic Impacts

Finally, NO_3^- leaching can have negative impacts on net revenue for agricultural producers. The over-application of N fertilizer relative to crop needs is an unnecessary cost for agricultural producers if a majority of the fertilizer is leached out of the root zone (Janzen et al. 2003; Miao et al. 2014). Various fertilizer N recovery (FNR) studies in Canada have shown that wheat is able to use up to 50 % of the fertilizer applied (Janzen et al. 1990; Tran and Tremblay 2000). Poor FNR may not always occur in situations when NO_3^- is leached. Multiple research studies in Europe show that when fertilizer N is applied at the optimum rate and time, cereal crops are able to use >90 % of it (Macdonald et al. 1989; Johnston 1995; Jenkinson 2001). Macdonald et al. (1989) found no significant difference between soil nitrate concentrations measured post winter wheat harvest on plots with and without fertilizer application suggesting that post-harvest NO_3^- levels were driven more by mineralized N than the newly added fertilizer N. Management practices that focus on the utilization of both mineralized N and fertilizer N could potentially help to reduce NO_3^- leaching losses and save the producer money on fertilizer application.

The need for high levels of efficient food production is crucial, but nitrate contamination of ground and surface waters might not have to be such a large negative externality. Continued research toward management practices that decrease NO_3^- leaching and sustain food production is imperative for addressing this issue. An understanding of why NO_3^- leaching is prevelant in agricultural regions can help to provide an objective perspective when selecting management practices.

The Role of Agriculture in Leaching

To properly address how agriculture contributes to NO_3^- leaching, an understanding of why agricultural regions can become susceptible to this problem is imperative. As a natural ecosystem is converted to agricultural use, the possibility for N loss increases substantially (Turner and Rabalais 2003) because natural ecosystems generally conserve N (Bouchard et al. 1992). In a Colorado dryland agroecosystem, soil NO_3-N levels below a depth of 1.8 m were greater under cultivated than native soils at two of three sites, with a mean value of 72 and 10 kg N ha^{-1}, respectively (Evans et al. 1994). Furthermore, tilling facilitates organic matter mineralization by breaking apart soil aggregates and exposing more organic matter to decomposition (Peterson et al. 1998), releasing inorganic N to the soil profile. Cultivated crop residue can contribute to NO_3^- leaching because it decomposes at a faster rate than soil organic matter (Kirchmann et al. 2002).

With increased dependence on N fertilizer in agriculture, more N can be lost through hydrologic pathways if N uptake does not increase at a similar rate (Galloway et al. 2003; Huang et al. 1996). Furthermore, the combination of agricultural practice and natural factors such as shallow and/or coarse soils, shallow groundwater, and high precipitation may increase the probability of NO_3^- leaching out of the root zone in regions that exhibit these characteristics (Kurunc et al. 2011; Xin-Qiang et al. 2011; Jabloun et al. 2015).

Groundwater Nitrate Contamination in Dryland Agroecosystems

The agricultural land of the northern Great Plains (NGP) region of North America encompasses 41% of the region's ~125 million-ha area (Padbury et al. 2007). Much of this agricultural land is farmed using dryland practices, which are characterized by the absence of irrigation (Thamke and Nimick 1998), low amounts of annual precipitation (300-500 mm, Hansen et al. 2012; Peterson et al. 1993), and use of summer fallow (Hansen et al. 2012). Drought-tolerant wheat (*Triticum aestivum L.*) is the most common crop grown in the region (Padbury et al. 2007; Tanaka et al. 2010; Hansen et al. 2012) and is planted in either the spring or fall.

A major farming practice in the NGP that can contribute to NO_3^- leaching is summer fallow (Bauder et al. 1991; Bauder et al. 1993; Nimick and Thamke 1998). Summer fallow is a method of soil moisture conservation during which no crop is planted on a field during the growing season to secure crop yields for the following growing season. From a soil moisture conservation perspective, research has shown no significant differences between the amount of soil moisture stored in a fallow year at late spring and early fall, implying that the fallowing may not improve water storage during this time period due to evaporation (Farahani et al. 1998). Moreover, this practice may contribute to NO_3^- leaching because without a crop utilizing NO_3^- and water during a year's wet periods, NO_3^- is available to move out of the soil profile through percolation into groundwater. Although summer fallowed land in the NGP has declined from 17 to 4 million ha in the past 40 years (Tanaka et al. 2010), the practice still has a substantial foothold in the region.

One area of the NGP where NO_3-N concentrations above the drinking water standard have been frequently observed is in the Judith River Watershed (JRW, Figure 1.2) of central Montana. The 720,000-ha JRW encompasses much of Fergus and Judith Basin counties and the Judith River that flows through the watershed is a tributary of the Missouri River. In 2012, the area cultivated in winter wheat for both counties was 79,157 ha and 24,929 ha for spring wheat (National Agricultural Statistics Service 2012). In the same year, 40,263 ha of cropland were in summer fallow. With the total acreage of summer fallow approximately 40 % of the total wheat acreage, summer fallow is still a widely used practice in the JRW.

In the early 1990s, a private well testing program found that 22 out of 66 wells in Judith Basin County and 23 out of 145 wells tested in Fergus County were above the EPA drinking water standard (Bauder et al. 1993). In 2009, a Montana Department of Agriculture (MDA) monitoring program found that 9 out of 11 wells on agricultural sites in the JRW had nitrate concentrations above 10 mg L^{-1} (Schmidt and Mulder 2010). An MDA monitoring well (M-1) located in the center of the watershed revealed that between these two study years (1993 and 2009), NO_3-N concentrations increased from ~10 mg L^{-1}

to ~ 23 mg L^{-1} (Schmidt and Mulder 2010). These data suggest that groundwater NO_3-N concentrations have risen since at least the 1990s. The JRW has a semi-arid climate and irrigation is minimal; therefore, it is somewhat surprising that some NO_3-N concentrations are above the drinking water standard in groundwater.

One possible explanation for the high NO_3-N concentrations in JRW drinking water is the interaction of land use with the underlying stratigraphy (Bauder et al. 1991). Within the watershed, large gravel deposits, likely formed by alluvial and glacial outwash influence, overlay marine shales approximately 10 to 30 m from the soil surface (Montana Bureau of Mines and Geology 2007). A large percentage of cultivation occurs on soils with ~30 to 100 cm of fines (Natural Resources Conservation Service 1967) with low percentages of coarse material, above gravel contacts containing >50% coarse material. This stratigraphy creates a situation where shallow aquifers are substantially influenced by land use. The texture of fines in the areas of cultivation are predominantly loams and clay loams (Natural Resources Conservation Service 2015b), which on average reach field capacity at ~0.32 cm cm^{-1} and wilting point at ~0.21 cm cm^{-1} to 48 cm soil depth (Natural Resources Conservation Service 2015a). At the average depth of fines (68 cm), only ~7.5 cm of precipitation per water year is needed to fill the soil profile from wilting point. The long term precipitation average for the JRW (1981-2010) is approximately 40 cm per water year (Western Regional Climate Center 2013), meaning that theoretically the soil profiles under cultivation likely reach field capacity between harvest and early spring. Based on this analysis, the use of summer fallow as a water conservation tool in this region may be unnecessary. Furthermore, the locations of many of the wells with high NO_3-N levels are on soils with shallow gravel contacts. The combination of stratigraphy, soil formation, and agricultural practices creates an ideal situation for NO_3^- leaching in this region.

Alternative Management Practices for Dryland Agriculture

Research on NO_3^- leaching in dryland agricultural systems has shown that controlling the amount of NO_3^- that leaches into groundwater could be achieved through a combination of both improved crop management and fertilizer use (Bauder et al. 1993; Westfall et al. 1996; Thamke and Nimick 1998; Campbell et al. 2006). Continuous cropping paired with no-till farming practices has been shown to decrease N losses from agroecosystems compared to crop-fallow practices by improving N use efficiency (NUE; Campbell et al. 1984; Westfall et al. 1996). For example, in the Canadian prairies, continuous wheat decreased NO_3^- leaching when compared to a fallow-wheat system (Campbell et al. 2006).

Another method for decreasing NO_3^- leaching and increasing NUE is to manage fertilizer application rate and timing. For example, a study in Colorado found that less N fertilizer needed to be applied in the spring to match fall fertilized wheat yields (Vaughn et al. 1990). Other possible strategies that have potential for increasing NUE include split application (Gravelle et al. 1988; Mascagni and Sabbe 1991; Woolfolk et al. 2002) and controlled-release fertilizers (Mikkelsen et al. 1994; Shoji et al. 2001). Providing fertilizer N to the plant when it is needed will help to reduce N losses and excess costs for over-application (Huang et al. 1998).

Though the use of split application and controlled-release fertilizer have shown promising results in improving NUE in other areas of the United States, results from limited research in the NGP have been less conclusive. For example, there was no consistent advantage to using split application or controlled-release fertilizers over a single application in multiple ecoregions in Canada (Grant et al. 2012). However, a study in Oklahoma found that wheat grain N uptake was significantly higher using split application during three of the five study years (Mohammed et al. 2013). Furthermore, a study in Colorado found less N in the 80-110 cm soil depth using controlled-release fertilizer than urea fertilizer on irrigated barley, indicating the possibility for less downward movement of N (Shoji et al. 2001). The discrepancy in research between the NGP and other locations throughout the United States could be influenced by climate and points to the need for research on these practices in the NGP.

Despite research on the effects of these alternative management practices on NUE, very little or no work has been done on the effects of these practices on nitrate leaching rates, especially in the NGP. As a result, there is a need to quantify this amount and to target the most effective practices. When considering alternative management practices that could decrease the amount of NO_3^- leached to groundwater, it is equally important to focus on practices that could be economically beneficial. If practices do not provide an economic incentive for the agricultural producer, the likelihood of adoption will decrease (Nowak 1992).

Project Objectives

The fact that groundwater NO_3-N concentrations in the JRW are high and appear to be increasing invokes a sense of urgency and importance to target management practices that may help alleviate the issue. It is important to note that our study on management practices is one component of a broader study being conducted in the JRW. Nitrate sources (Miller 2013), landform scale leaching interactions (Sigler et al. 2013; Ewing et al. 2014), and the effectiveness of community-researcher collaboration (Sigler et al. 2014; Jackson-Smith unpublished 2015) are other components that have been studied.

The goal of this work was to compare effects of alternative management practices (AMPs) and grower standard practices (GSPs) on nitrate leaching amounts and net revenue, on, dryland grain, commercial farms in the JRW. Chapter 2 compares effects of three contrasting AMPs and GSPs on grain yield and protein, and net revenue. Chapter 3 compares effects of the same three AMPs and GSPs on nitrate leaching losses.

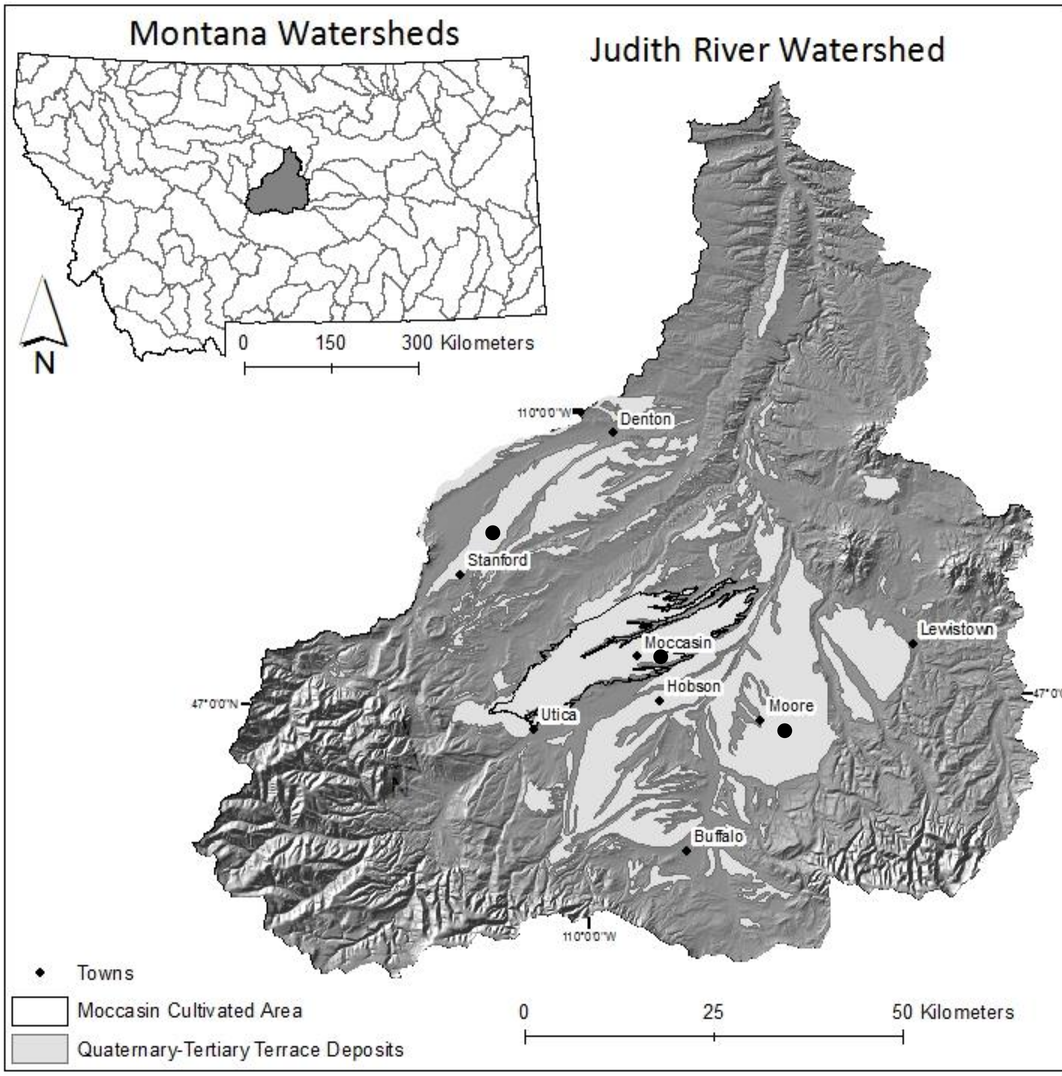

Figure 1.1. Map of the Judith River Watershed and its location within the state of Montana (Miller 2013, Used with permission). Light grey layer shows the cultivated areas within the watershed. Large black dots indicate approximate field site locations; field A near Stanford, field B near Moccasin, and field C near Moore. (Data Sources: Montana Natural Resources Information Systems, USGS National Elevation Dataset)

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CHAPTER TWO

MANAGEMENT PRACTICES TO REDUCE NITRATE LEACHING FROM SHALLOW SOILS: GRAIN YIELD AND PROTEIN, AND NET REVENUE RESPONSES

Abstract

In the Judith River Watershed (JRW) of central Montana, groundwater nitrate (NO_3^-) contamination from NO_3^- leaching has become a major concern for agricultural producers and watershed stakeholders. We conducted a 2-yr study on commercial farms that compared three practices (pea, controlled release urea, split application) with the potential to reduce NO_3^- leaching with standard practices (fallow, conventional urea, single broadcast urea) on wheat grain yield and protein, and net revenue. Despite lower yield and grain protein in year 1 of the study compared to fallow, pea in rotation resulted in \$83 ha^{-1} of increased net revenue in year 2.

Introduction

Nitrate (NO_3^-) leaching out of the root zone into groundwater and surface water has negative impacts on human health (Knobeloch et al. 2000), the environment (Goolsby et al. 1999), and potentially, agricultural net revenue (Johnson et al. 1991; Huang et al. 1996). Groundwater NO_3-N concentrations above the drinking water standard (10 mg L^{-1}) have been found in the Judith River Watershed (JRW) of central Montana in the semiarid northern Great Plains (NGP) since at least the early 1990s (Bauder et al. 1991; Bauder et al. 1993; Schmidt and Mulder 2010). There is evidence that the interaction between agricultural practices and shallow unconfined aquifers, overlaid with gravelly soils, is contributing to the issue (Schmidt and Mulder 2010). Further research on agricultural alternative management practices (AMP) that could reduce leaching is a critical first step to provide producers and policy makers with sound recommendations. A focus on AMPs that promote both environmental and economic sustainability will greatly increase the likelihood of adoption (Nowak 1992; Gabriel et al. 2013). Specifically, a producer's awareness of the environmental benefits, economic benefits, and practicality of a particular practice have been found to be the most important adoption factors (Reimer et al. 2012).

Cereal cropping rotations within the JRW are typically wheat-fallow or wheat-spring grain-fallow with urea fertilizer application in the spring. It has been suggested that summer fallow, where a crop is not grown for a growing season, could increase the amount of NO_3^- that leaches within semiarid regions (Bauder et al. 1993; Campbell et al. 2006). Agricultural practices that have shown potential in reducing NO_3^- leaching are cover crops (Tonitto et al. 2006) and reducing the extent of fallow with diversified cropping systems (Westfall et al. 1996). Furthermore, with dependence on N fertilizer in this region, research should focus on the influence of fertilizer N application as it pertains to NO_3^- leaching. Fertilizer alternative management practices such as, controlled-release fertilizers (Mikkelsen et al. 1994; Shoji et al. 2001), split fertilizer applications (Mascagni and Sabbe 1991; Mohammed et al. 2013) and fertilizer application timing methods (Vaughn et al. 1990) have shown potential in reducing NO_3^- leaching.

Replacing summer fallow with a crop has potential to reduce NO_3^- leaching and increase net revenue in the NGP (Tanaka et al. 2007). Within Montana, intensive cropping with legumes has shown promise to increase both soil quality (O'Dea et al. 2015) and net revenue (Miller et al. 2015). In southwest Montana, economic return was higher for a pea-wheat rotation than for fallow-wheat (Miller et al. 2015); however, increasing cropping intensity could negatively impact the subsequent wheat crop due to decreased soil water (Carr et al. 2001) and soil NO_3^- (Soon and Clayton 2003). Despite some success shown with crop rotational practices, it is a worthwhile endeavor to explore alternative methods in reducing NO_3^- leaching and increasing net revenue.

Controlled-release and split-application fertilizer practices have potential to reduce NO_3^- leaching from agricultural fields without reducing soil water and NO_3^-. Both have been found to increase nitrogen use efficiency (NUE) in cereal production in Oklahoma and Colorado (Woolfolk et al. 2002; Shoji et al. 2001). Limited research has been conducted on the effects of these practices on grain yield, quality, and net revenue within the NGP region. A two-year study conducted in western Canada found that controlled-release urea (CRU) and split-application had lower or equal economic returns for wheat than conventional urea (CU), partly due to higher management costs (Khakbazan et al. 2013). Another study in western Canada found that spring-applied CRU and split application provided higher small grain yields than broadcast urea (Malhi et al. 2010). Given these mixed results, the lack of published studies on these practices elsewhere in the NGP, and the importance of economics for AMP adoption, additional research on the effects of these AMPs is imperative.

In the JRW study, the 2012 farmer community survey (Jackson-Smith unpublished 2012), along with ongoing collaboration with participating producers and stakeholders, led to selection of three alternative practices for focused study: replacing fallow with an annual legume, controlled-release fertilizer, and split fertilizer application. Producers thought that these three practices maintained practicality within already established operations.

Our objectives were to determine the effects of AMPs on grain yield and protein, and net revenue. Three AMPs (pea, CRU, split application) were compared to grower standard practices (GSP; summer fallow, CU, single broadcast urea [SBU]). Local agricultural producers and stakeholders played a pivotal role in selecting AMPs that have a high chance for adoption, making this project distinct in its community involvement aspect.

Materials and Methods

Study Design

The study was conducted on three dryland commercial farms located in the JRW from 2012 to 2014 (Figure 1.1, Chapter 1; Table 2.1). The farms were chosen based on 1) the grower's willingness to participate in the study and 2) location, to make certain that major landform types within the JRW were represented. It is important to note the possibility of field selection bias and that growers willing to participate may not be a representative sample of the entire JRW. Within each landform, field selection criteria included 1) presence of shallow gravel contacts within soils (30-100 cm), 2) only one soil series or two-component complex per field, and 3) use of summer fallow or barley in the first year of the study (2012). These sites were located on commercial grain farms that had been under no-till management for 10-15 yrs. Field A was located ~6.5 km northeast of Stanford, field B was ~4 km east of Moccasin and field C was ~7 km east of Moore (refer to watershed map, Figure 1.1, Chapter 1). Air temperature and precipitation over the course of the study were recorded on site using automated gauges (HOBO® data loggers, Onset®, Bourne, MA) between Apr and Nov. Other temperature and precipitation data were obtained from the Moccasin Agrimet data collection station (United States Department of Interior, station MWSM), located 7 to 20 km from the study fields.

At each farm, full scale operational fields were divided into subfields so that multiple management practices could be compared on each field. Each management interface separated each GSP subfield from its respective AMP subfield (selected randomly). Pea was compared to fallow, CRU to CU, and split application to SBU. Subfields ranged from 10 to 20 ha. In total, there were eight treatment interfaces and each treatment was duplicated at a different farm (Table 2.2). Between years, fertilizer treatments were replicated on the same fields (i.e. 2013 and 2014) and the pea treatment was replicated on different fields because winter wheat followed pea or fallow in rotation. All three AMPs were tested at field A, only pea at field B, and only fertilizer treatments at field C (Table 2.2).

Each treatment interface was located ~100 m from the north and south edges of the field. The 100-m "buffer" helped minimize potential agricultural machinery disturbance from turning and/or multiple seeding/chemical passes. Between the buffer zones, ten locations were established per subfield, equidistant along the interface (20 sample locations per interface). Sample locations were approximately 8.5 m from the interface line to avoid treatment edge effects across the interface (Figure 2.1).

Details of fertilizer application are shown in Appendix A (Tables A.1-A.4). In the 2013 treatment year (i.e. late summer 2012 to late summer 2013), growers A, B, and C applied 100, ~70, and ~80 kg N ha^{-1}, respectively to winter wheat (WW; Table A.1-A.4). Grower A and C applied ~50 and ~65 kg N ha^{-1}, respectively, of Environmentally Smart Nitrogen (ESN; Agrium Inc., Calgary, AB) as the form of CRU with their seed. Grower A applied ~30 kg N ha^{-1} broadcast urea, in the spring, on the CRU field (Table A.3). For the split applications, ~43 kg N ha^{-1} was broadcast-applied as urea in early Apr, and the remainder (~30 kg N ha^{-1}) was applied as urea ammonium nitrate (UAN) in late May/early June using flat fans (Table A.4). In the 2014 treatment year (i.e. late summer 2013 to late summer 2014), grower A applied approximately 90 kg N ha^{-1} to spring wheat (SW) fields, except on A3 where the split application was not completed due to very dry conditions (Table A.4). Grower A and B applied ~100 and ~70 kg N ha^{-1}, respectively to WW, and grower C applied ~92 kg N ha^{-1} to SW (Table A.1-A.4). Grower A and C applied 28 kg N ha^{-1} of ESN with SW seed and 39 and 49 kg N ha^{-1}, respectively, as broadcast urea later in the spring (Table A.3). For the second application at field C, 28 kg N ha^{-1} of UAN was applied with streamer bars (Table A.4).

Due to the nature of experimenting on commercial farms, there was some variation of these practices among individual producers and locations. During the 2013 growing season, there was poor pea emergence on the seeder pass that started at the B3-B4 interface due to a mechanical issue. To correct this issue, sample locations were moved into the fully emergent stand of pea directly east of the original locations. In addition, due to a particularly dry fall in 2012, grower C was unable to seed all fields on the same date because of seeder shank breakage. Field C1 was seeded on 29 Sept 2012 and C2 and C3 on 17 Oct 2012. However, stand counts conducted on 25 May 2013 indicated that there were no stand differences between the C1 and C2 subfields.

Sampling Methods

Sampling to characterize subfield soils was conducted prior to initiation of treatments, between Apr and Aug 2012. Two soil core samples were taken at each sample site with either a hand core (2-cm diam.) or custom-made truck mounted hydraulic probe (3-cm diam.) Both subsamples were taken within 1 m of each other; to a depth of 15 cm. Samples were placed into plastic-lined paper bags and transported in coolers on ice.

Each year, within ten days prior to commercial harvest, a biomass sample was collected at the ten locations along treatment interfaces. Biomass sample area was ~0.6 m^2 in 2012, ~0.5 m^2 in 2013, and ~0.7 m^2 in 2014, encompassing one to two 1-m rows. Samples were collected using a rice sickle to cut the stems at soil contact. Subsequent year sample locations were located ~1 m north of the previous year. All biomass and soil sampling locations were logged with a GPS unit (Trimble Navigation Ltd., Sunnyvale, CA).

Laboratory Procedures

Soil samples were weighed moist, oven-dried at 40^{o} C for 7-14 d, and re-weighed to determine gravimetric water content (GWC) and dry bulk density based on core dimensions. Volumetric water content (VWC) was determined for each sample by multiplying GWC by dry bulk density. Each soil sample was ground with a mortar and pestle and sieved through a 2-mm sieve. Nitrate was extracted from the fines with 1 M KCl (Bundy et al 1994) and extracts analyzed with cadmium reduction using Lachat Flow Injection Analysis (QuikChem Method 12-107-04-1-B, Lachat Instruments Inc., Milwaukee, WI). Each surface soil (0 to 15 cm) core was analyzed for pH (1:1 soil:water), electrical conductivity (EC; 1:1 soil:water), Olsen phosphorus ($NaHCO_3$ extraction, colorimetric determination), potassium (ammonium acetate pH 7), total-N (Elementar Vario Max Total-N by combustion analyzer), soil organic carbon (SOC; Walkley Black), and soil texture (Hydrometer method ASTM 422) by Agvise Laboratories.

Wheat, pea, and weed biomass samples were dried at 40^{o} C for 7-14 d and then weighed. Wheat samples were threshed with a Vogel Stationary Grain Thresher (Almaco, Nevada, IA). Pea samples were manually threshed and sieved to collect the grain. All grain samples were cleaned with a grain blower. All grain and stubble were finely ground (<0.5 mm) in a Udy mill (Cyclone Lab sample mill, Udy Corporation, Fort Collins, CO). Total-N and C were analyzed from a 0.1-g subsample using an automatic combustion analyzer (TruSpec CN, LECO Corporation, St. Joseph, MI). Grain protein was calculated by multiplying grain total-N by 5.7 (Jones 1941; Tkachuk 1969).

Economic Data Collection

Enterprise budgets were constructed to calculate the net revenue generated for each practice. Seeding, pesticide application, and fertilizer application costs were obtained through interviews with the three growers involved in the project (Appendix A; Table A.5 – A.6). Machinery costs were obtained through North Dakota State University Extension Service custom rates and included items such as fuel, machinery repair, and usage costs for applications, harvesting, and grain hauling (Aakre 2014; Appendix A; Table A.7). Property, land rental, and insurance costs were not included in this analysis because it was assumed that these costs would not change over the course of the study. For gross revenue, wheat prices were obtained from the Montana Wheat & Barley Committee by averaging July 15 to Sept 15 Great Falls and Billings' grain elevator price data (Montana Wheat & Barley Committee 2014). Protein premiums and discounts were based on averaged grain elevator schedules in Great Falls, Moccasin, and three Billings locations (Bekkerman unpublished 2015). Yield and protein data from the ten locations within each subfield were used with the grain elevator data to estimate gross revenue. Pea prices (\$250 to \$300 Mg^{-1}) were determined through grower interviews.

Calculations. Net revenue (\$ ha^{-1}) at each treatment was derived using Equation 2.1 for each study year.

$$Eq(2.1)\colon Net\ Revenue(\$\ ha^{-1}) = Gross\ Revenue(Crop\ Price \times Yield) - Costs\ (Seeding + Pesticides + Fertilizer + Total\ Machinery\ Cost)$$

Fertilizer treatments included only data for the year that the treatment was administered. For pea and fallow treatments, net revenue from either pea or fallow and wheat were included. Based on the protein discount/premium averages collected, three models were constructed that calculated protein discounts and premiums (adjustments) informed by the protein content (g kg^{-1}) in each sample (Equations 2.2 - 2.4).

$$Eq(2.2): WW\ Protein\ Adjustment\ 2013(\$\ Mg^{-1}) = 0.22(Protein\ Level) - 25.35$$

$$Eq(2.3): WW\ Protein\ Adjustment\ 2014(\$\ Mg^{-1}) = 0.24(Protein\ Level) - 29.03$$

$$Eq(2.4): SW\ Protein\ Adjustment\ 2014(\$\ Mg^{-1}) = 2.08(Protein\ Level) - 299.46$$

A range of protein levels (WW = 100-150 g kg^{-1}; SW = 120-170 g kg^{-1}) were used to determine the adjustments. The protein adjustments were capped at 160 g kg^{-1} for both WW and SW based on grain elevator scheduling. These adjustments were then added to the base wheat price for 2013 (WW = \$243 Mg^{-1}) and 2014 (WW = \$188 Mg^{-1}; SW = \$193 Mg^{-1}).

Statistical Analysis

The statistical analysis was conducted with R statistical software (The R Foundation for Statistical Computing, Vienna, Austria, 2013). A hierarchal mixed effects model using the *lmer* function from the *lme4* package was used to perform the analysis (Gelman and Hill 2007; Bates et al. 2015). This model was chosen due to the ability to account for variability at field and location, fit as random effects and due to the assumption that results from our study could differ among multiple other studies (Borenstein et al. 2010). Our interest lay in the explanatory variables, not the fields and locations themselves; to accomplish this, the model nests each of the ten sample location pairs within each subfield pair.

Each year, separate models were constructed for yield, protein, and net revenue as dependent variables. Explanatory variables of interest were treatment, grower, growing season precipitation, N fertilizer amount, total-N, SOC, pH, depth to gravel contact, coarse fraction, Olsen P, EC, and crop type (only in 2014). Careful consideration was taken toward the inclusion of explanatory variables in the model. Due to low numbers of observations (n=3-5) for grower, growing season precipitation, and N fertilizer amount, near to perfect collinearity became an issue between them. To correct this, a categorical management and climate (Mgmt&Clim) variable was created that accounted for these variables. Plot matrices were used to locate any highly correlated variables that remained. Any correlation, r, value above 0.5 (or < -0.5) was assumed to be highly correlated and in this case only variables of greater interest to the study were retained. Variables excluded in this analysis were sand, SOC, and pH because of their relationship with clay, total-N, and EC respectively. In addition, post model collinearity diagnostics were performed on each model by checking variance inflation factor values using the *car* package. With no further evidence of collinearity, the final models included treatment, Mgmt&Clim., total-N, depth to rock, clay, Olsen P, coarse fraction, EC, and crop (only in 2014) as the explanatory variables.

Confidence intervals were calculated for explanatory variables by using the adjusted available degrees of freedom based on the number of paired locations and fields. Since treatment was administered at field level, the correct available sample size for testing treatment effects was six, while the available sample size for testing the effect of explanatory variables was 120. From the model output in R, coefficient estimates and associated standard errors were used to calculate strength of evidence (p-value) for treatment and soil characteristic effects.

Results and Discussion

Precipitation Context

Winter wheat seeding for the first year of the study (2013) followed a very dry 2012 water year (Oct 2011 to Sept 2012) in central Montana. In the 2013 treatment year, field A received similar growing season (May-July) precipitation (187 mm) to the long term average (LTA; 181 mm), whereas fields B and C received 241 and 343 mm, respectively (Table 2.3). In the 2014 treatment year, all fields received below average precipitation during the growing season and field A received ~40 % less than the other fields.

Grain Yield

In the 2013 treatment year, winter wheat yielded less when grown after pea than after fallow (Table 2.4). This result is consistent with research in western Canada and North Dakota that found continuous cropping decreased wheat yields compared to fallow (Campbell et al. 1983; Carr et al. 2001). In the 2014 treatment year, no differences in winter wheat yield after pea and fallow were observed. The difference in result between years is consistent with two other Montana studies that found wheat after pea yielded lower, equal, or higher than wheat after fallow, with lower yields observed in drought conditions (Miller and Holmes 2005; Miller et al. 2006). Precipitation amounts and timing likely caused these differences between years. In the 2012 water year, the JRW received approximately 150 mm less rainfall than the LTA, whereas in 2013, precipitation was within 40 mm of the LTA. Notably, Sept 2013 precipitation averaged 60 mm on the two pea-fallow fields, which was far greater than 2012 precipitation amounts of only 0 to 1 mm (Table 2.3). Winter wheat seeded after pea in fall 2012 would have been substantially disadvantaged because of lower soil moisture.

For both treatment years, no wheat grain yield differences were observed for CRU-CU and the split application-SBU comparisons (Table 2.4). This result agrees with work in Canada showing increased wheat yields are not consistently observed with CRU (Malhi et al. 2010; Khakbazan et al. 2013), but is not consistent with other results in Canada where higher yields were attained with split applied fertilizer than with CRU or CU (Malhi et al. 2010).

One possible reason for the lack of an effect with these two fertilizer treatments in our study, but a large effect in the 2013 pea-fallow comparison, is that both water and N were likely limiting growth more after pea than in the fertilizer treatments. The two fertilizer AMP and respective GSP subfields were not different in soil N and soil moisture at the start of each treatment year (Aug 2012 and 2013). In Montana, winter wheat with less than 125 g kg^{-1} protein content indicates low N limited yield (Engel et al. unpublished 2005) and wheat grain protein was under this threshold after pea but not after fallow, suggesting N uptake by pea was not sufficiently counteracted by pea residue mineralization. The average absolute difference between model estimated and measured means for each crop treatment across years (n=12) was only 0.02 Mg ha^{-1}, substantially less than the mean yield of 3.4 Mg ha^{-1}, indicating the model was working well at the combined subfield level (Appendix A; Table A.8).

For both study years, greater depth to rock was consistently correlated with more wheat grain yield (Table 2.5), not surprising given that wheat crops on deeper soils likely had more access to available N and soil moisture. In the 2013 treatment year, EC was positively correlated with yield (Table 2.5). In general, an increase in salinity can stunt plant growth and decrease wheat yields above 6 dS m^{-1} (Maas and Grattan 1999; Steppuhn et al. 2001), yet EC concentrations on our study fields were approximately 0.3 to 0.5 dS m^{-1}, well below the threshold. The positive relationship between EC and yield could be attributed to a groundwater influence on field A. Mottling and/or visible shallow groundwater was observed within 90 cm of the soil surface from 16 of 32 field A soil pits. Consequently, some of these locations exhibited visual crop greenness longer into the growing season. Electrical conductivity has also been found to be correlated with clay content (Corwin and Lesch 2005; McCutcheon et al. 2006) so could indirectly be related to water holding capacity. In 2014, soil coarse fraction was negatively related with yield, likely due to lower water holding capacity in coarser soils. This is consistent with research in Canada where soils with lower water holding capacity facilitated lower spring wheat yield (He et al. 2013). Finally, in the 2014 treatment year, grower management and climate on field B and C was positively related to yield, likely because fields B and C received 40 % more 2014 growing season (May-July) precipitation than field A (Table 2.3). Also, spring wheat was seeded on field A in late May, compared to early May on field C, partially contributing to lower yields on field A (~1.4 Mg ha^{-1}) than field C (2.5 Mg ha^{-1}).

Grain Protein

In 2013, winter wheat grain protein was lower after pea than fallow (Table 2.4). This result is not consistent with other research where lower soil water availability in general was found to increase protein (Campbell et al. 1997; Selles et al. 2006). Higher grain protein following summer fallow is likely due to post-harvest soil NO_3-N concentrations which on average were ~43 kg N ha^{-1} less after pea than fallow. Grain protein can be lower when there is limited access to N at flowering stage (Terman et al. 1969) and the higher soil NO_3-N concentration following fallow would have provided a protein advantage for wheat that following spring. However, no difference in winter wheat protein was observed in the 2014 treatment year between pea and fallow treatments. It is possible that higher precipitation amounts in fall 2013 facilitated mineralization of pea residue and increased soil N availability by the winter wheat growing season, boosting grain protein. The 2014 result is consistent with another Montana study where equal to or higher wheat grain protein was observed following pea than fallow (Miller et al. 2006).

Grain protein content was higher with CRU than CU in 2013. No difference was observed in grain protein between these two practices in 2014. The different findings between years are similar to another NGP study that resulted in variable effects on grain protein when using CRU (Grant et al. 2012). The 2013 result could be attributed to a late release of N into the soil profile due to a dry fall and lack of substantial precipitation until

May. There were no protein differences observed between the split application and SBU treatments for both study years. This finding is not consistent with research that demonstrates a pre-flowering (booting to heading stage) foliar application can increase wheat grain protein (Gravelle et al. 1988; Woolfolk et al. 2002). Our foliar applications were at earlier stages (late tillering and stem elongation) and could have been too early to impact protein concentrations. Volatilization losses from a very late foliar application (25 June 2014) when soil temperatures were high could have negated any protein benefit from later application. The average absolute difference between model estimated and measured means for each crop treatment across years (n=12) was only 0.4 g kg^{-1}, substantially less than the mean protein of 129 g kg^{-1}, indicating that the model was working well at the crop treatment level (Appendix A; Table A.8).

Salt content was correlated with less wheat grain protein in 2013 (Table 2.5). This is likely due to a dilution effect from increased grain yield on higher EC soils indirectly decreasing protein. A surprising result was that Olsen phosphorus (P) was positively correlated with higher grain protein in both years of the study (Table 2.5). This result is not consistent with other research that demonstrates either no phosphorus impact on grain protein (Campbell et al. 1996) or a negative impact on protein due to dilution from increased yields (Russell et al. 1958). Our result is likely due to an unexplained interaction between P and another variable. For example, lower wheat yield would have been observed on soils with lower water holding capacity which would have increased grain protein but less P would have been removed from these low yielding soils.

Net Revenue

Pea-wheat produced higher net revenue ($P<0.1$) than fallow-wheat in 2014, but not 2013 (Table 2.6). The equal and higher net revenue results of pea-wheat to fallow-wheat is likely due to revenue earned in the pea year that at least covered costs associated with the practice. Other studies in the NGP have also found economic benefits to growing pea in a wheat rotation (Walburger et al. 2004; Burgess et al. 2012; Miller et al. 2015). The finding that CRU and split application did not increase net revenue over the GSP is consistent with results from a study across five sites in western Canada (Khakbazan et al. 2013). The average absolute difference between estimated and measured means for each crop treatment across years (n=12) was only \$6.20 ha^{-1}, substantially less than the mean net revenue of \$339 ha^{-1} (Appendix A; Table A.8).

In both study years, depth to rock was consistently positively related with higher net revenue (Table 2.5) likely due to higher yield observed on deeper soils. Electrical conductivity content was also positively correlated with net revenue, though only in the 2013 treatment year. Again, the correlation of EC content with higher yields observed in this particular year likely influenced the relationship of EC with gross revenue earned from wheat.

Link to Sustainability of Practices

Environmental and economic benefits are major concerns for growers who are considering new practices. In this study, pea treatment was the only practice that showed benefits in both areas since the practice increased net revenue and decreased NO_3^- leaching in one of the two years (Chapter 3: Results and Discussion). This research suggests that a practice that controls both NO_3^- and water movement through the system is likely more effective at reducing NO_3^- leaching than practices that only affect soil water nitrate concentrations. Furthermore, wheat rotations incorporating legumes have shown soil quality benefits by increasing the supply of N in soil in an eight year study (O'Dea et al. 2015). With the possibility of decreased NO_3^- leaching and increased soil quality, replacing fallow with an annual legume may have long term environmental and economic benefits, which also include reduced fertilizer use and increased net revenue.

Conclusion

This study revealed that in the short term (2 yr), replacing fallow with pea was the most promising AMP for increasing net revenue. Pea grown to grain decreased subsequent wheat grain yield and protein in 2013 and had no effect in 2014, but the net revenue of pea-wheat was either equal to or greater than fallow-wheat. The parallel NO_3^- leaching study (Chapter 3) showed that less NO_3^- leached from the pea-wheat than the fallow-wheat system in one treatment year (2013), demonstrating that the treatment can decrease leaching and increase net revenue. The fertilizer practices neither increased revenue nor decreased leaching (Chapter 3) and thus future research and policy should likely focus on cropping systems that decrease the extent of fallow. The results of this study also indicate the importance of soil texture and hence water and nitrate storage capacity on yield and net revenue. Agricultural producers cannot change soil character/storage potential but this research reveals which practices can be beneficial in this environment.

Though results from this study were from only 2-yrs, the possibility that a fallow replacement could increase net revenue while simultaneously decreasing leaching has extensive implications. Not only could it have a positive influence on the local economy but the quality of life for producers and the lives of those affected by NO_3^- contaminated drinking water could improve. Research that addresses the long-term effects of a fallow replacement is vital for understanding longevity of decreasing NO_3-N concentrations in groundwater and sustainability.

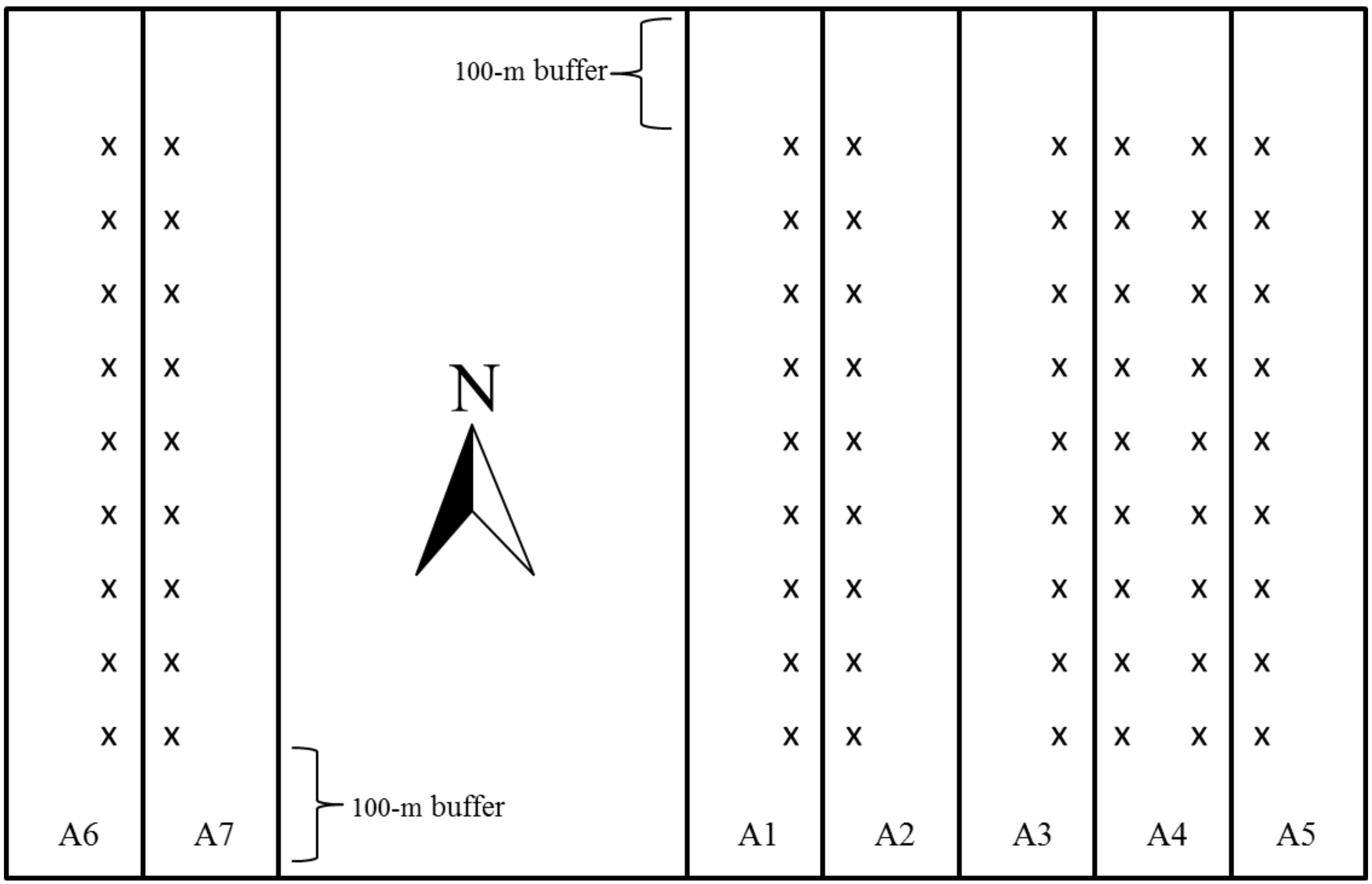

Figure 2.1. Study design on field A. A black “X” represents the sampling location where biomass and soil samples were collected. Each sample location was approximately 8.5 m from the interface line. Buffer zones were located on the southern and northern edges of the field. CRU = Controlled-release urea; CU = Conventional urea; Split = split fertilizer application; SBU = single spring broadcast urea.

Table 2.1. Field descriptions, locations and average soil characteristics for study fields in the Judith River Watershed of central Montana.

Study Site	Field A	Field B	Field C
Proximity	Stanford	Moccasin	Moore
Location	47.2068^{o} N -110.1601^{o} W	47.0523^{o} N -109.8569^{o} W	46.9935^{o} N -109.6128^{o} W
Elevation(m)	1233	1248	1287
Landform	Stanford Terrace	Moccasin Terrace	Moore Fan
Soil Texture	Clay Loam	Clay Loam	Clay Loam
Depth to Gravel (cm)†	70 (28.7)‡	61 (22.1)	71 (33.1)
pH	7.1 (0.4)	7.5 (0.2)	6.5 (0.5)
Soil Organic Carbon (g kg^{-1})	31 (3.9)	40 (3.3)	37 (2.5)
Total-N (g kg^{-1})	1.4 (0.2)	1.8 (0.2)	1.6 (0.1)
Olsen P (mg kg^{-1})	16(5.8)	16 (5.2)	23 (6.0)
Extractable K (mg kg^{-1})	295 (57.9)	307 (50)	302 (27.2)

† Based on soil pits excavated in Summer 2012. Field A = 28 pits; Field B = 17 pts; Field C = 19 pits.

‡ Numbers in parentheses are standard deviations

Table 2.2. Management comparisons and crop rotations on Judith River Watershed NO_3^- leaching study treatment fields for 2011 to 2014.

Comparison‡	Interfaces	2011	2012	2013	2014
		------------------Crop§------------------			
Pea - Fallow(2012)	A5-A4, B1-B2	B	P-F†	WW†	--
Pea - Fallow(2013)	A6-A7, B4-B3	WW	B	P-F†	WW†
CRU - CU	A1-A2, C1-C2	B	F	WW†	SW†
Split App. - SBU	A3-A4, C3-C2	B	F	WW†	SW†

† Tested crop-year

‡ CRU = Controlled Release Urea (ESN®); CU = Conventional Urea; Split App. = Split Fertilizer Application; SBU = Single Broadcast Urea (for specifics see Chapter 2: Materials and Methods)

§ P = Pea Crop; F = Fallow; WW = Winter Wheat; SW = Spring Wheat; B = Barley

Table 2.3. Monthly precipitation amounts and temperature averages by water year for study locations within the Judith River Watershed of central Montana.

	-Agrimet (CARC)‡-			-------Field A‡‡-------			-------Field B‡‡-------			-------Field C‡‡-------			LTA†
	2012	2013	2014	2012	2013	2014	2012	2013	2014	2012	2013	2014	1981-2010
	-------Precipitation (mm)-------												
Oct-Mar††	83	68	103	--	--	--	--	--	--	--	--	--	90
April	70	16	17	--	9	22	--	9	18	--	12	35	36
May	32	82	35	35	103	30	54	106	40	42	174	42	67
June	26	79	55	37	59	34	31	105	53	35	127	63	67
July	16	36	33	52	25	13	48	30	38	39	42	23	47
August§§	17	19	165	13	30	70	13	47	158	20	57	182	48
September	1	78	55	0	43	52	1	78	55	2	87	46	35
TOTAL	245	379	463	137	268	223	147	374	361	138	499	391	391
	-------Temperature (°C)-------												
April	7	3	5	--	4§	6	--	4§	6	--	4§	6	5
May	10	11	10	10	11	11	10§	12	12	10	12	11	10
June	15	14	13	17	15	14	17	16	14	17	16	14	15
July	22	20	20	23	21	22	23	22	22	23	21	21	19
August	20	20	18	21	21	19	21	22	20	21	21	19	19

† The long term average (LTA) data was obtained from the Western Regional Climate Center (WRCC) station in Moccasin, MT

†† Oct - Mar includes data from previous year (e.g. 2012 denotes Oct 2011 to Mar 2012)

‡ Central Agriculture Research Station (CARC) gauge is located 3 km west of Moccasin, MT

‡‡ On field rain gauges were removed in the fall and reinstalled in the spring

§ Temperature average includes fewer days than a full month

§§ Aug precipitation in 2014 was after maturity on field A and after harvest on B and C

Table 2.4. Model predicted winter and spring wheat yield and protein content from the Judith River Watershed based on mixed models from ten ~0.5-0.7 m^2 biomass samples per side of interface.

Comparison[†]	Fields	2013[‡‡]				2014[‡‡]			
		Yield (Mg ha^{-1})		Protein (g kg^{-1})		Yield (Mg ha^{-1})		Protein (g kg^{-1})	
		AMP	GSP	AMP	GSP	AMP	GSP	AMP	GSP
Pea - Fallow	A&B	3.4** (0.4)[§]	4.1 (0.2)	121* (5.1)	132 (4.4)	3.1 (0.3)	3.3 (0.3)	131 (7.2)	144 (7.1)
CRU - CU	A&C[††]	4.0 (0.3)	4.3(0.4)	131** (3.7)	118 (4.1)	1.9 (0.3)	2.0 (0.3)	154 (6.6)	156 (8.0)
Split app. - SBU[§§]	A[‡]&C	4.6 (0.3)	4.3 (0.3)	107 (4.0)	104 (3.3)	2.7 (0.4)	2.6 (0.3)	124 (11)	129 (7.4)

*Significantly different than the GSP at the 0.1 probability level

**Significantly different than the GSP at the 0.05 probability level

† Pea vs fallow treatment is based on wheat crop after year of treatment; CRU = Controlled release urea (ESN); CU = Conventional urea; SBU = single spring broadcast urea

†† In 2014, C2W yield average based on nine wheat samples

‡ In 2014, split application was not completed on A3E and was analyzed as a GSP with lower N fertilizer rate

‡‡ Winter wheat grown on all treatment interfaces in 2013. In 2014, winter wheat was only grown on pea-fallow subfields and spring wheat grown on fertilizer subfields.

§ Numbers in parentheses are standard deviations.

§§ In 2014, split application and SBU averages include only subfields C3 and C2.

Table 2.5. Soil and climate parameters impact (coefficient) on yield, protein, and net revenue for study fields in the Judith River Watershed of central Montana. Soil parameters, except depth to rock, came from 0-15 cm cores.

Parameter†	-----------------------------2013-----------------------------			-------------------------------2014-------------------------------		
	Yield (Mg ha^{-1})	Protein (g kg^{-1})	NR (\$ ha^{-1})	Yield (Mg ha^{-1})	Protein (g kg^{-1})	NR (\$ ha^{-1})
Mgmt&Clim. (B)	-0.24	-7.94	-304** (-603,-4.7) ‡	0.74** (0.3,1.2)	-33.2* (-74,7.9)	281** (99,464)
Mgmt&Clim. (C)	0.01	-30.8* (-69,7.7)	-93.2	1.07** (0.6,1.5)	-32.3** (-65,-0.12)	156** (3.8,307)
Crop (WW)	NI††	NI	NI	1.42* (-0.6,3.4)	-1.8	126
Total-N (g kg^{-1})	4.45	93.4	1844	2.68	58.7	294
Depth to Rock (cm)	0.01** (0.002,0.02)	0.06	2.83** (0.6,5.1)	0.01** (0.008,0.02)	-0.29** (-0.4,-0.17)	1.6** (0.9,2.3)
Clay (g kg^{-1})	-0.01	-0.04	-1.5	-0.01	0.05	0.09
Olsen P (g kg^{-1})	-0.02	0.70** (0.3-1.1)	-3.6	-0.01	0.5** (-0.02,0.93)	-0.8
Coarse Fraction (kg kg^{-1})	-0.59	-2.2	-373	-1.6** (-3.1,-0.12)	-24.7	-96
EC (dS m^{-1})	2.68** (0.65,4.7)	-27.0* (-59,5.4)	704** (173,1235)	0.5	-14.8	-3.3

*Significant at the 0.1 probability level

**Significant at the 0.05 probability level

† Mgmt&Clim. = Categorical variable that includes confounding variables such as weather, seeding date, etc.

†† NI = Not included in the model due to collinearity criterion

‡ 95 % confidence intervals for significant variables are in parentheses

Table 2.6. Model predicted net revenue for management practices in 2013 and 2014 from the Judith River Watershed NO_3^- leaching study.

			----------------2013----------------			----------------2014----------------	
Comparison†	Fields	Crop	Net Revenue ($ ha^{-1}) AMP	Net Revenue ($ ha^{-1}) GSP	Crop	Net Revenue ($ ha^{-1}) AMP	Net Revenue ($ ha^{-1}) GSP
Pea - Fallow	A&B	WW	583 (114)‡	533 (56)	WW	243* (39)	160 (33)
CRU - CU	A&C	WW	618 (83)	677 (107)	SW	-39 (36)	4.1 (36)
Split app. - SBU§	Field A	WW	628 (78)	647 (85)	SW	-24 (56)	38 (38)

* Significantly different than the GSP at the 0.1 probability level

† Pea vs Fallow comparisons include net revenue for two years; CRU = Controlled release urea (ESN); CU = Conventional urea; SBU = Single spring broadcast urea

‡ Numbers in parentheses are standard deviations of twenty measurements (two subfields, ten locations per subfield)

§ In 2014, split application and SBU averages include on subfields C3 and C2

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CHAPTER THREE

NITRATE LEACHING RESPONSES TO AGRICULTURE ALTERNATIVE MANAGEMENT PRACTICES IN A SEMIARID REGION

Abstract

Nitrate (NO_3^-) leaching into groundwater is an issue of concern in the United States and throughout the world. Groundwater nitrate concentrations above the drinking water standard of 10 mg L^{-1} are common in the Judith River Watershed (JRW) of central Montana, and appear to be increasing. Nitrate leaching from agricultural land is a concern for both farmers and stakeholders in the region. Further research on agricultural alternative management practices (AMP) is imperative to address this issue, and research has shown that practices that benefit both the environment and net revenue have the highest likelihood of adoption. We conducted a 2-yr study on commercial farms in the JRW that compared nitrate leaching amounts from three AMPs (pea, controlled release urea [CRU], split application) that have the potential to reduce NO_3^- leaching with grower standard practices ([GSP]; fallow, conventional urea [CU], single broadcast urea [SBU]). Eight management interfaces (six tested per year) were established on three landforms, and each management practice was analyzed in duplicate each year. Biomass and soil was collected at ten evenly spaced sampling locations on each side of the interface. A nitrogen (N) – mass balance approach was used to calculate the amount of NO_3^- leached during each crop year. In 2013, less NO_3^- leached under wheat after pea (20 kg N ha^{-1}) than wheat after fallow (56 kg N ha^{-1}), and the two fertilizer treatments had no effect on

nitrate leaching. In 2014, no AMP leached less than its corresponding GSP. We conclude that replacing pea with fallow has the greatest potential to reduce NO_3^- leaching and future research should focus on practices that decrease deep percolation, such as fallow replacement.

Introduction

Increased groundwater nitrate-nitrogen (NO_3-N) concentrations have been observed in agricultural regions where NO_3^- is lost to leaching (Power and Schepers 1989; Puckett et al. 2001). Nitrate-N concentrations above the Environmental Protection Agency's (EPA) drinking water standard of 10 mg L^{-1} have been linked to adverse health effects such as methemoglobinemia (Avery 1999; Knobeloch et al. 2000; Fewtrell 2004) and cancer (Ward et al. 1996; De Roos et al. 2005). Along with health concerns, NO_3^- leaching can be an economic loss to agricultural producers (Huang et al. 1996; Janzen et al. 2003; Miao et al. 2014). In the semiarid Judith River Watershed (JRW) of central Montana in the northern Great Plains (Figure 1.1, Chapter 1), NO_3-N concentrations in a groundwater monitoring well have increased from 10 to 23 mg L^{-1} between 1993 and 2009 (Schmidt and Mulder 2010; Miller 2013). Much of the small grain agriculture in the JRW is on shallow soils, 30 to 100 cm deep, that are susceptible to leaching. Furthermore, the role of total-N mineralization as a contributor to the issue has not been adequately quantified (Goolsby et al. 1999).

Research that focuses on alternative management practices (AMP) that can reduce NO_3^- leaching and are economically feasible is imperative. The use of cover crops

(Tonitto et al. 2006) and increased cropping intensity/diversity (Westfall et al. 1996) have shown promise in reducing NO_3^- leaching. Fertilizer AMPs such as spring instead of fall application (Vaughn et al. 1990), split fertilizer applications (Mascagni and Sabbe 1991; Mohammed et al. 2013), and controlled release fertilizers (Mikkelsen et al. 1991; Shoji et al. 1990) increase N use efficiency (NUE), which could decrease leaching losses. Replacing fallow with an annual legume, controlled-release fertilizer, and split N application were the three AMPs chosen by our producer and stakeholder advisory groups to be compared with grower standard practices (GSP) in the JRW.

The practice of summer fallow, where crops are not grown for an entire season, has been linked to increased NO_3^- leaching from agricultural fields in the NGP (Bauder et al. 1993; Nimick and Thamke 1998), especially when compared to continuous cropping. In a 37-yr crop rotation study in Canada, a continuous wheat rotation leached four-fold less NO_3^- than a fallow-wheat system (Campbell et al. 2006). Continuous cropping can decrease the downward movement of water and increase NUE (Campbell et al. 1984; Westfall et al. 1996). Increasing crop diversity, such as use of annual legumes in cereal rotations, can increase both soil quality (O'Dea et al. 2015) and net revenue (Zentner et al. 2001; Miller et al. 2015). One possible concern with growing annual legumes is high amounts of mineralizable N in the crop residue which could eventually increase NO_3^- leaching (Thomsen et al. 2001; Campbell et al. 2006). In western Canada, soil NO_3-N concentrations were found to be higher in the spring following pea compared to following wheat (Miller et al. 2003). However, there is little research on the effects of growing annual legumes within a fallow-inclusive no-till cereal rotation on NO_3^- leached quantities in the NGP.

Fertilizer alternative management practices that improve NUE could also decrease the amount of NO_3^- that leaches from agricultural fields. Two such practices are the use of controlled-release urea (CRU; Mikkelsen et al. 1994; Shoji et al. 2001) and split N applications (Mascagni and Sabbe 1991; Mohammed et al. 2013) because they control rate of release (CRU) or timing (split application). In Colorado, CRU had higher fertilizer recovery than CU at 80–110 cm on barley fields (Shoji et al. 2001). In Oklahoma, split-application on winter wheat resulted in higher N uptake than top-dressing, in three out of five study years (Mohammed et al. 2013). Results from these two studies are not consistent with results from a study in Canada where mixed NUE results were found while using fertilizer AMPs (Grant et al. 2012). The discrepancies between previous study results on NUE, combined with the dearth of nitrate leaching studies in dryland portions of the NGP, reveal the need for research in this area.

The objective of this study was to quantify the amount of NO_3-N that leaches out of the root zone as a result of different management practices. Nitrogen budgets were constructed to estimate nitrate leaching for three AMPs (pea, CRU, split application) and three respective GSPs (fallow, conventional urea [CU], single broadcast urea [SBU]).

Materials and Methods

Study Design

An in-depth description of the study design is presented in Chapter 2: Materials and Methods.

Sampling Methods

Post-harvest soil samples were collected as close to harvest as possible (~July 15 to Sept) during each study year (2012-2014) to estimate the amount of N taken up from soil, while minimizing N mineralization between biomass harvest and soil sampling. In summer 2014, soil sampling was conducted only on sampling locations where soil pits were previously excavated as part of a companion study (n=4 per interface; Ewing et al. unpublished 2015). Two 3-cm diameter soil cores were collected ~1 m apart at each sample site with a custom made truck mounted hydraulic probe. Cores were sampled to the deepest possible depth, meaning gravel contact, anomalous rock, or probe depth limit (only 6 of ~1130 cores). Each core was sectioned into 15-cm depths and the total sample depth recorded. In 2013 and 2014, samples were collected ~1 m north of the previous year’s soil sample. All 15-cm soil samples used in the soil characteristic analysis were collected by the same method as described in Chapter 2: Materials and Methods.

Wheat and pea biomass samples were collected as described in Chapter 2: Materials and Methods. *Lactuca serriola L.* (prickly lettuce) was sampled within ~10 m of each sample site on the pea fields to estimate N fixation using natural abundance

(Shearer and Kohl 1986). Both soil and biomass sample sites were logged using a GPS unit (Trimble Navigation Ltd., Sunnyvale, CA).

Laboratory Procedures

Soil samples were analyzed for NO_3-N and characterized as described in Chapter 2: Materials and Methods. Biomass samples were analyzed for total-N as described in Chapter 2: Materials and Methods. Pea grain and stubble, along with weed samples, were analyzed for ^{15}N at UC Davis' Stable Isotope Facility with a PDZ Europa ANCA-GSL elemental analyzer interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (Sercon Ltd. Chesire, UK).

N Balance Method to Calculate Nitrate Leaching

An N-balance method was used to estimate the amount of NO_3-N that leached from each of the treatment subfield locations (Eq. 3.2; Meisinger and Randall 1991; Ju et al. 2006). Available N inputs included N fertilizer, net N mineralization, and N deposition (wet and dry). Available N outputs were defined as volatilization, denitrification and crop N uptake from the soil (not fixed). Soil NO_3^- storage was defined as the difference in soil NO_3-N concentrations between soil sampling periods. Soil ammonium (NH_4^+) concentrations in a companion study at Montana State University were ~3 kg N ha^{-1} during summer 2013 (Ewing, S., unpublished data 2015); a small amount compared to N inputs and outputs (~120 – 154 kg N ha^{-1}) but at times comparable to nitrate N levels; however, for purposes of this study only nitrate, and not NH_4^+, was considered.

Eq 3.2: NO_3-$N\ leached\ (kg\ N\ ha^{-1}) =$
$inputs\ (N\ fertilizer + net\ N\ mineralization + N\ Deposition) -$
$outputs\ (volatilization + denitrification + Crop\ N\ uptake) - (Soil\ NO_3^-{}_{final} -$
$Soil\ NO_3^-{}_{initial})$

Fertilizer N amounts were obtained through grower management interviews and included all fertilizer applications in the treatment year (Table A.1-A.4). Total-N deposition was estimated at 1.6 kg N ha^{-1} using data from the EPA's atmospheric deposition station in Glacier National Park (Elevation 976 m; U. S. Environmental Protection Agency 2014). This value was used for both years of the study.

Total-N uptake included a combination of above ground (AG) measurements and below ground (BG) estimates. The total-N amount in the BG biomass was estimated at 20 % of the aboveground N, using literature wheat root N:shoot N ratios (Andersson et al. 2005). The fraction of N derived from the atmosphere (FNdfa) was calculated using the natural abundance method (Eq. 3.1; Shearer and Kohl 1986).

Eq 3.1: $FNdfa = \frac{\delta^{15}N_o - \delta^{15}N_t}{\delta^{15}N_o - \delta^{15}N_a}$
$\delta^{15}N_o = \delta^{15}N\ from\ non\text{-}N\ fixing\ reference\ plant\ (i.e. prickly\ lettuce)$

$\delta^{15}N_t = \delta^{15}N\ from\ pea$

$\delta^{15}N_a = \delta^{15}N\ from\ pea\ grown\ under\ N\ free\ condition$ (McCauley 2011)

Eq 3.2: $Total\ N\ fixed\ (kg\ N\ ha^{-1}) = FNdfa \times total\ plant\ kg\ N\ ha^{-1}$

The amount of N fixed was determined by multiplying FNdfa by total-N in above ground biomass (Eq 3.2). Once the total amount of N fixed from the atmosphere was calculated it was subtracted from the total-N in the plant to derive the amount of N the pea crop used from the soil.

Broadcast urea volatilization amounts were estimated at 14% of applied N based on average ammonia volatilization losses from eight Central and North Central Montana trials that used a micrometeorological technique (Engel et al. 2011). Five volatilization measurements were from fields located within 130 km of the JRW (Engel et al. 2011) and three were from fields located in the JRW or within 11 km of the watershed (Engel unpublished data 2014). For both years, it was assumed that seed placed starter and CRU fertilizer did not volatilize. It was assumed that 7 % of liquid UAN volatilized in both study years based on a Manitoba, Canada study conducted in late May (Grant et al. 1996).

N Mineralization and Denitrification

A model was designed to estimate net N mineralization – denitrification on each of the treatment subfields based on changes in soil NO_3^- measurements taken intermittently during the study. The literature points to multiple variables that influence N mineralization rates including soil temperature (Stanford et al. 1973; Curtin et al. 2012), soil moisture (Stanford et al. 1972; Paul et al. 2003; Heumann et al. 2011) and soil total-N (Vigil et al. 2002). Other variables discussed in the literature include crop residue C:N ratio (Vigil and Kissel 1995; Booth et al. 2005; Patron et al. 2007), microbial biomass (Booth et al. 2005), soil organic matter (Booth et al. 2005; Heumann et al. 2014) and previous crop (Soon and Arshad 2002).

Seven fallow "subfield-periods" of ~40 to 200 days during 2012-2013 were used to inform the model. Because these were fallow periods, fertilizer input and N-uptake were known to be zero. Based on water budgets, NO_3^- leaching was also assumed to be zero. Specifically, evapotranspiration (estimated measurements from flux tower on Field C) plus the change in soil water storage (0 to 30 cm) was either greater than or equal to the precipitation amount during the seven field periods. Volatilization and N deposition were assumed to be negligible during these time periods because unfertilized fields generally have very low volatilization (Grant et al. 1996) and N deposition rates for these periods would be less than the low annual average in western Montana mountains of 1.6 kg N ha^{-1} (U. S. Environmental Protection Agency 2014). Denitrification was the only output that could substantially influence apparent rates of nitrate production. Therefore, the final assessment provides a measure of net NO_3^- production (NNP) that includes mainly N mineralization and denitrification (Eq. 3.3), where SNO_3^- is the soil nitrate concentration in kg N ha^{-1}.

$$Eq\ 3.3: NNP\ (kg\ N\ ha^{-1}\ day^{-1}) = (SNO_3^-{}_{final} - SNO_3^-{}_{initial})/days = N\ \ mineralization\,rate - denitrification\,rate$$

Soil NO_3-N concentrations were analyzed from soil core sampling that occurred during fallow time periods, at four to ten locations within subfields in the upper 15 or 30 cm (Table 3.1), and a NNP rate (kg N ha^{-1} day^{-1}) calculated (Eq 3.3). A model of NNP was then developed in order to estimate NNP rates during each year of the study.

Model Design. R statistical software (The R Foundation for Statistical Computing, Vienna, Austria, 2013) was used to determine the best fit for the parameters, informed by the suggested literature model fits (Appendix B.1.1). Independent variables considered for the model were soil organic matter, previous crop, soil C:N ratio, depth to rock, potassium, clay content, pH, electrical conductivity, soil temperature, soil moisture and total-N. Only soil temperature, soil moisture and total-N were found to be significant explanatory variables; the others were excluded. With these three variables, model selection was determined by Akaike Information Criterion (AIC), which was calculated for each model tested, as well as its ability to predict NNP at the subfield level (Eq. 3.4).

$$Eq\ 3.4\text{: NNP} = -0.71 + 0.06T - 0.002T^2 + 0.66VWC + 0.0001TN$$

Net NO_3^- production and temperature were related quadratically. Log transformations and interactions between variables were both analyzed but did not provide realistic results. For example, an interaction term and logarithmic transformed interaction term between VWC and total-N fit the data better (higher R^2) but caused NNP to increase as total-N concentration fell below ~2600 kg N ha^{-1}, counter to literature. Modeled NNP amounts were averaged by subfield-period and compared against average actual measured NNP to test the model's ability to estimate NNP at the subfield level (Figure B.1.4).

The model was used to calculate NNP for four arbitrary distinct time periods each year (Aug/early Sept soil sampling to Nov 14; Nov 15 to Apr 30; May 1 to June 30; July 1 to Aug/early Sept soil sampling). Net nitrate production for each time period was summed to calculate an annual amount for use in the NO_3^- leaching mass balance equation. This model used average soil temperature and VWC for each specific subfield-period, whereas total-N was assumed to be constant and consistent with inventories observed at the beginning of the study.

Soil Temperature. Averaged daily soil temperature data in the model originates from on-field temperature probes (HOBO Pro v2, Onset Computer Corporation, Bourne, MA) and the Moccasin, MT NRCS Soil Climate Analysis Network (SCAN) soil temperature probe at 20 cm (Natural Resources Conservation Service 2015; Hydra Probe Analog (2.5V), Stevens Water Monitoring Systems, Inc. Portland, OR). On-field temperature probe measurements at 20 cm were used for both 0 to 15 and 15 to 30-cm depths. The SCAN probe was located under perennial grass and the data were only used in the model when applicable to a similar field management (i.e. a growing crop). When daily temperature was below 0 ^{o}C, it was assumed that N mineralization was negligible (Zheng et al. 1993) and these periods were excluded.

Soil Moisture. Volumetric water content (VWC) was measured continuously using five on-field moisture probes (10HS Soil Moisture Sensor, Decagon Devices, Inc. Pullman, WA) and from soil core samples collected periodically. Daily VWCs from the three probes on Field C were averaged to represent the fallow (2012) –winter wheat (WW) – spring wheat (SW) cropping system on fields A and C, and VWCs from the two Field B probes were averaged to represent fallow (2013) – WW system on fields A and B. Based on a relationship found between core and probe VWC, an equation (Eq. 3.5) was used to correct the field C probe data to better match core sample VWCs and field B probe data.

$Eq\ 3.5: Field\ C\ Corrected\ VWC = 1.27(Probe\ VWC) - 0.13$

Data gaps, over winter measurements and pea growing periods were times within the data where other corrections were required (Appendix B.1.2). Probe VWCs were assumed to be similar in subfields on field A as on Field B or C with the same cropping system based on comparisons with soil core VWCs.

Soil Total-N. Soil total-N values used in the model were from 0 to 15-cm soil samples collected in 2012. Total-N was converted from percentage to kg N ha^{-1} using core length, bulk density, and coarse fragment measurements. A correction factor, the ratio found between 15 to 30-cm and 0 to 15 total-N concentrations analyzed from soil pit samples via combustion (Costech Analytical Technologies, Inc. Valencia, CA; Ewing unpublished data 2014), was used to estimate 15 to 30-cm total-N on each field. Specifically, correction factors were 0.94, 0.85, and 0.78 for fields A, B and C, respectively.

Nitrate Storage

Nitrate storage was defined as the difference in total soil core sample NO_3-N concentrations between each post-harvest sampling period. The total soil NO_3-N concentration was a sum of NO_3-N at all depth intervals (0 to 15 cm, 15 to 30 cm, etc.) that were able to be collected before encountering rocks (15 to 105 cm depth). Within the N-balance calculation, soil core NO_3-N concentrations from each location were averaged by subfield and the average was used for each sampling location. This was done to smooth the natural variability among locations and allow N-leaching to be calculated at all ten locations within a subfield sampled in 2012, 2013, and 2014.

Statistical Analyses

Statistical analyses comparing NO_3^- leaching estimates between treatments were performed using the mixed effects model described in Chapter 2: Materials and Methods. In the NO_3^- leaching models, total-N was excluded because it was used to estimate NNP in the N-balance. In the 2013 treatment year, 12 locations on field A were excluded from the model because leaching estimates were below 0 kg N ha^{-1}. These locations were removed because the negative values indicated that groundwater N was an additional N input that could not be accounted for in the model. Notably, mottling and/or visible shallow groundwater was observed in 16 of 32 soil pits excavated on field A (Ewing et al. unpublished 2015). Evidence of shallow groundwater was not observed in soil pits excavated at the other fields, where groundwater levels were known to be approximately 5 to 10 m below the soil surface (Ewing et al. unpublished 2015). Also, leaching was not calculated for the 2013 A6 and A7 subfields (pea and fallow) because of shallow groundwater influence based on negative leaching values (-150 to -3 kg ha^{-1}) at all ten A6 locations and spotty late season pea greenness that coincided with very high pea biomass and non-fixed N uptake.

Results and Discussion

Net Nitrate Production

Both soil temperature and total-N were positively correlated with NNP (Table 3.2), consistent with other research (Cassman and Munns 1980; Vigil et al. 2002). The temperature squared (T^2) variable was negatively correlated with NNP, likely a response to increased denitrification rates at higher temperatures. At average VWC (0.25 cm^3 cm^{-3}) and total-N (2800 kg N ha^{-1}), NNP becomes negative above 23 ^{o}C, suggesting denitrification rates exceed mineralization rates at higher temperature. In model construction, subfield-periods with the warmest temperatures also had the highest VWCs which would have created more ideal conditions for denitrification (De Klein and Van Logtestijn 1996). It is surprising that VWC was not significantly related (P=0.12) to NNP, as observed by others (Curtin et al. 2012). It is likely that there were not enough VWC values (n = 5) in the model to adequately capture the effect.

Overall, the NNP model was able to predict roughly 80 % of the subfield-period level variability in NNP (R^2 = 0.81). The average predicted NNP among the seven subfield-periods was 0.13 kg N ha^{-1} d^{-1} and average absolute difference was 0.03 kg N ha^{-1} d^{-1} (Table 3.3). Squared forecast errors (SFE) were calculated for each subfield-period by removing one subfield-period data set at a time and using each new model to predict NNP for the omitted subfield-period data. For the seven subfield-periods, the average high SFE (n=2) was 0.019 which was ~70 % higher than the average low SFE (n=5). Despite some error in the NNP model, when the two models with the highest SFE were used to estimate NO_3^- leaching, the treatment results did not change.

The average predicted NNPs under wheat in 2013 and 2014 were ~70 kg N ha^{-1} and ~62 kg N ha^{-1}, respectively. It is important to note that using NNP data from fallow periods does create some uncertainty in estimating NNP rates when a crop is growing, though VWC is likely the most important factor affected by a crop, and calibration data VWCs were as low as 0.16 cm^3 cm^{-3}, not that different than under a crop. A study in western Canada found 54 to 70 kg N ha^{-1} of net N mineralization occurred from harvest through the following wheat growing season (Campbell et al. 2008) which reasonably agrees with our estimates. The lowest predicted NNP values (24 to 43 kg N ha^{-1}) were in the 2013 treatment year under pea and fallow fields that had been barley the previous year. VWCs after barley were near the wilting point from late summer to mid fall when post-fallow fields were near field capacity, resulting in large differences in fall mineralization estimates between post-barley and post-fallow fields.

Nitrate Leaching Results

In the first year of the study (2013), ~48 kg N ha^{-1} on average, leached out of the root zone of winter wheat fields. In 2014, ~53 kg N ha^{-1} on average, leached. These rates are similar to preliminary lysimeter and 1-Demensional model NO_3^- leaching quantities in a companion study (Sigler et al. 2013). In the 2013 treatment year, wheat after pea leached less NO_3^- (20 kg N ha^{-1}) than wheat after fallow (56 kg N ha^{-1}; Table 3.4). Our results are similar to a crop rotation leaching study in Canada where fallow-wheat leached more than continuous wheat and wheat- lentil rotation was no different than continuous wheat (Campbell et al. 2006). Though NO_3^- leaching was not estimated on pea and fallow subfields the previous year (2012), the lower soil NO_3^- and VWC after pea should have reduced leaching the subsequent year. Post-harvest soil NO_3^- concentrations were on average 16 and 59 kg N ha^{-1} for pea and fallow, respectively. Also, soil VWC at the start of the 2013 treatment year was close to wilting point at pea harvest (~0.16 cm^3 cm^{-3}) and field capacity on fallow (~28 cm^3 cm^{-3}). On the 2013 pea-fallow B fields, low quantities of NO_3^- leached from pea (~5 kg N ha^{-1}) and fallow (8 kg N ha^{-1}; Table 3.5). This is likely due to low N mineralization rates during a dry fall in 2012 post barley, possible denitrification losses in a wet 2013 spring higher than model predicted, and no N fertilization. Leaching is likely highest towards the end of the fallow period due to higher

VWC in the soil profile and a longer period for NO_3^- accumulation from N mineralization to occur.

In the 2014 treatment year, no leaching difference between wheat after pea and wheat after fallow was observed, in part because soil NO_3-N concentrations at the start of the 2014 treatment year only differed by 17 kg N ha^{-1} between pea and fallow (compared to ~43 kg N ha^{-1} in 2012). With a particularly wet fall, ~60 mm in the month of September (Chapter 2: Table 2.3), pea residue and soil organic matter likely began to mineralize and increased the amount of NO_3^- available to leach. Other research has suggested that pea in rotation can increase NO_3^- leaching when compared to continuous wheat (Thomsen et al. 2001), but this was in Denmark which receives ~50 % more precipitation than our study area.

In the 2013 treatment year, neither fertilizer AMP had an effect on NO_3^- leaching. This finding is not consistent with research that demonstrates the potential of reduced NO_3^- leaching (Mikkelsen et al. 1994; Nakamura et al. 2004) when using these practices. Conceptually, increased NUE from these practices (Shoji et al. 2001; Mohammed et al. 2013) could decrease NO_3^- leaching but mixed NUE results in the NGP (Grant et al. 2012) reveal that these practices are not universally effective. The varied results among studies point to the importance of environmental factors on the extent that these two AMPs affect NO_3^- leaching. April to June precipitation in 2013 likely increased NO_3^- leaching losses, especially on field C where 313 mm of rainfall was recorded (Chapter 2: Table 2.3), close to the annual LTA. In the 2014 treatment year, no leaching effect was observed with split application but a surprising result was found when more NO_3^- leached under the CRU treatment than the CU treatment ($p<0.1$). Researchers in Minnesota expressed the possibility of post growing season NO_3^- leaching after using CRU when they found significantly higher residual NH_4^+ and NO_3-N soil concentrations after potato harvest with CRU than other fertilizer treatments in one study year (Venterea et al. 2011). They attributed this finding to incomplete CRU dissolution in the previous year. However, a carryover effect from 2013 to 2014 treatment year is unlikely in our study because ESN prill sampling in late June 2013 revealed that prills were only ~5 % of their original weight. Also residual soil NO_3-N concentrations post 2013 harvest only differed by 6 kg N ha^{-1} between CRU and CU subfields. The CRU leaching result in 2014 could be in response to an early May CRU application on field C compared to an early June CU application that facilitated quicker CRU release and led to low post harvest soil NO_3^- on field C1. The average absolute difference between model predicted and estimated means for each crop treatment across years (n=12) was only ~1.7 kg N ha^{-1}, substantially less than the mean NO_3^- leaching estimate of 50 kg N ha^{-1}, indicating that the model was working well at the crop treatment level (Table 3.7). The average NO_3^- leached is a substantial amount of lost NO_3^- considering that the average annual fertilizer input for the two year study was ~87 kg N ha^{-1} (Table 3.5 and 3.6). On average, 44 kg N ha^{-1} was incorporated back into the system from crop residue (stubble plus root) when averaged across years which is almost equivalent to the amount of NO_3^- leached. Assuming that 50 % of the leached NO_3^- is from mineralized N, total-N accumulation rates would be positive but very low compared to the total-N in the top 30 cm of ~5400 kg N ha^{-1}.

In 2014 on field C, the N balance estimated that 33 kg N ha^{-1} leached from the root zone between Aug sampling and Apr 30, whereas 26 kg N ha^{-1} leached during the May to Aug sampling period. The wet fall in 2013 likely increased the amount that leached in that overwinter period but this amount would have likely been substantially less in the first year where an exceptionally dry fall was observed. Nevertheless, it appears that both fall and spring can be important times for leaching events based on precipitation patterns.

In the 2014 treatment year, depth to rock was inversely related to NO_3^- leaching ($P<0.05$; Table 3.8). In deeper soils, it likely takes a greater flux of water for NO_3^- to leave the vadose zone, increasing the opportunity for NO_3^- uptake. In the 2013 treatment year, depth to rock did not influence NO_3^- leaching. Depth to rock was likely more influential at decreasing NO_3^- in 2014 because of timing of SW root maturity or increased water flux due to rainfall in fall 2013. In 2014, WW did not have an effect on NO_3^- leaching compared to SW. This was not expected because WW's more developed and deeper rooting systems can decrease downward movement of NO_3^- (Singh and Sekhon 1976).

Implications for Sustainability

Research has revealed that agricultural practices that encourage both environmental and economic sustainability have the highest adoption probability (Nowak 1992; Reimer et al. 2012; Gabriel et al. 2013). In our study, replacing fallow with pea decreased NO_3^- leaching in one year (2013) and increased net revenue in one year (2014; Chapter 2: Results and Discussion). Furthermore, replacing fallow with legumes can have soil quality benefits (O'Dea et al. 2015) which could increase the sustainability of this practice long-term. The possibility of increased NO_3^- leaching following pea due to increased N mineralization creates a unique opportunity to further the sustainability of this practice. If an agricultural producer could account for the N amount mineralized from pea residue than fertilizer application could be decreased while still preserving optimum yields. This strategy would not only decrease the possibility for leaching but save the producer money on fertilizer.

Conclusion

Our study results revealed that replacing pea with fallow can decrease NO_3^- leached amounts during the subsequent crop year in a semiarid region. Replacing fallow with pea was also the only practice that increased net revenue in one of the study years (Chapter 2: Results and Discussion). Neither fertilizer AMP decreased the amount of NO_3^- leached in either year, and in one year (2014), CRU increased the amount of NO_3^- leached compared to CU. The CRU result in 2014 merits additional research on the leaching potential of controlled-release fertilizers. Furthermore, the fertilizer AMPs did not affect net revenue in either year of the study (Chapter 2: Results and Discussion). Future research on practices that can decrease NO_3^- leaching in the NGP should focus on reducing the extent of fallow, because it has the highest likelihood of adoption for economic reasons. Cropping intensification is especially important on shallow soils where N leaching losses are greater.

Table 3.1. Field sampling periods used in the net nitrate production model and the number of cores sampled at each date along with the depth.

Sub Field	Initial Sampling Date	Final Sampling Date	15 cm cores*	30 cm cores*
A4	5/8/2012	8/13/2012	10	0
A4	8/13/2012	11/6/2012	10	0
A5	8/13/2012	11/6/2012	10	0
B1	8/13/2012	11/6/2012	10	0
B2	8/13/2012	11/6/2012	10	0
B3	8/14/2012	5/14/2013	5	3
B3	7/2/2013	8/6/2013	4	4

* This is the number of cores per sample date

Table 3.2. Net nitrate production model results, showing independent variable coefficients and p-values.

Model	R^2	P-value
	0.23	0.0025
Variable	Coefficient	
Intercept	-0.71	0.002
T	0.062	0.042
T^2	-0.0021	0.033
VWC	0.66	0.124
TN	0.000093	0.035

T = Soil Temperature; T2 = Soil Temperature Squared; VWC = Volumetric Water Content; TN = Total-N

Table 3.3. Net nitrate production measured subfield averages compared to model predicted subfield averages.

Subfield	Period	Measured (kg N ha^{-1} day^{-1})	Predicted (kg N ha^{-1} day^{-1})	AD†
A4	8 May - 13 Aug 2012	0.21	0.18	0.03
A4	13 Aug - 6 Nov 2012	0.17	0.17	0.01
A5	13 Aug - 6 Nov 2012	0.06	0.13	0.08
B1	13 Aug - 6 Nov 2012	0.13	0.10	0.03
B2	13 Aug - 6 Nov 2012	0.27	0.25	0.02
B3	14 Aug 2012 - 14 May 2013	0.02	0.00	0.02
B3	7 July - 16 Aug 2013	0.08	0.09	0.01
	Average	0.13	0.13	

† AD = Absolute Difference

Table 3.4. Model predicted NO_3^- leaching for each management practice for 2013 and 2014.

		2013			2014		
Comparison†	Fields	Crop	NO_3^- Leached (kg N ha^{-1})		Crop	NO_3^- Leached (kg N ha^{-1})	
			AMP	GSP		AMP	GSP
Pea - Fallow	A&B	WW	20**(2.5)‡	56 (3.6)	WW	41 (10)	39 (7.0)
CRU - CU	A&C	WW	47 (3.6)	50 (3.8)	SW	69*(6.2)	54 (6.7)
Split app. - SBU‡‡	A††&C	WW	61 (3.6)	55 (4.4)	SW	58 (10)	54(6.7)

*Significantly different than the GSP at the 0.1 probability level

**Significantly different than the GSP at the 0.05 probability level

† Pea vs fallow treatment is based on wheat crop after year of treatment; CRU = Controlled release urea (ESN); CU = Conventional urea; SBU = single spring broadcast urea

†† In 2014, split application was not completed on Field A and was analyzed as a GSP with lower N fertilizer rate

‡ Numbers in parentheses are standard deviations

‡‡ In 2014, split application and SBU averages included only subfields C3 and C2

Table 3.5. Average 2013 subfield N-balance measurements and estimates used to calculate NO_3^- leached in the Judith River Watershed of central Montana.

Subfield	-----------Inputs (kg N ha^{-1})-----------			-------Outputs (kg N ha^{-1})-------			NO_3^- Leached§ (kg N ha^{-1})
	N Fertilizer	NNP†	N Deposition††	Volatilization	N-Uptake‡	Δ Nitrate‡‡	
A1	98	61	1.6	4.0	152	-34	38
A2	98	64	1.6	11	176	-41	17
A3	98	53	1.6	9.0	131	-37	50
A4W	98	60	1.6	11	112	-22	59
A4E	98	59	1.6	11	152	-41	37
A5	98	60	1.6	11	177	-34	35
B1	67	65	1.6	7.1	104	8	16
B2	67	94	1.6	7.1	135	-54	75
B3	0.0	41	1.6	0.0	0.0	35	8.0
B4	0.0	43	1.6	0.0	21	18	5.0
C1	81	73	1.6	0.0	119	-16	52
C2W	81	76	1.6	8.9	118	-32	64
C2E	81	83	1.6	8.9	128	-33	62
C3	84	89	1.6	7.2	132	-36	71

† NNP = N mineralization - denitrification

†† N deposition includes both wet and dry

‡ N-uptake includes data from aboveground biomass measurements and belowground estimates

‡‡ Δ Nitrate = soil NO_3^- final - soil NO_3^- initial

§ 12 locations from field A were removed from average due to evidence of shallow groundwater influence

Table 3.6. Average 2014 subfield N-balance measurements and estimates used to calculate NO_3^- leached in the Judith River Watershed of central Montana.

Subfield	-----------Inputs (kg N ha^{-1})-----------			-------Outputs (kg N ha^{-1})-------			NO_3^- Leached (kg N ha^{-1})
	N Fertilizer	NNP†	N Deposition††	Volatilization	N-Uptake‡	Δ Nitrate‡‡	
A1	87	62	1.6	5.4	74	6.5	65
A2	87	66	1.6	9.3	75	14	55
A3	59	56	1.6	5.4	46	9.7	56
A4	87	59	1.6	9.3	58	23	58
A6	102	48	1.6	11.4	114	-13	40
A7	102	62	1.6	11.4	128	-30	57
B3	71	61	1.6	7.5	137	-33	22
B4	71	70	1.6	7.5	106	-12	41
C1	92	61	1.6	6.9	87	-13	74
C2W	92	62	1.6	10.7	94	2.3	48
C2E	92	64	1.6	10.7	96	-2.2	53
C3	92	70	1.6	8.9	97	-0.1	58

† NNP = N mineralization - denitrification

†† N deposition includes both wet and dry

‡ N-uptake includes data from aboveground biomass measurements and belowground estimates

‡‡ Δ Nitrate = soil NO_3^- final - soil NO_3^- initial

Table 3.7. Average model predicted and estimated NO_3^- leached quantities by treatment for each year of the Judith River Watershed study.

Treatment†	NO_3^- Leached (kg N ha^{-1})					
	------------2013------------			------------2014------------		
	Pred.	Est.	AD††	Pred.	Est.	AD
Pea	20	19	1.6	41	41	0.0
Fallow	56	57	0.3	39	39	0.0
CRU	47	45	1.6	69	69	0.0
CU	50	41	9.0	54	52	1.5
Split App.‡	61	59	1.6	58	58	0.0
SBU	55	59	4.1	54	53	0.7
Averages	48	47	3.1	52	52	0.4

† CRU = controlled release urea; CU = conventional urea; SBU = single broadcast urea

†† AD = absolute difference

‡ In 2014, split application and SBU averages include only subfields C3 and C2

‡‡ Pred. = values from mixed effects model; Est. = values from N-balance equation

Table 3.8. Soil and climate parameter correlations with NO_3^- leaching. Soil parameters, except depth to rock, came from 0-15 cm cores.

Parameter†	--------2013-------- NO$_3^-$ Leaching (kg N ha^{-1})	--------2014-------- NO$_3^-$ Leaching (kg N ha^{-1})
Mgmt&Clim. (B)	19.6* (-7,46)	-21.1** (-40,-2.6)
Mgmt&Clim. (C)	18.5* (-6,43)	1.40
Crop (WW)	NI††	-8.98
Depth to Rock (cm)	-0.19	-0.29** (-0.4,-0.15)
Clay (g kg^{-1})	0.48	-0.91
Olsen P (g kg^{-1})	-0.15	0.001
Coarse Fraction (kg kg^{-1})	-19.8	30.4
Salts (dS m^{-1})	-16.9	12.9

*Significant at the 0.1 probability level

**Significant at the 0.05 probability level

† Mgmt&Clim. = Categorical variable that includes confounding variables such as weather, seeding date, etc.

†† NI = Not included in the model due to collinearity criterion

‡ 95 % confidence intervals for significant variables are in parentheses

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CHAPTER FOUR

SUMMARY

Small grain farming in the northern Great Plains (NGP) is a vital industry for the United States and the world but with many high intensity land use practices there is a possibility for negative externalities. The loss of nitrogen (N) through nitrate (NO_3^-) leaching is an increasing concern for agricultural producers and stakeholders in the Judith River Watershed (JRW) of central Montana, and other watersheds in the region. Not only can NO_3^- leaching be an economic loss to a producer but the potential health implications associated with NO_3^- contaminated drinking water merits some form of action. This problem presents a unique challenge for research of finding alternative agricultural management practices (AMP) that can reduce NO_3^- leaching but also promote the welfare of those implementing them.

Our 2-yr study compared three AMPs (pea, controlled release urea [CRU], split application) to grower standard practices (GSP; fallow, conventional urea, spring broadcast urea) in grain yield, grain protein, net revenue, and the amount of NO_3^- that leached from each practice. The three AMPs were selected by producer-researcher collaboration with a focus on practicality within already established operations. The practices were tested on large scale commercial farms further increasing the capability of this study to provide invaluable implementation information to commercial farmers in the region.

Over the course of the project, the average annual loss of NO_3^- to leaching was substantial (50 kg N ha^{-1}) in comparison to the addition of fertilizer N (87 kg N ha^{-1}). The results of this study revealed that replacing pea with fallow can decrease NO_3^- leaching. This practice utilizes more NO_3^- and water in the vadose zone hindering the ability for NO_3^- to move downward. Replacing pea with fallow was also the only practice that increased net revenue over the respective GSP. When peas are grown as a commodity they provide an extra year of revenue opposed to a year of costs when using fallow. The two fertilizer AMPs did not decrease NO_3^- leaching, and in one year, CRU actually increased it. The fertilizer AMP results in our study were greatly impacted by climate and revealed that careful consideration needs to be made before using them. Delayed application and slow release fertilizers should reduce NO_3^- leaching in concept, but it appears that many factors can influence their ability to do so. Further research is required on the use of split applications and the possibility of a more effective application time that could reduce NO_3^- leaching.

We have gained an understanding on precipitation timing in how it relates to NO_3^- leaching and that spring and fall are sensitive periods for leaching potential. The soil characteristic analysis revealed that depth to rock was an important factor for leaching and net revenue. It is imperative that soil characteristics be taken into account when researching and proposing AMPs.

In conclusion, our project results demonstrate that reducing the practice of fallow in this region has potential to reduce NO_3^- leaching and increase net revenue if pea is grown as a replacement. To increase the accuracy of future NO_3^- leaching estimates in this region, it will be beneficial to research in situ N mineralization measurements over longer time periods and in various cropping systems. With study duration of only two yrs, the long term sustainability of replacing pea with fallow should be investigated further. It will be important to observe the potential economic profitability that a fallow replacement has over the course of many years. Future research should likely focus on the inclusion of other crops in a wheat cropping system and their effectiveness at reducing NO_3^- leaching. Finally, research that includes local stakeholders in the scientific process can provide invaluable education opportunities and empower the community to take ownership of finding solutions to issues.

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APPENDICES

APPENDIX A

CHAPTER 2: SUPPLEMENTAL MANAGEMENT AND ECONOMIC INFORMATION

Table A.1. Management table for 2013 and 2014 pea-fallow treatments on field A. Data obtained through periodic grower management interviews.

Pea	A4	A5	A6	A7
Seeding Date	--	Fall 2012	5/14/2013	--
Cultivar	--	Windham Yellow	Montech 4152	--
Starter Date	--	At Seeding	5/14/2013	--
Starter	--	16-24-12-10	15-26-12-10	--
Nitrogen Rate (kg N ha^{-1})	--	9	8	--
Pesticide Application 1	Roundup w/Banvel	Roundup	Roundup	Roundup
Pesticide Application 2	Roundup w/Banvel	Assure II	Assure II	Roundup w/Banvel
Winter Wheat	--Following Year Management--			
Seeding Date	9/13-14/2012	9/13-14/2012	10/10/2013	10/10/2013
Cultivar	Dec-stone†	Dec-stone†	Jagalene	Jagalene
Starter Date	9/13-14/2012	9/13-14/2012	10/10/2013	10/10/2013
Starter	15-24-9-7	15-24-9-7	15-25-12-10	15-25-12-10
Nitrogen Rate (kg N ha^{-1})	20	20	20	20
Broadcast Urea Date	4/6/2013	4/6/2013	April 2014	April 2014
Broadcast Urea	46-0-0	46-0-0	46-0-0	46-0-0
Nitrogen Rate (kg N ha^{-1})	78	78	82	82
Pesticide Application 1	--	--	Roundup	
Pesticide Application 2	--	--	Powerflex,Audit1:1,Barash240	

† Combination of both December and Yellowstone winter wheat variety

Table A.2. Management table for 2013 and 2014 pea-fallow treatments on field B. Data obtained through periodic grower interviews.

Pea	B1	B2	B3	B4
Seeding Date	4/5/2012	--	--	4/4/2013
Cultivar	5221 Yellow	--	--	Montech 4152
Pesticide Application 1	Prowl	Roundup	Roundup	Prowl
Pesticide Application 2	--	Roundup w/2-4-D	--	--
Winter Wheat	------------------------------------Following Year Management------------------------------------			
Seeding Date	9/17/2012	9/17/2012	9/17/2013	9/17/2013
Cultivar	Brawl	Brawl	Brawl	Brawl
Starter Date	9/17/2012	9/17/2012	9/17/2013	9/17/2013
Starter	15-27-7-7	15-27-7-7	15-27-5-5	15-27-5-5
Nitrogen Rate (kg N ha^{-1})	16	16	17	17
Broadcast Urea Date	3/25/2013	3/25/2013	4/10/2014	4/10/2014
Broadcast Urea	46-0-0	46-0-0	46-0-0	46-0-0
Nitrogen Rate (kg N ha^{-1})	52	52	54	54
Pesticide Application 1	Olympus		Beyond	
Pesticide Application 2	Beyond/E99/29-0-0		--	--

Table A.3. Management information for 2013 and 2014 controlled-release urea and conventional urea treatment comparisons. Data obtained through periodic grower interviews.

Winter Wheat	A1	A2	C1	C2
Seeding Date	9/13-14/2012	9/13-14/2012	9/29/2012	10/17/2012
Cultivar	Dec-stone†	Dec-stone†	Hawkin	Hawkin
Starter Date	9/13-14/2012	9/13-14/2012	9/29/2012	10/17/2012
Starter	16-26-12-10, 44-0-0	15-24-9-7	15-25-10-7, 44-0-0	15-25-10-7
Nitrogen Rate (kg N ha^{-1})	20, 49	20	17, 64	17
Broadcast Urea Date	4/6/2013	4/6/2013	--	4/7/2013
Broadcast Urea	46-0-0	46-0-0	--	46-0-0
Nitrogen Rate (kg N ha^{-1})	29	78	--	64
Pesticide Application 1	Not Applied		Powerflex,Trumpcard,Tilt2,Grounded	
Spring Wheat	--------------------------Following Year Management--------------------------			
Seeding Date	5/29-30/2014	5/29-30/2014	5/2/2014	5/2/2014
Cultivar	Vida	Vida	Vida	Vida
Starter Date	5/29-30/2014	5/29-30/2014	5/2/2014	5/2/2014
Starter	15-25-12-10, 44-0-0	15-25-12-10	13-20-10-10, 44-0-0	13-20-10-10
Nitrogen Rate (kg N ha^{-1})	20, 28	20	15, 28	15
Broadcast Urea Date	5/24/2014	5/24/2014	6/7/2014	6/7/2014
Broadcast Urea	46-0-0	46-0-0	46-0-0	46-0-0
Nitrogen Rate (kg N ha^{-1})	39	67	49	77
Pesticide Application 1	Roundup		Gold Sky, Patrol, Tilt	
Pesticide Application 2	Axial, WildCard		--	--

† Combination of both December and Yellowstone winter wheat variety

Table A.4. Management information for 2013 and 2014 split application and single broadcast urea treatment comparisons. Data obtained through periodic grower interviews.

Winter Wheat	A3	A4	C2	C3
Seeding Date	9/13-14/2012	9/13-14/2012	10/17/2012	10/17/2012
Cultivar	Dec-stone†	Dec-stone†	Hawkin	Hawkin
Starter Date	9/13-14/2012	9/13-14/2012	10/17/2012	10/17/2012
Starter	15-24-9-7	15-24-9-7	15-25-10-7	15-25-10-7
Nitrogen Rate (kg N ha^{-1})	20	20	17	17
Broadcast Urea Date	4/6/2013	4/5/2013	4/7/2013	4/7/2013
Broadcast Urea	46-0-0	46-0-0	46-0-0	46-0-0
Nitrogen Rate (kg N ha^{-1})	50	78	64	36
Liquid UAN Date	6/19/2013	--	--	5/23/2013
Liquid UAN	28-0-0	--	--	32-0-0
Nitrogen Rate (kg N ha^{-1})	28	--	--	31
Pesticide Application 1	Not Applied		Powerflex,Trumpcard,Tilt2,Grounded	
Spring Wheat	---------------------------Following Year Management----------------------------			
Seeding Date	5/29-30/2014	5/29-30/2014	5/2/2014	5/2/2014
Cultivar	VIDA	VIDA	VIDA	VIDA
Starter Date	5/29-30/2014	5/29-30/2014	5/2/2014	5/2/2014
Starter	15-25-12-10	15-25-12-10	13-20-10-10	13-20-10-10
Nitrogen Rate (kg N ha^{-1})	20	20	15	15
Broadcast Urea Date	5/24/2014	5/24/2014	6/7/2014	6/7/2014
Broadcast Urea	46-0-0	46-0-0	46-0-0	46-0-0
Nitrogen Rate (kg N ha^{-1})	39‡	67	77	49
Liquid UAN Date	--	--	--	6/25/2014
Liquid UAN	--	--	--	28-0-0
Nitrogen Rate (kg N ha^{-1})	--	--	--	28
Pesticide Application 1	Roundup		Gold Sky, Patrol, Tilt	
Pesticide Application 2	Axial, WildCard		--	--

† Combination of both December and Yellowstone winter wheat variety

‡ Split application was not completed on this field in 2014

Table A.5. Pesticide costs for each field per year. Data obtained through period grower interviews.

Input ($ L^{-1})	2012	2013	2014
Roundup	2.65-3.96	3.30-5.81	3.72
Banvel	8.45-8.96	7.40	--
Assure II	31.70	31.70†	--
2,4-D	8.45	--	--
Prowl	11.62	11.62	--
Olympus	--	648.10	--
Beyond	--	165.14	101.44
E-99	--	10.14	--
Powerflex	--	33.81	--
TrumpCard	--	47.90	--
Tilt	--	25.87‡	25.87
Patrol	--	8.28‡	5.52
Grounded	--	5.40‡	--
Gold Sky	--	--	35.38
PF,Audit1:1,Bar.§	--	--	85.51

† Based on price in 2012

‡ Based on price in 2014

§ Price based on mixture of Powerflex, Audit 1:1, and Barrage

Table A.6. Seed and fertilizer cost for each field per year. Data obtained through period grower interviews.

Input	Field A			Field B			Field C	
	2012	2013	2014	2012	2013	2014	2013	2014
Wheat Seed (\$ Mg^{-1})	--	514	294-478[‡]	--	661	588	287[§]	233[§]
Pea Seed (\$ Mg^{-1})	184	845[†]	--	441	500	--	--	--
Starter (\$ Mg^{-1})	688	665	660	--	683	518	665	498
Broadcast Urea (\$ Mg^{-1})	--	683	507	--	578	496	590	507
ESN (\$ Mg^{-1})	--	816	754	--	--	--	805	754
Liquid Urea (\$ Mg^{-1})	--	654	--	--	--	--	747	504

† Includes the cost of a fungicide treatment applied with the pea

‡ \$294 Mg^{-1} = Winter Wheat; \$478 Mg^{-1} = Spring Wheat

§ Grower used own seed. Cost includes cleaning cost and an estimated opportunity cost based on previous year wheat prices from Montana Wheat and Barley Committee.

Table A.7. Machinery costs associated with the alternative management practices for both years of the study.

Activity†	Cost
Air Seeding w/o Fert (\$ ha^{-1})	35.46
Air Seeding w/ Fert (\$ ha^{-1})	38.60
Broadcast Fert (\$ ha^{-1})	15.20
Liquid Fert (\$ ha^{-1})	22.09
Broadcast Liquid Herb (\$ ha^{-1})	15.69
Incorporate BL‡ Herb (\$ ha^{-1})	17.57
Land Rolling (\$ ha^{-1})	13.47
Harvest Cost (\$ ha^{-1})	74.75
Hauling Cost (\$ Mg^{-1})	7.72

† Data obtained from NDSU 2013 custom rates report (Aakre 2014)

‡ BL = Broadcast Liquid

Table A.8. Averaged model predicted (Pd.) and measured (Meas.) yield, protein and net revenue by treatment for each year of the Judith River Watershed study.

2013	Pea			Fallow			CRU†			CU†			Split Application§			SBU†		
	Pd.	Meas.		Pd.	Meas.		Pd.	Meas.		Pd.	Meas.		Pd.	Meas.		Pd.	Meas.	
Yield (Mg ha^{-1})	3.4	3.4	(0.0)‡	4.1	4.1	(0.1)	4.0	4.0	(0.0)	4.3	4.3	(0.0)	4.6	4.6	(0.0)	4.3	4.3	(0.1)
Protein (g kg^{-1})	121	121	(0.0)	132	132	(0.5)	131	131	(0.0)	118	118	(0.3)	107	107	(0.0)	104	103	(0.8)
Net Revenue ($ ha^{-1})	583	583	(0.0)	533	536	(2.9)	618	618	(0.0)	677	699	(22)	628	628	(0.0)	647	622	(25)
----2014----																		
Yield (Mg ha^{-1})	3.1	3.1	(0.0)	3.3	3.3	(0.0)	1.9	1.9	(0.0)	2.0	2.0	(0.1)	2.7	2.7	(0.0)	2.6	2.7	(0.1)
Protein (g kg^{-1})	131	131	(0.0)	144	144	(0.0)	154	154	(0.0)	156	158	(2.1)	124	124	(0.0)	129	128	(1.0)
Net Revenue ($ ha^{-1})	243	243	(0.0)	160	160	(0.0)	-39	-39	(0.0)	4.1	21	(17)	-24	-24	(0.0)	38	30	(8.2)

† CRU = Controlled release urea; CU = conventional urea; SBU = Single broadcast urea

‡ Absolute differences in parentheses

§ In 2014, split application and SBU averages include only subfields C3 and C2

APPENDIX B

CHAPTER 3: SUPPLEMENTAL METHODS INFORMATION

B.1. N-balance Methods

B.1.1. Model Design Background

Numerous empirical observations describe the effects of soil temperature, soil moisture, and soil total-N on N mineralization. For soil temperature, researchers have used exponential (Dessureault-Rompre et al. 2010), linear and quadratic (Cassman and Munns 1980; Booth et al. 2005) fits for their data. Soil moisture and N mineralization are often linearly related (Heumann et al. 2011; Curtin et al. 2012) as are soil total-N concentrations and N mineralization (Vigil et al. 2002). An interaction term between soil moisture and temperature was also widely used in the literature (Cassman and Munns 1980; Curtin et al. 2012; Guntinas et al. 2012).

B.1.2. Data Corrections Made to Volumetric Water Content

The five probes that were chosen to be used were C1.10, C2.01, C2.10, B3.03, and B3.04 because they had the fewest issues with battery failure resulting in data gaps and agreed the best with soil core sample VWCs. Individual probes used were averaged by field. For both probe averages used in the model, wilting point was assumed to be ~0.15 cm^3 cm^{-3} and field capacity was ~0.30 cm^3 cm^{-3}. Although pressure plate analyses were not conducted on our soils, soil water retention curves from the NRCS SCAN site at Moccasin show wilting point at 19 cm^3 cm^{-3} and field capacity at 0.30 cm^3 cm^{-3} in the upper 15-cm (Natural Resource Conservation Service 2015).

Probe malfunctions created data gaps which were connected using a linear interpolation (Figure B.1.1). Over winter measurements were corrected because the probe data showed that VWC often decreased during the time period. This suggested that probes were not accurate when frozen because there was not a crop present to utilize soil water. Soil moisture data from the VWC probes were excluded from the data set on days where soil temperature was below 0 ^{o}C. To correct for when a pea crop was present on a field, a similar method was used as with the data gaps but soil core data were used to inform the probe VWC values for the time period (Figures B.1.2 and B.1.3). For this correction, both core data and probe data were combined into one dataset. Precipitation data were also used to estimate the approximate date at which soil moisture content reached field capacity.

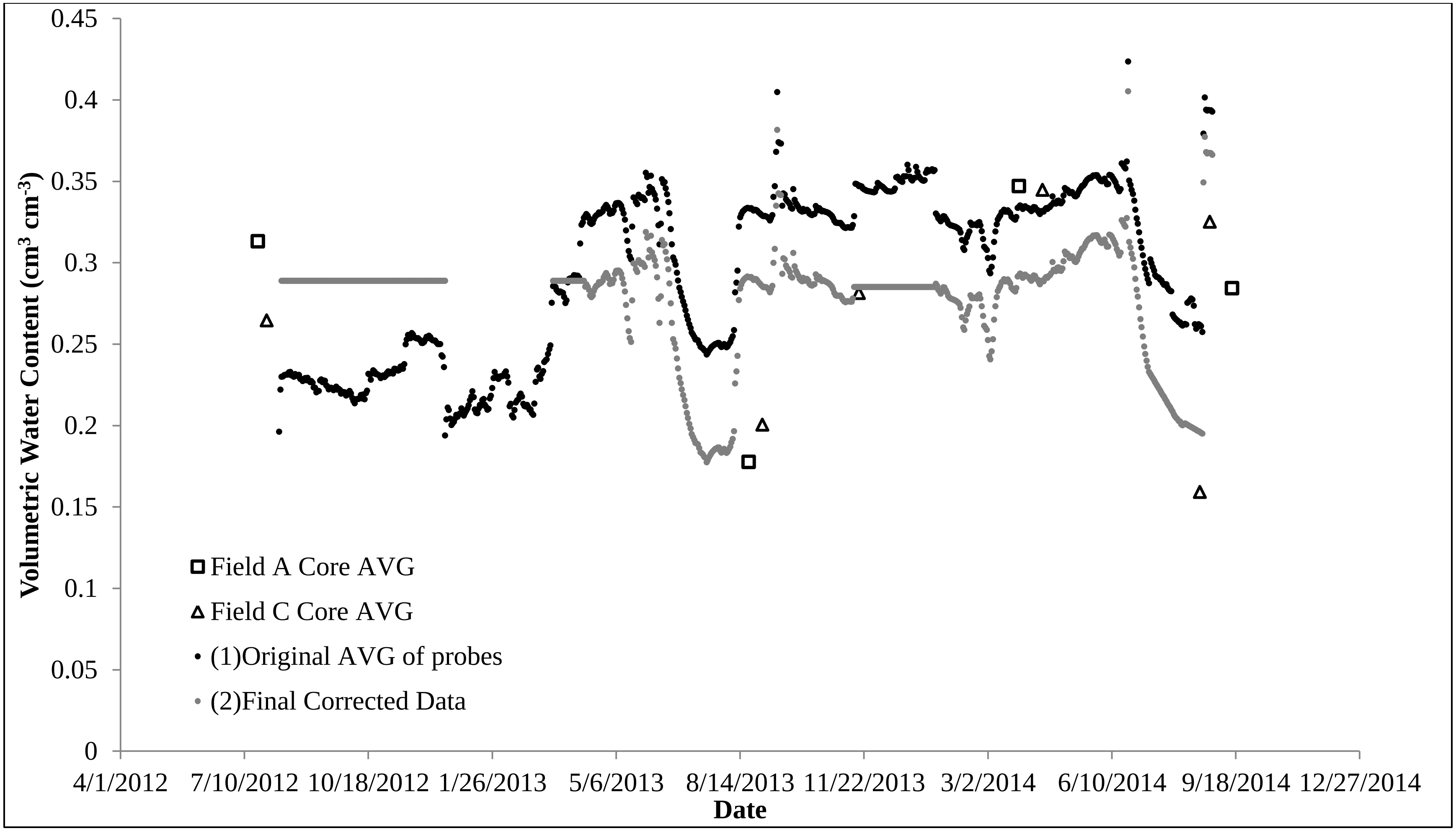

Figure B.1.1. Volumetric water contents used on subfields A1-A4, B2, and C1-C3. Data (2) are the final corrected data used in the model.

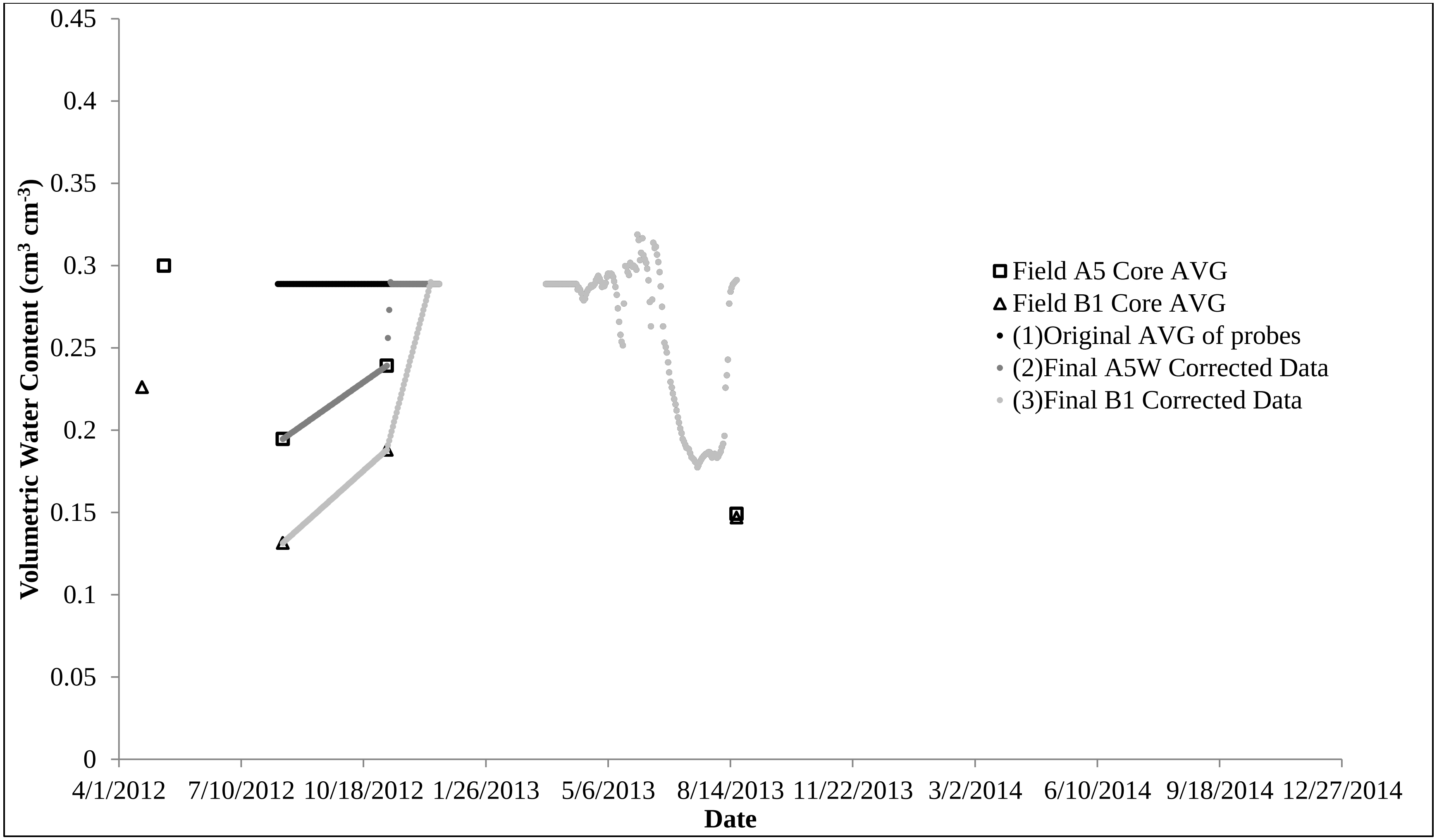

Figure B.1.2. Volumetric water content used on subfields A5 and B1. Both data (2) and (3) were corrected using a linear interpolation with core data between July 2012 and December 2012 because the fields were post pea crop.

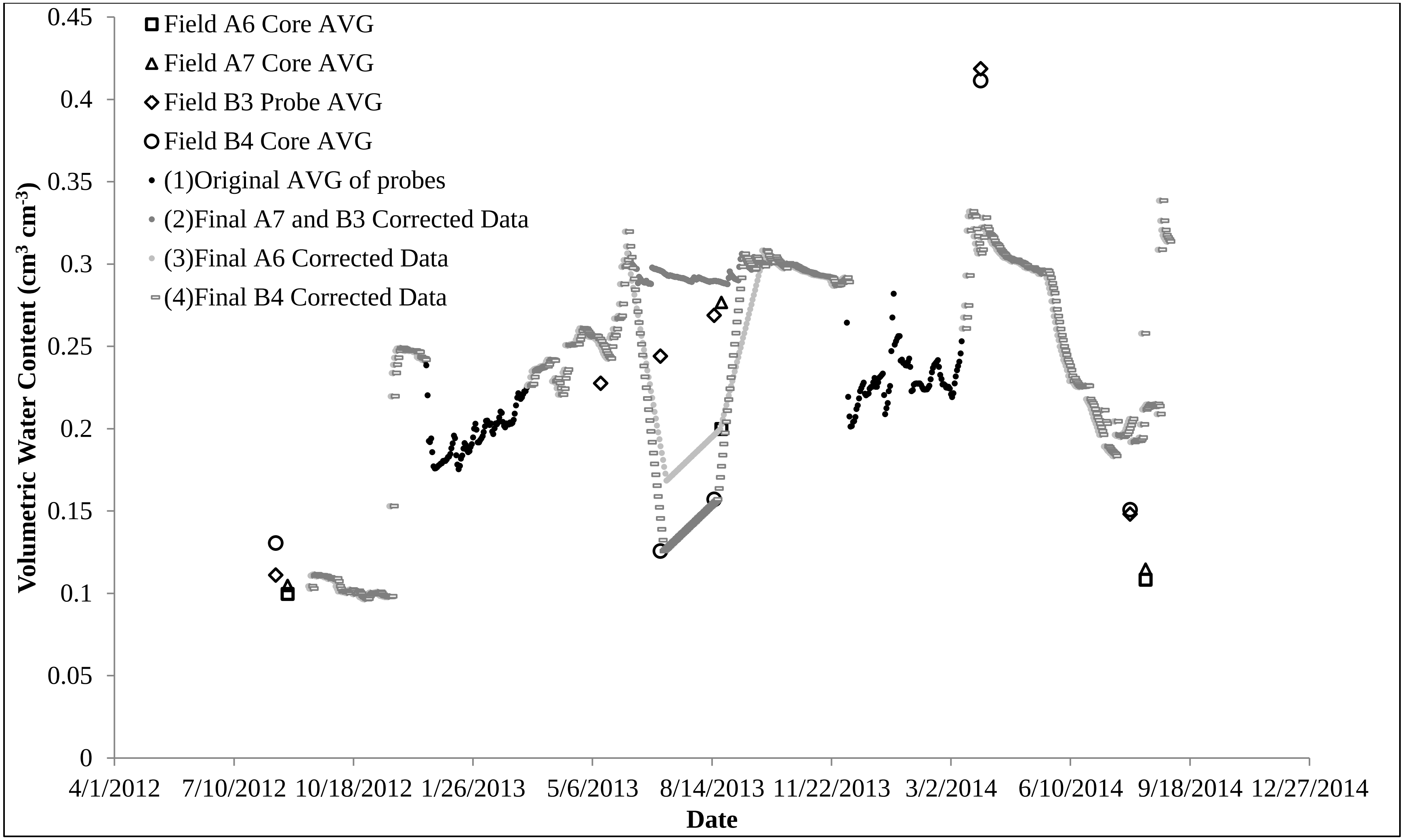

Figure B.1.3. Volumetric water content used on subfields A6-A7 and B3-B4. Data (3) and (4) were corrected with core data to resemble pea water uptake during summer 2013.

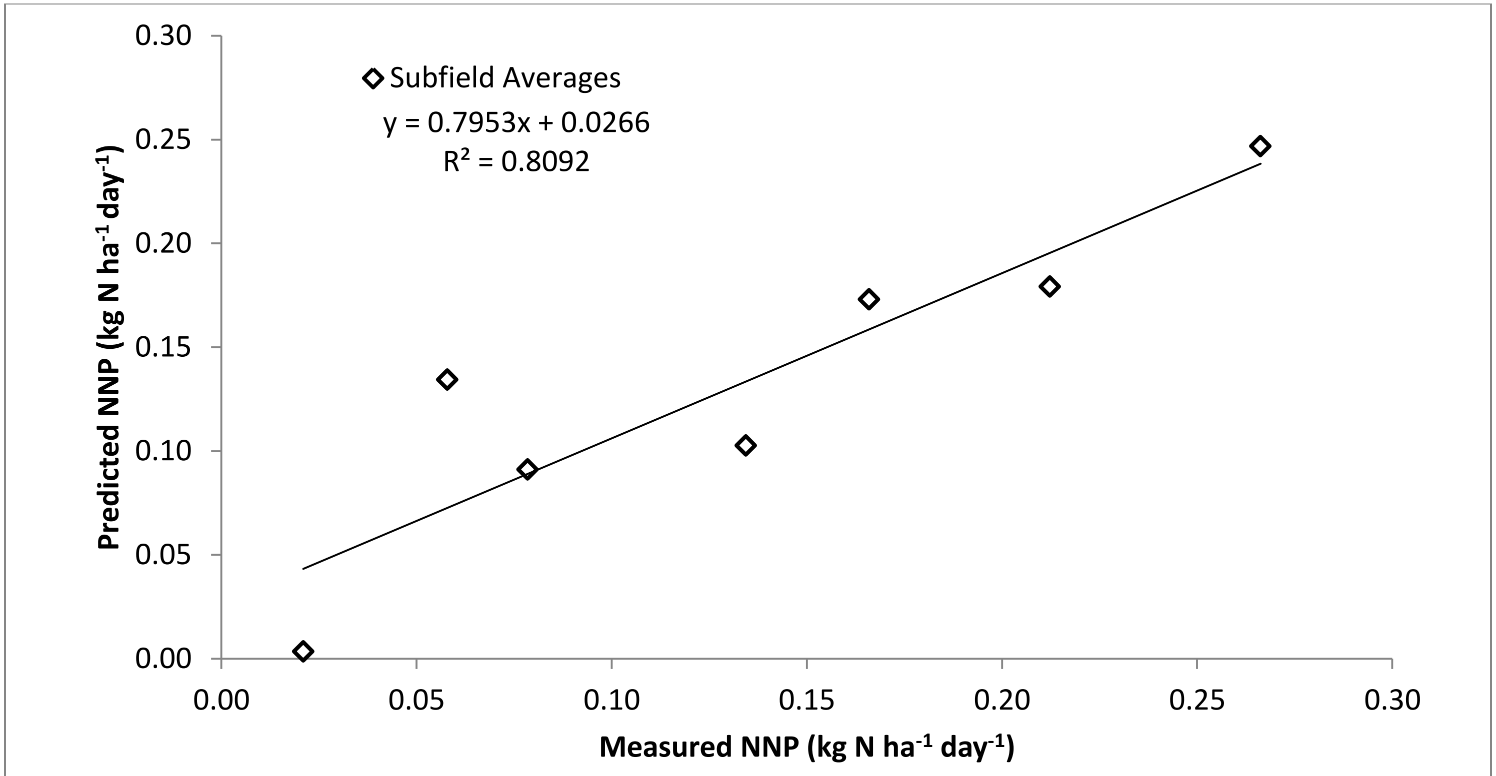

Figure B.1.4. Model predicted net nitrate production (NNP) in kg N ha^{-1} day^{-1} compared to measured NNP by subfield average. Model was used in the NO_3^- leaching calculation.

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