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ORIGINAL ARTICLE

Fallow replacement and alternative nitrogen management for reducing nitrate leaching in a semiarid region

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Abstract Nitrate (NO_3^-) leaching into groundwater is a growing global concern for health, environmental, and economic reasons, yet little is known about the effects of agricultural management practices on the magnitude of leaching, especially in dryland semiarid regions. Groundwater nitrate–nitrogen (nitrate–N) concentrations above the drinking water standard of 10 mg L^{-1} are common in the Judith River Watershed (JRW) of semiarid central Montana. A 2-year study conducted on commercial farms in the JRW compared nitrate leaching rates across three alternative management practices (AMP: pea, controlled release urea, split application of N) and three grower standard practices (GSP: summer fallow, conventional urea, single application of urea). Crop biomass and soil were collected at ten sampling locations on each side of a management interface separating each AMP from its corresponding GSP. A nitrogen (N) mass balance approach was used to estimate the amount of nitrate leached annually. In 2013, less nitrate leached the year after the pea AMP (18 $\pm$ 2.5 kg N ha^{-1}) than the year after the fallow GSP (54 $\pm$ 3.6 kg N ha^{-1}), whereas the two AMP fertilizer treatments had no effect on nitrate leaching compared to GSPs. In 2014, leaching rates did not differ between each AMP and its corresponding GSP. The results suggest that replacing fallow with pea has the greatest potential to reduce nitrate leaching. Future leaching research should likely focus on practices that decrease deep percolation, such as fallow replacement with annual or perennial crops, more than on N fertilizer practices.

Keywords Split application · Controlled release urea · Gravel · Northern Great Plains · Mineralization · Pulse crop

Introduction

Groundwater nitrate contamination due to leaching is an extensive and growing concern, especially in agricultural regions with shallow groundwater (Power and Schepers 1989; Dubrovsky et al. 2010; Puckett et al. 2011). Puckett et al. (2011) showed that average nitrate–N concentrations in 424 shallow unconfined groundwater wells across the United States increased from <2 mg L^{-1} in the early 1940s to 15 mg L^{-1} by 2003. Nitrate–N concentrations above the Environmental Protection Agency's (EPA's) primary drinking

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water standard of 10 mg L^{-1} have been linked to adverse health effects such as methemoglobinemia (Knobeloch et al. 2000; Fewtrell 2004) and cancer (De Roos et al. 2003; Ward et al. 2005). Beyond health concerns, nitrate leaching can result in economic losses for agricultural producers (Janzen et al. 2003; Miao et al. 2014), making greater N use efficiency (NUE)—conceived here as the ratio of crop N uptake to plant available N—a promising shared goal for sound environmental stewardship and economic sustainability. In the 720,000-ha semiarid Judith River Watershed (JRW) of central Montana, a large agricultural production region in the northern Great Plains (NGP), nitrate–N concentrations in shallow wells are commonly above the EPA drinking water standard. Notably, concentrations in the only long-term monitoring well in the watershed approximately doubled since 1994, increasing from ca. 10 to ca. 20 mg L^{-1} in two decades (Schmidt and Mulder 2010; Miller 2013).

Much of the small-grain agriculture in the JRW is on soils with shallow water tables and gravel contacts 30–100 cm below the ground surface. These soils are not only especially vulnerable to leaching loss of solutes, but make agricultural production on them particularly sensitive to management changes and drought. As a result, producers are often hesitant to adopt new strategies, limiting their flexibility to develop environmentally and economically successful management solutions. Soils with shallow gravel contacts are not confined only to the JRW but exist on alluvial and fluvial landforms throughout Montana, including the 200,000-ha Flaxville Gravels located in northeast Montana (Nimick and Thamke 1998).

Reducing N application rate is one option for decreasing leaching (Goulding 2000) and increasing NUE, yet decreased N rates can reduce net revenue especially when grain protein discounts are steep (Miller et al. 2015). Controlled-release urea (CRU: Mikkelsen et al. 1994; Shoji et al. 2001) and split N applications (Mascagni and Sabbe 1991; Mohammed et al. 2013) can also decrease leaching and increase NUE by lowering soil nitrate concentrations during vulnerable periods for leaching. Nitrate leaching effects of CRU and split applications have not been evaluated in the semiarid NGP, although Grant et al. (2012) found that they produced inconsistent effects on NUE in a wetter portion of the NGP.

Cropping systems can also affect efficiency of water and nitrogen use. Summer fallow, a production method in which crops are not grown for an entire season to conserve water, has been linked to high groundwater nitrate concentrations in the NGP (Custer 1976; Bauder et al. 1993; Nimick and Thamke 1998). Consistent with this finding, continuous cropping can decrease the downward movement of water and increase NUE (Campbell et al. 1984; Westfall et al. 1996). In western Canada, a continuous wheat rotation leached 180 kg N ha^{-1} less nitrate than fallow-wheat over the 37-year study (Campbell et al. 2006). Consequently, intensifying cropping systems and increasing crop diversity, such as replacing fallow with annual legumes in cereal rotations, could reduce leaching, with potential to also increase net revenue in the NGP (Zentner et al. 2001; Miller et al. 2015), yet concerns persist about increased risk from increased water use of intensified cropping systems.

Published nitrate leaching research for dryland semiarid regions in general is scant. Limited research on this topic may reflect an underestimation of the importance of leaching in low precipitation regions, but elevated groundwater nitrate concentrations in other parts of Montana and the Great Plains demonstrate its significance (Bauder et al. 1993; Scanlon et al. 2008; Schmidt 2009). In addition, rates of deep percolation from the root zone in semiarid regions are relatively small and difficult to quantify, especially in areas with highly variable gravel contact depths. Here we employ a mass balance leaching quantification approach (Meisinger and Randall 1991) that relies on spatially extensive sampling, and complements a companion lysimeter study (Sigler et al. in review).

This study seeks to address the research gap in measuring nitrate leaching rates in regions of the NGP with shallow gravel contacts and shallow water tables by identifying Alternative Management Practices (AMPs) that are most likely to be adopted by agricultural producers and then empirically testing and comparing the performance of each AMP to reduce nitrate leaching rates. Although soils with shallow gravel contacts are less common outside the JRW, these soils are more sensitive to management changes than deeper soils, and hence can provide insight to practices that can affect leaching from deeper soils during wetter periods. All studies were conducted on producer fields using their equipment, with management practices, including N rates, developed in close collaboration with producers.

Materials and methods

Study area

The 720,000-ha JRW (HUC 10040103) is located in the Upper Missouri River watershed in central Montana, within the North American NGP agroecoregion (Fig. 1; Padbury et al. 2002). Agricultural land encompasses 41% of the agroecoregion's approximately 125 million ha, with the vast majority of this agricultural land farmed using dryland practices. The JRW ranges in elevation between 760 and 1290 m, and the town of Moccasin, located near the JRW's center, receives approximately 40% of its average 390 mm annual precipitation in May and June (WRCC Gage # 245761). Annual crops in the watershed are primarily small grains, which are especially prevalent on gravel-mantled strath terraces (cut by a meandering river into relatively soft bedrock) with shallow aquifers, where elevated nitrate concentrations are most common (Schmidt and Mulder 2010). Only 1% of the annual crops grown in the two counties within JRW are irrigated (USDA 2016). Summer fallow represents approximately 27% of the cropped land (USDA 2014) in any given year, reflecting that on average, fallow is used once in a 3- or 4-year rotation.

The soils capping the terrace gravels in the JRW are derived primarily from alluvium, loess, and shale, with mainly clay loam and loam textures. They are primarily mapped as Typic Calciustolls of the Judith series, Vertic Argiustolls of the Danvers series, and Typic Argiustolls of the Tamaneen and Doughty series, which collectively are characterized by variable and shallow gravel contacts that vary from 30 to 100 cm depth, with about 20 g kg-fines^{-1} of soil organic carbon (SOC) in the upper 15 cm, and 100–600 g $CaCO_3$ kg-fines^{-1} (12–72 g C kg-fines^{-1}) in calcic subsurface horizons that predominated below ca. 30 cm (NRCS 2016).

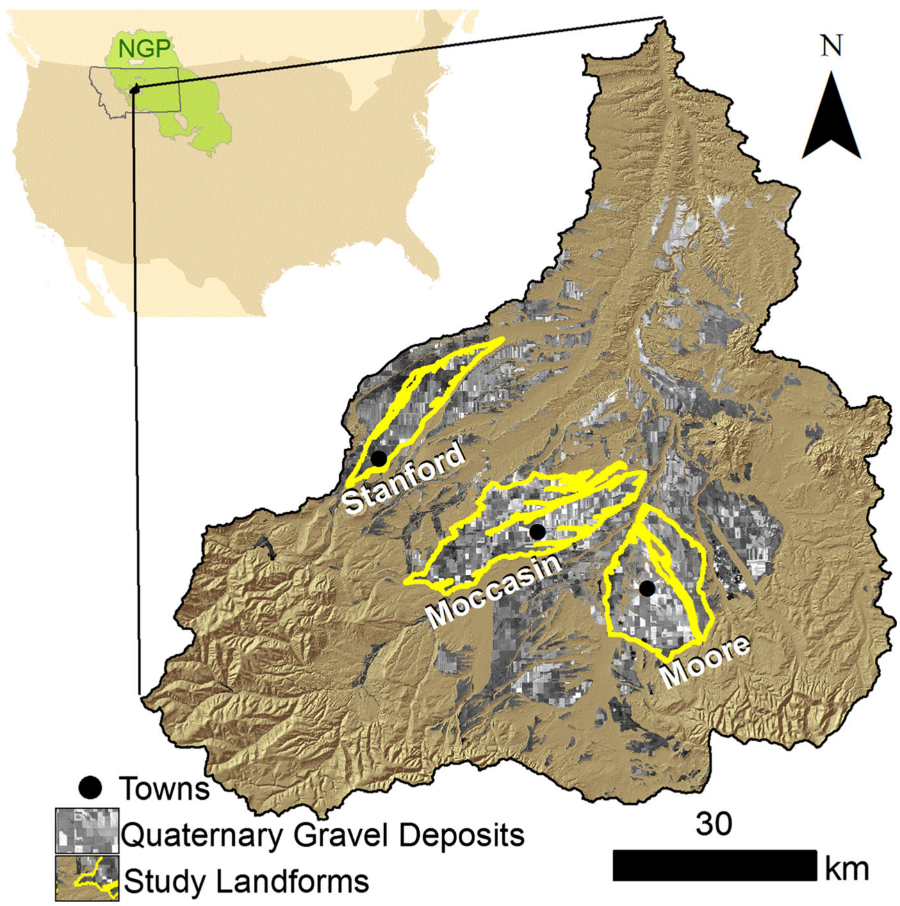


Fig. 1 Judith River Watershed location within Montana and the northern Great Plains (NGP). Aerial imagery illustrates the large extent of cropland over gravel deposits. Fields A, B, and C were located within the Stanford Terrace, Moccasin Terrace, and Moore Bench landforms, respectively; Data Sources: Montana Natural Resources Information Systems, USGS National Elevation Dataset, Commission for Environmental Cooperation, LandSat 8 imagery, MBMG Belt, Big Snowy and Lewistown Geologic Maps, USGS White Sulphur Springs Geologic Map

Study design

The study was conducted from 2012 to 2014 on three dryland commercial farms located in the JRW (Fig. 1; Table 1). Farms were chosen to ensure representation of major landform types and geographic distribution within the JRW. A representative sample of locations is particularly important for helping attenuate potential selection bias of producers who were willing to participate. Within each landform, field selection criteria included (1) presence of shallow gravel contacts within soils (<100 cm), (2) only one soil series or two-component complex per field, and (3) use of summer fallow-winter wheat-spring grain rotation (a typical JRW rotation). Fields A, B, and C were located a few km from Stanford, Moccasin, and Moore, respectively (Fig. 1) and were on farms that had been under no-till management for 10–15 years.

We surveyed randomly selected producers (Jackson-Smith and Armstrong 2012) and obtained input from both a stakeholder advisory committee and a producer research advisory group to identify three AMPs with the highest adoption potential for the JRW: pea grown for grain (followed by winter wheat), and wheat fertilized with either a controlled-release urea (CRU) or a split N application (see Supplementary Material for details on selection process). These AMPs were empirically compared with grower standard practices (GSP): summer fallow (followed by winter wheat), and wheat fertilized with conventional urea (CU) applied in a single broadcast (SB). All three AMPs were tested at Field A (Fig. 2). The sampling design at Fields B and C mimicked that at Field A, yet the only AMP tested at Field B was pea (2012 and 2013), and only fertilizer treatments were tested at Field C (2013 and 2014).

Fig. 2 Study design at each field, illustrating sampling ▶ locations (+), non-sampled buffers at field ends (*gray shading*), treatment interfaces (*dashed lines*), crop, and treatment. *WW* winter wheat, *SW* spring wheat, *Fal* fallow, *SB* single broadcast urea, *CU* conventional urea, *CRU* controlled release urea, *Split App* split application. In 2013, subfield A4 simultaneously served as a Post Fal treatment (east sample sites) and SB (west sample sites)

At each farm, full scale operational fields were divided into 10–20 ha subfields (Fig. 2). Each north–south management boundary ("interface") separated a GSP subfield from its respective AMP subfield (selected randomly). In total, there were eight GSP-AMP interfaces (four at Field A, two at Field B, and two at Field C) so that each treatment was studied in duplicate in both space (i.e., pea studied at Field A and B) and time (Table 2). Fertilizer treatments used in 2013 on winter wheat were repeated on the same fields in 2014 on spring wheat, and pea and fallow treatments were on different subfields in 2012 than in 2013 so that winter wheat could follow pea or fallow in rotation.

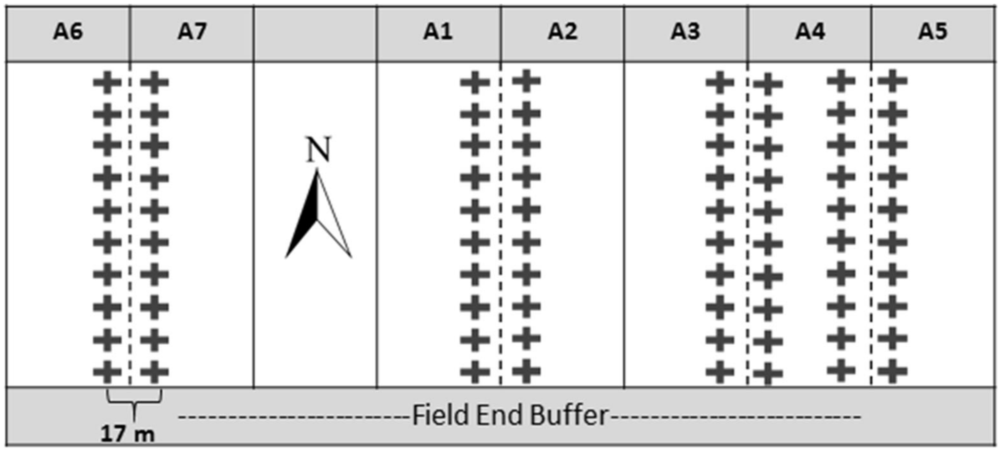
Sampling was designed to capture variation with treatment in full size working fields. Each subfield excluded areas located approximately 100 m from the

Table 1 Field descriptions, locations and average soil characteristics in the upper 15 cm for study fields in the Judith River Watershed of central Montana

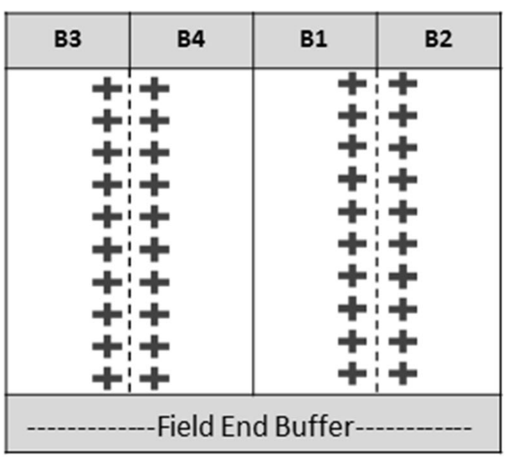
Study site	Field A	Field B	Field C
Proximity	Stanford	Moccasin	Moore
Landform	Strath Terrace	Strath Terrace	Alluvial Fan
Soil texture	Clay Loam	Clay Loam	Clay Loam
Depth to gravel contact (cm)[a]	70 (29)[b]	61 (22)	71 (33)
pH	7.1 (0.4)	7.5 (0.2)	6.5 (0.5)
Soil organic carbon (g kg^{-1})	18 (2.3)	24 (1.9)	22 (1.5)
TN (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)
Exchangeable K (mg kg^{-1})	295 (58)	307 (50)	302 (27)

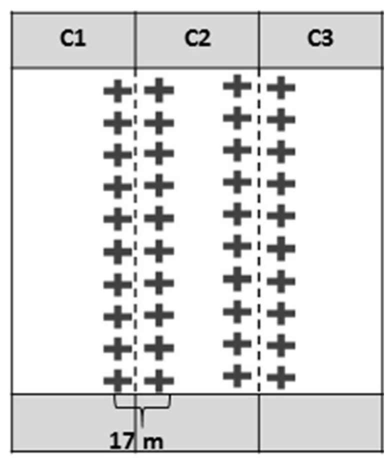
[a] Based on 28, 17, and 19 soil pits excavated at Fields A, B, and C, respectively (Summer 2012)

[b] Numbers in parentheses are standard deviations based on 40 (Fields B, C) to 80 (Field A) samples



		A6	A7	A1	A2	A3	A4	A5
2012	Crop Treat	- -	- -	Fal -	Fal -	Fal -	Fal Fal	Pea Pea
2013	Crop Treat	Pea Pea	Fal Fal	WW CRU	WW CU	WW Split App	WW SB/ Post Fal	WW Post Pea
2014	Crop Treat	WW Post Pea	WW Post Fal	SW CRU	SW CU	SW Split App	SW SB	- -





		B3	B4	B1	B2
2012	Crop Treat	- -	- -	Fal Fal	Pea Pea
2013	Crop Treat	Pea Pea	Fal Fal	WW Post Fal	WW Post Pea
2014	Crop Treat	WW Post Pea	WW Post Fal	- -	- -

C1	C2	C3
Fal -	Fal -	Fal -
WW CRU	WW CU SB	WW Split App
SW CRU	SW CU SB	SW Split App

Table 2 Treatment comparisons and crop rotations on Judith River Watershed nitrate leaching study treatment fields, 2011–2014

Treatment comparison	Test fields	Crop[a]			
		2011	2012	2013	2014
Pea–Fallow (2012)	A and B	B	P-F[b]	WW[b]	–
Pea–Fallow (2013)	A and B	WW	B	P-F[b]	WW[b]
CRU–CU[c]	A and C	B	F	WW[b]	SW[b]
Split App-SB	A and C	B	F	WW[b]	SW[b]

[a] *P* pea crop, *F* fallow, *WW* winter wheat, *SW* spring wheat, *B* barley

[b] Tested crop-year

[c] *CRU* controlled release urea (ESN®), *CU* conventional urea, *Split App* split fertilizer application, *SB* single broadcast urea

north and south field edges to minimize potential agricultural machinery disturbance from turning and/or multiple seeding/chemical passes. Ten sample locations were established per subfield in pairs equidistant from the interface (20 sample locations per interface). Sample locations were approximately 8.5 m from the interface line to avoid treatment edge effects (i.e., 17 m between paired sampling locations). The areas surrounding each sampling locations were near-level, with slopes <2%, and neither runoff, nor evidence of runoff, was observed. Soil pits excavated in summer 2012 at 4–5 locations on each side of each interface revealed presence of groundwater within 1.2 m of the land surface in 8 of 33 pits at Field A, likely due in part to near record precipitation amounts received from Sept 2010 to Aug 2011 (483 mm; Moccasin Agrimet). A network of 12 monitoring wells installed within Field A in 2012 indicated dropping water levels for the rest of that year, and in 2013, none of the measured water levels in the field were within 1.2 m of ground surface, making concerns of crop N uptake from groundwater less of a concern as the study progressed. There was no evidence of groundwater within the ca. 1 m root zone at either Field B or C, where groundwater levels were ca. 10 and 3–15 m below the surface, respectively, in monitoring wells and nearby domestic wells.

Rates of fertilizer N were within 3 kg N ha^{-1} for each AMP and its respective GSP (Table 3). In the 2013 treatment year (i.e., late summer 2012 to late summer 2013), CRU (ESN®; Agrium Inc., Calgary, AB, USA) was applied with winter wheat seed. In the 2014 treatment year, growers A and C applied about 1/3 of the total N rate as CRU with spring wheat seed and the remaining N was broadcast as urea later in the spring. While this is technically a 'split application', for purposes of this paper, we have designated the split application as broadcast urea followed by a liquid N application. Specifically, urea ammonium nitrate (UAN) was applied in late May/early June 2013 using flat fan nozzles. For the second application at Field C in 2014, UAN was applied with streamer bars, whereas on Field A, the second application was not performed due to very dry conditions. Additional details of fertilizer application are presented in John (2015).

Instrumentation and sampling

Instrumentation for air temperature, soil temperature, precipitation, and volumetric water content (VWC) are described in the Supplementary Material. Sampling to characterize subfield soils was conducted prior to initiation of treatments, between Apr and Aug 2012. Two soil core sub-samples were taken at each sample location with either a hand core or custom-made truck mounted hydraulic probe to a depth of 15 cm. Sub-samples were mixed, placed into plastic-lined paper bags, and transported in coolers on ice to Montana State University prior to further processing. Each year, within 10 days prior to commercial harvest, a biomass sample was collected from a ca. 0.6-m^2 area at each location using a rice sickle to cut the stems at soil contact.

Post-harvest soil samples were collected as close to harvest as possible (ca. July 15–Sept) during each study year (2012–2014) to minimize N mineralization between biomass harvest and soil sampling that would bias the N balance. Two 3-cm diameter soil cores were collected at each sample location with a truck-mounted hydraulic probe to the deepest possible depth; that is, either to gravel contact, anomalous rock, or the 110-cm probe depth limit (only 6 of 1130 cores). The two core

Table 3 Nitrogen fertilizer rates for each grower standard practice (GSP), and each alternative management practice applied to winter wheat in 2013 and spring wheat in 2014 (except winter wheat post-fallow GSP and post-pea)

Year, crop, field, and treatment	kg N ha^{-1}			
	Starter (w/seed)	Broadcast urea	UAN	Total N
2013—Winter wheat, all fields				
Field A				
GSP (Post-Fal[a], SB[a], CU[a])	20	78	–	98
Post-pea (SB)	20	78		98
CRU[a] (w/seed, Post-Fal)	69	29	–	98
Split App[a] (CU and foliar UAN, Post-Fal)	20	50	28	98
Field B				
GSP	16	52	–	68
Post-pea	16	52	–	68
Field C				
GSP	17	64	–	81
CRU	81	–	–	81
Split App	17	36	31	84
2014—Spring wheat, all fields except Post-Pea and Post-Fal				
Field A				
Fertilizer GSP (SB, CU)	20	67	–	87
CRU (w/seed)	48	39	–	87
Split App	20	39	–[b]	59
Post-Fal[c] GSP (w. wheat, SB, CU)	20	82	–	102
Post-Pea (w. wheat, SB, CU)	20	82	–	102
Field B				
Post-Fal	17	54	–	71
Post-Pea	17	54	–	71
Field C				
GSP	15	77	–	92
CRU	43	49	–	92
Split App (streamed UAN)	15	49	28	92

[a] *Post Fal* post fallow, *SB* single broadcast urea, *CU* conventional urea, *CRU* controlled release urea, *Split App* split application; treatments on Fields B and C were the same as on Field A

[b] Due to very dry conditions on Field A in 2014, the grower opted not to apply foliar N in the split application treatment

[c] Post-Fal in 2014 was the GSP contrasted with Post-Pea; both seeded to winter wheat

samples were then sectioned into 15-cm depth intervals and combined into one sample per depth. In 2013 and 2014, both biomass and soil samples were collected ca. 1 m north of the previous year's soil sampling location based on coordinates logged with a GPS unit (Trimble Navigation Ltd., Sunnyvale, CA, USA).

Laboratory procedures

Soil samples were weighed moist, oven-dried at 50 °C for 7–14 days, 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 broken up with a mortar and pestle and sieved (<2 mm). Nitrate was extracted from the fines in a 1:10 soil:solution ratio with 1 M KCl (Bundy and Meisinger 1994) and extracts analyzed with cadmium reduction and colorimetry using Lachat Flow Injection Analysis (QuikChem Method 12-107-04-1-B, Lachat Instruments Inc., Milwaukee, WI, USA). Each surface core (0–15 cm) collected in 2012 was analyzed for pH (1:1 soil:water), electrical conductivity (EC; 1:1 soil:water), Olsen phosphorus (Frank et al. 2012), exchangeable potassium (ammonium acetate pH 7), total-N (TN; Elementar Vario Max by combustion analyzer), soil organic carbon (SOC) by Walkley–Black (Combs and Nathan 2012), and soil texture (Hydrometer method ASTM 422).

Wheat biomass samples were dried at 50 °C for 7–14 days, weighed, and threshed with a Vogel Stationary Grain Thresher (Almaco, Nevada, IA, USA). All grain and stubble were finely ground (<0.5 mm) in a Udy mill (Cyclone Lab sample mill, Udy Corporation, Fort Collins, CO, USA), and 0.1-g subsamples analyzed for TN using an automatic combustion analyzer (TruSpec CN, LECO Corporation, St. Joseph, MI, USA).

N Balance method to calculate nitrate leaching

To estimate the amount of nitrate–N that leached from each of the treatment subfield locations, we used a simple inorganic N mass balance (Meisinger and Randall 1991; Ju et al. 2006) of annual inputs, outputs (losses) and soil storage change:

$$\begin{aligned} NO_3-N\,\mathrm{leached}\left(\mathrm{kg\,N\,ha^{-1}\,year^{-1}}\right) &= \mathrm{inputs}\left(N_{fert} + \mathrm{net}\,N_{min} + N_{dep}\right) \\ &\quad - \mathrm{outputs}\left(N_{vol} + N_{denit} + N_{uptake}\right) \\ &\quad - \left(\mathrm{soil}\;NO^{-}_{3\,final} - \mathrm{soil}\,NO^{-}_{3\,initial}\right) \end{aligned} \tag{1}$$

In this conceptualization, an inorganic-N mass balance is approximated by the combination of inputs including N fertilizer, net N mineralization (total mineralization–total immobilization), and N deposition (wet and dry); and inorganic N outputs including N fertilizer volatilization, denitrification, and crop N uptake (grain, stubble and root) from the soil inorganic N pool. Although inorganic N includes both ammonium–N and nitrate–N, nitrate–N concentrations generally exceeded ammonium–N concentrations, and ammonium–N pools in late summer were generally small compared to the overall N balance. Nitrogen runoff was not included based on nearly flat fields with no evidence of runoff.

Fertilizer N amounts were determined through grower management interviews. In both study years, TN deposition was estimated to be 1.6 kg N ha^{-1} year^{-1} using data from the EPA's nearest atmospheric deposition station in Glacier National Park (Elevation 976 m; U. S. Environmental Protection Agency 2014). The TN amount in the root biomass was set equal to 20% of the aboveground N, based on wheat root N:shoot N ratios (Andersson et al. 2005).

Broadcast urea volatilization amounts were estimated at 14% of applied N based on measured average ammonia volatilization losses from early spring (Mar 24–Apr 20) applications for eight central and north central Montana trials, including three within the JRW (Engel et al. 2011; Engel and Jones 2014). For both years, it was assumed that seed placed starter and CRU fertilizer did not volatilize, and that 7% of liquid UAN volatilized based on a Manitoba, Canada study conducted in late May (Grant et al. 1996). The change in soil nitrate storage (soil nitrate$_{final}$–soil nitrate$_{initial}$) was defined as the difference in total soil core sample nitrate–N pools between each post-harvest sampling period. The total soil nitrate–N pool (kg N ha^{-1}) was a sum of nitrate–N at all depth intervals (0–15 cm, 15–30 cm, etc.) that could be collected before encountering rocks (15–105 cm depth), averaged across each subfield.

N Mineralization and denitrification

Although neither net N mineralization nor denitrification was measured in this study, only the difference between the two was necessary to complete the N balance (see Eq. 1). This difference was predicted for each treatment subfield based on changes in soil nitrate measurements taken during seven fallow "subfield-periods" (e.g., Aug 13–Nov 6 2012 on subfield A4) of ca. 40–200 days during 2012–2013. Because these were fallow periods, fertilizer input and N-uptake were known to be zero, and there were likely negligible amounts of volatilization (Grant et al. 1996) and deposition. Based on water budgets, nitrate leaching out of the depth interval sampled was also assumed to be zero; specifically, evapotranspiration estimated from flux tower measurements on and near Field C (Vick et al. 2016) was greater (within uncertainty) than the precipitation amount minus the change in soil water storage (final–initial) during the seven subfield-periods. Therefore, the final assessment provides a measure of net nitrate production (NNP, kg N ha^{-1}) that includes mainly nitrate derived from net N mineralization and lost to denitrification (Eq. 2):

$$\mathrm{NNP}\left(\mathrm{kg\,N\,ha^{-1}\,day^{-1}}\right) = \left(\mathrm{Soil\,NO^{-}_{3\,final}} - \mathrm{Soil\,NO^{-}_{3\,initial}}\right)/\mathrm{days}. \quad (2)$$

The assumption that change in soil nitrate equals net mineralization–denitrification also assumes that nitrification rates are at least as great as net mineralization rates in this system. Using NNP data from only fallow periods does create some uncertainty in estimating NNP rates when a crop is growing; however, VWC is a key factor affected by a crop, and calibration VWCs were as low as 0.16 cm^3 cm^{-3}. This value is not much different from the lowest recorded under a crop and suggests the model could adequately predict NNP under both fallow and crop. Soil nitrate–N concentrations were analyzed from soil core sampling that occurred during fallow sampling time periods, at ten locations within five subfields in the 0–15 cm depth and four locations in two subfields in the 0–15 and 15–30 cm depths to calculate 66 distinct NNP observations.

Net nitrate production model

R statistical software (The R Foundation for Statistical Computing, Vienna, Austria, 2013) was used to estimate a linear regression specification of NNP at the subfield level, which characterizes a more practical model for assessing and making insights about farm-level management decisions than NNP at specific 0.6 m^2 sampling locations. Existing literature describes a number of variables that influence N mineralization and denitrification rates; those cited most often include soil temperature (Stanford et al. 1973, 1975; Curtin et al. 2012), soil water content (Paul et al. 2003; Heumann et al. 2011) and soil TN (Vigil et al. 2002). Other variables discussed in the literature include crop residue C:N ratio (Booth et al. 2005; Patron et al. 2007), soil organic matter (Booth et al. 2005; Heumann et al. 2014) and previous crop (Soon and Arshad 2002). We modeled NNP as a function of previous crop, soil C:N ratio, depth to gravel contact, soil test K, clay content, pH, EC, soil temperature (T), VWC, and soil TN pool (kg N ha^{-1}). Post-estimation statistics determined that T, VWC, and TN provided the best fit to the NNP data (see Supplementary Material for details).

The selected model was then used to predict NNP in both the 0–15 and 15–30 cm depths for four distinct time periods each year (Aug/early Sept–Nov 14; Nov 15–Apr 30; May 1–June 30; July 1–Aug/early Sept). Net nitrate production in the upper 30 cm for each time period was summed to calculate an annual input for inclusion in the nitrate leaching mass balance equation. Although some mineralization likely occurs at greater depths, soil nitrate profiles from 1.4-m pits excavated after fallow indicated that more than 55% of N mineralization occurred in the upper 30 cm (unpub. data), likely in part due to much lower TN concentrations in lower depths and gravel contacts near 65 cm.

Nitrate leaching model

The nitrate leaching rate (Eq. 1) was estimated using a hierarchal mixed effects model with the *lmer* and *lme4* packages (Gelman and Hill 2007; Bates et al. 2015) in R statistical software. This estimation approach was chosen for its ability to account for unobserved or unquantifiable variability across fields within a particular production location (e.g., topographical heterogeneity at a farm) and across farm locations in the sample (e.g., differences in producers' farming decisions). We specified differences across fields and locations as random effects because we assumed that the size of the effect of unobserved factors could differ from one farm and one field to another. This specification provides for greater flexibility in generalizing the results, especially in light of the relatively modest sample size (Borenstein et al. 2010). Lastly, the hierarchal model design allows nesting each of the ten sample location pairs within each subfield pair, which further improves the ability to control for unobserved variability along the management interface. Because the second N application was not applied to the Split App subfield (A3) in 2014, we treated that subfield as a GSP with a lower N rate, rather than eliminating it, to maintain estimation power.

A separate model was constructed for each study year. Twelve of the 60 sampling locations from the 2013 winter wheat at Field A (Fig. 2) were excluded from further analysis based on negative leaching values, suggesting that winter wheat took up groundwater N from those locations. There was no evidence of groundwater N uptake by wheat in 2014 at Field A, and groundwater was too deep at Fields B and C to be an issue. Explanatory variables of interest were treatment, grower, growing season precipitation, N fertilizer amount, TN, SOC, soil pH, depth to gravel contact, coarse fraction, Olsen P, EC, and crop type

Table 4 Monthly precipitation amounts and temperature averages by water year (2012–2014) for study locations within the Judith River Watershed

Month	Agrimet (CARC)[a]			Field A			Field B			Field C			LTA[b]
	2012	2013	2014	2012	2013	2014	2012	2013	2014	2012	2013	2014	1981–2010
Precipitation (mm)													
Oct–Apr[c]	153	84	120	–	–	–	–	–	–	–	–	–	126
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[d]	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
May–July	74	197	123	124	187	77	92	241	131	116	343	391	181
Total	245	378	463	–	–	–	–	–	–	–	–	–	390
Temperature (°C)													
April	7	3	5	–	4[e]	6	–	4[e]	6	–	4[e]	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

[a] Central Agriculture Research Center (CARC) gauge is located 3 km west of Moccasin, MT

[b] The long term average (LTA) was obtained from the Western Regional Climate Center (WRCC) station at CARC

[c] Oct–Apr includes data from previous year (e.g., 2012 denotes Oct 2011–Mar 2012); no Oct–Apr data are shown for test fields because on-field rain gauges were removed in the fall and reinstalled in the spring

[d] The large Aug 2014 precipitation event occurred from Aug 23–24 which was after wheat reached maturity and prior to spring wheat and soil sampling on field A; all sampling on Fields B and C was conducted prior to this event

[e] Temperature average includes fewer days than a full month

(only in 2014). We also created a categorical interaction variable that accounts for unobservable differences in grower management and climate (Mgmt&Clim) across farms. After adjusting for available degrees of freedom based on paired location-fields, variables estimated to statistically explain variation in leaching were Mgmt&Clim, depth to gravel contact, clay, Olsen P, coarse fraction, EC, and crop (only in 2014). For a more detailed description of the modeling, see John (2015).

Economic assessment

We recognized that practices that reduce leaching will have a higher likelihood for adoption if they maintain or increase revenue. To assess net revenue for each tested management practice, an enterprise budget analysis was conducted (Supplementary Material; John 2015).

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–Sept 2012) in the JRW (Table 4). In the 2013 treatment year, Field A received similar growing season (May–July) precipitation (187 mm) to the long term average (LTA) of 181 mm, whereas fields B and C received 241 and 343 mm, respectively (Table 4). Much of the growing season precipitation occurred from late May to early June, making for ideal leaching conditions on soils with shallow gravel contacts. Above average precipitation occurred at the beginning of the 2014 treatment year (namely in September 2013), yet all fields received well below average precipitation during the growing season and Field A received ca. 40% less than the other fields.

Net nitrate production model

The selected NNP model included four explanatory variables (Eq. 3).

$$NNP = -0.72 + 0.063\,T - 0.0021\,T^2 + 0.65\,VWC + 0.096\,TN \quad (3)$$

where T is soil temperature in °C, VWC is cm^3 water cm^{-3} soil and TN is total N in upper 15 cm in $Mg\ N\ ha^{-1}$. Both T and TN were positively correlated with NNP ($p < 0.05$), consistent with other research (Cassman and Munns 1980; Vigil et al. 2002). The negative relationship between T^2 and NNP possibly reflects increased denitrification rates at higher temperatures (Stanford et al. 1975). Using the study's

Table 5 Mean 2013 inputs and outputs used in N balance to calculate nitrate leached in the Judith River Watershed of central Montana for post-pea, post-fallow, controlled release urea (CRU), conventional urea (CU), split application (Split app), and single broadcast urea (SB), all for winter wheat year

Treatment	Inputs (kg N ha^{-1})			Outputs (kg N ha^{-1})		
	Fertilizer	NNP[a,b]	Deposition[c]	Volatilization	Uptake[b,d]	Δ Nitrate–N[e]
Pea	83	61	1.6	9	128	−17
Fallow	83	77	1.6	9	156	−48
CRU	90	67	1.6	2	136	−25
CU	90	70	1.6	10	147	−37
Split App	91	71	1.6	8	132	−37
SB	90	71	1.6	10	120	−28

[a] NNP (net nitrate production, predicted) = net N mineralization–denitrification

[b] Each NNP and uptake value is the mean for two subfields and hence up to 20 locations. Values from 12 Field A locations (out of 60) were not included in means due to evidence of shallow groundwater

[c] N deposition includes both wet and dry

[d] N-uptake includes aboveground biomass measurement data and belowground estimates

[e] Δ Nitrate–N = post-harvest soil nitrate–N_{final}–post-harvest soil nitrate–$N_{initial}$

Table 6 Mean 2014 N inputs and outputs used in N balance to calculate nitrate leached in the Judith River Watershed of central Montana for winter wheat year post-pea and post-fallow, and spring wheat year with controlled release urea (CRU), conventional urea (CU), split application (Split App.), and single broadcast urea (SB)

Treatment	Inputs (kg N ha^{-1})			Outputs (kg N ha^{-1})		
	Fertilizer	NNP[a,b]	Deposition[c]	Volatilization	Uptake[b,d]	Δ Nitrate–N[e]
Pea	87	58	1.6	9	110	−13
Fallow	87	60	1.6	9	133	−32
CRU	90	61	1.6	6	81	−3
CU	90	64	1.6	10	85	8
Split App[f]	92	70	1.6	9	97	0
SB	92	64	1.6	11	96	−2

[a] NNP (net nitrate production, predicted) = net N mineralization–denitrification

[b] Each NNP and uptake value is the mean for two subfields and hence up to 20 locations

[c] N deposition includes both wet and dry

[d] N-uptake includes aboveground biomass measurement data and belowground estimates

[e] Δ Nitrate–N = post-harvest soil nitrate–N_{final}–post-harvest soil nitrate–$N_{initial}$

[f] The second application on Field A was not completed due to very dry conditions (Table 4), so results shown for split application and SB are only for Field C

average VWC ($0.25\ cm^3\ cm^{-3}$) and TN ($2.8\ Mg\ N\ ha^{-1}$), NNP becomes negative above 23 °C, suggesting denitrification exceeds net mineralization at higher temperature, consistent with other findings of net N loss under hot, dry conditions (Amundson et al. 2003). While VWC was only weakly related ($p = 0.12$) to NNP, this could simply be a result of a relatively small number of independent VWCs (n = 5) to provide sufficient power to adequately capture the effect.

Although the four independent variables only explained 18% of the variability in the 66 NNP measurements, overall, the NNP model explained most of the variability in NNP when averaged across each subfield-period ($R^2 = 0.81$). This result is more relevant, because it was more important and practical to be able to accurately estimate NNP, and hence leaching, for a particular subfield and treatment than at a particular location within a subfield. Moreover, any changes in producers' management strategies would not be made at the scale of a sampling location (ca. 1 m^2), but would be more likely made at a substantially larger scale. The average in-sample predicted NNP among the seven subfield-periods was $0.13\ kg\ N\ ha^{-1}\ day^{-1}$ and average absolute difference from measured NNP was $0.03\ kg\ N\ ha^{-1}\ day^{-1}$. For the seven subfield-periods, the average high squared forecast error (n = 2) was 0.020 which was 60% higher than the average low squared forecast error (n = 5). When the two worst (i.e., highest squared forecast error) models were used to estimate nitrate leaching, none of the three AMP versus GSP treatment effects changed for either year, providing confidence that any errors in NNP prediction did not affect our final conclusions on treatment effects (discussed below).

N Budget inputs and outputs

Fertilizer was the largest N input, averaging approximately $90\ kg\ N\ ha^{-1}$ in both years (Tables 5, 6). The average predicted NNPs under wheat were ca. 70 and ca. $62\ kg\ N\ ha^{-1}$ in 2013 and 2014, respectively, or about 75% of fertilizer N inputs. A study in Saskatchewan Canada, also in the NGP, found $54–70\ kg\ N\ ha^{-1}$ of net N mineralization occurred from harvest through the following wheat growing season (Campbell et al. 2008), reasonably coinciding with our mean NNP estimates. Winter wheat N uptake was very high in 2013, averaging $136\ kg\ N\ ha^{-1}$, due to nearly ideal growing conditions that resulted in grain yields of $3.4–4.6\ Mg\ ha^{-1}$. Winter wheat N uptake in 2014 was nearly as high ($110–133\ kg\ N\ ha^{-1}$) as in 2013, yet spring wheat, grown on the fertilizer treatment fields following winter wheat, was drought-stressed resulting in low yields ($1.9–2.6\ Mg\ ha^{-1}$) and N uptake (ca. $80\ kg\ N\ ha^{-1}$). Soil nitrate–N changes (final–initial) during each wheat year ranged from $-17\ kg\ N\ ha^{-1}$ (in pea treatment) to $-48\ kg\ N\ ha^{-1}$ (in fallow treatment) in 2013, and were close to 0 for all

Table 7 Model predicted nitrate leaching for each management practice for 2013 and 2014 treatment years in the Judith River Watershed

Comparison	Fields	2013 Nitrate leached[a] ($kg\ N\ ha^{-1}$)		2014 Nitrate leached[a] ($kg\ N\ ha^{-1}$)	
		AMP	GSP	AMP	GSP
Post-pea-Post-fallow[b]	A, B	18* (2.5)[c]	54 (3.6)	39 (10)	38 (7.0)
CRU-CU[d]	A, C	46 (3.6)	50 (3.9)	69 (6.1)	53 (6.8)
Split App[e]-SB[d]	A, C	61 (3.6)	55 (4.4)	58 (10)	54 (6.7)

* Significantly different ($p < 0.05$) than the GSP

[a] Calculating nitrate leached from Tables 5 and 6 will produce different values than predicted here because values here include effects from environmental variables that varied across interfaces

[b] Post-Pea and -fallow leached amounts are for the following winter wheat year

[c] Numbers in parentheses are standard deviations

[d] *CRU* controlled release urea, *CU* conventional area, *SB* single broadcast urea

[e] In 2014, split application was not completed on Field A, so only Field C results shown for that treatment-year

fertilizer treatments in 2014. Volatilization and deposition estimates were consistently much smaller than the other inputs and outputs. The dominance of plant N uptake and fertilizer N in the N balance, both of which likely had relatively low error, minimizes uncertainty around the N leaching estimate (Meisinger and Randall 1991).

Nitrate leaching results

Among the six nitrate leaching comparisons, only one treatment difference was observed (Table 7); namely, in the 2013 treatment year, less nitrate leached during the winter wheat year post-pea (18 ± 2.5 kg N ha^{-1}) than from winter wheat post-fallow (54 ± 3.6 kg N ha^{-1}). Reduced leaching after pea was likely a result of both lower soil nitrate and lower VWC after pea than after fallow. Specifically, 2012 post-harvest soil nitrate pools were on average 16 and 59 kg N ha^{-1} for pea and fallow, respectively. Also, soil VWC was close to wilting point at pea harvest (ca. 0.16 cm^3 cm^{-3}) and field capacity on adjacent fallow subfields (ca. 0.28 cm^3 cm^{-3}); this would have reduced deep percolation and N mineralization rates more post-pea than post-fallow. Although there is a recognized degree of uncertainty in the results given the number of assumptions needed when using a N mass balance approach, these results are broadly consistent with our companion study (Sigler et al. in review), which found that mean lysimeter nitrate concentrations at the gravel contact were significantly lower ($p < 0.05$) the year after a crop (8.6 ± 7.3 mg L^{-1}) than the year after fallow (25.4 ± 14.0 mg L^{-1}) during the 2-year study, helping confirm that fallow increases leaching potential. Our findings are also consistent with results from a much longer term study in Saskatchewan Canada (Campbell et al. 2006), demonstrating that fallow replacement could have both short- and long-term benefits on water quality.

In contrast to 2013, no leaching difference was observed between wheat post-pea and wheat post-fallow during the 2014 treatment year (Table 7). This may reflect that soil nitrate–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} at start of 2013 treatment year). This was in turn likely due to precipitation in the first month of the 2014 treatment year (Sept 2013) averaging about two times the long-term average, compared to <6% of average for Sept 2012 (Table 4). Abnormally high soil moisture going into the 2014 treatment would have initiated mineralization of pea residue and soil organic matter, bringing the amount of nitrate available to leach from pea fields much closer to that from fallow than in the previous year.

In both treatment years, neither fertilizer AMP had an effect ($p = 0.05$) on nitrate leaching. While CRUs have been found to decrease leaching in other regions (Mikkelsen et al. 1994; Nakamura et al. 2004) and under irrigated conditions (Wilson et al. 2010), the CRU benefit in our study may have been negated due to N application timing differences between treatments. Notably, CRU was applied with the seed about 6 months prior to the GSP spring broadcast urea application in 2013. The placement and timing of the CRU AMP and urea GSP were not the same because fall-applied urea was not a standard practice (due to shallow gravel contact) and CRU can release too slowly to benefit wheat if spring broadcast in the NGP (McKenzie et al. 2007). The split application may not have been effective at reducing leaching in 2013 because on Field C the second application was completed on 23 May, immediately prior to 180 mm of precipitation in the next 12 days, enough to move substantial UAN below the root zone before it could be used by the crop. In the 2014 treatment year, the majority of N had leached by 30 April on Field C (John 2015), which was prior to urea application in both the GSP and AMP treatments, minimizing the potential for the split application to reduce leaching. In the semiarid NGP, effects of CRU and split application practices on leaching rates have not been published to our knowledge. Our results suggest that alternative N fertilizer practices can only be effective at reducing leaching if substantial leaching occurs after a standard single fertilizer N application, yet precipitation rates are not so high that they negate the AMP benefit on soils with shallow gravel contacts. Given that the primary precipitation period in most of Montana and some adjacent regions is from late April to early June, fertilizer N is generally applied by mid-April, and 2013 had spring precipitation well above the long-term average, it's possible that the fertilizer AMPs could reduce leaching in other weather years.

Estimated annual rates of nitrate leached during wheat years over the 2-year study of 18 ± 2.5–69 ± 6.1 kg N ha^{-1} (Tables 5, 6) were substantial considering that the average annual fertilizer input was

Table 8 Explanatory variable coefficients for 2013 and 2014 nitrate leaching models in the Judith River Watershed. Soil parameters, except depth to gravel contact, were from 0 to 15 cm cores

Parameter	2013	2014
Mgmt&Clim[a] (B)	17	−23* (−42, −4.5)[c]
Mgmt&Clim (C)	20	2.7
Crop (WW)	NI[b]	−6.8
Depth to Gravel Contact (cm)	−0.19	−0.29* (−0.44, −0.15)
Clay (g kg^{-1})	0.46	−0.95
Olsen P (mg kg^{-1})	−0.15	0.001
Coarse Fraction (kg kg^{-1})	−21	30
Salts (dS m^{-1})	−17	14

* Significant at $p < 0.05$

[a] Mgmt&Clim = Categorical variable that would account for confounding variables such as climate, seeding date, cultivar, etc.; Mgmt&Clim coefficients for Fields B and C were in comparison to Field A Mgmt&Clim effects

[b] NI (not included); winter wheat (WW) parameter not included in 2013 because winter wheat was only crop grown in 2013, whereas both winter and spring wheat were grown in 2014

[c] 95% confidence intervals for significant variables are in parentheses

87 kg N ha^{-1}, but less startling when both NNP and fertilizer were summed (ca. 150 kg N ha^{-1}). For comparison with our leaching rates, a companion study (Sigler et al. in review) found that the average N flux from soil to groundwater for the cropland-dominated Moccasin Terrace (Fig. 1) from 2012 to 2014 was approximately 10–20 kg N ha^{-1} $year^{-1}$. That work quantifies leaching rates with observations in groundwater, so values are integrated over time and space (26,000-ha landform), and are likely closer to an annual average N leaching rate across years and management. Leaching rates were likely elevated in our current study compared to landform leaching rates because of above average May–June precipitation in 2013 (121–225% of LTA) after fallow, and hence high soil nitrate levels, and above average precipitation from Sept 2013–Mar 2014 before spring wheat N uptake started (Table 4). Thus the rates determined here provide a reasonable measure of leaching loss for these wheat fields in these precipitation years, within these cropping systems. Most importantly, they indicate the vulnerability of these soils to low NUE when both fertilizer and mineralization are considered. Although nitrate leaching rates reported here were likely above average, the estimated rates likely were not overly rare, because precipitation amounts during the study were within normal ranges. Although amounts during the study were above average for key periods, from 1983 to 2012, there were six May–June periods wetter than 2013 (WRCC gage, Moccasin).

In the 2014 treatment year, depth to gravel contact was inversely related to nitrate leaching (Table 8). The effect of depth to gravel contact on leaching was considerable; notably, just a 60 cm difference in depth to gravel contact was associated with a predicted 17.4 ± 8.7 kg N ha^{-1} difference in leaching. This finding not only points out the threshold response of gravel depth, it also suggests that that there was still substantial leaching (ca. 20 kg N ha^{-1}) after pea on the deepest gravel contacts (ca. 125 cm), indicating fallow replacement alone won't eliminate leaching. Conversely, on the shallowest gravel contacts (ca. 30 cm), leaching from GSP management could exceed 70 kg N ha^{-1}. This result demonstrates the critical importance of identifying fields and areas within fields that have the shallowest gravel contacts to better inform appropriate management decisions, such as adoption of multiple, integrated AMPs, or planting perennials. Aerial imagery of crop greenness, calibrated with push probing, has accurately predicted locations within Field B that have gravel contact depths <45 cm (Sigler et al. unpub data).

In the 2013 treatment year, depth to gravel contact did not influence nitrate leaching, likely because a bulk of the May–June rain came in a very short period from late May to early June, minimizing the ability of winter wheat roots to take up nitrate before it reached gravel, even in soils with greater depths to gravel. Model error was small (1.7 kg N ha^{-1}) relative to predicted leaching rates (18–69 kg N ha^{-1}), indicating the model was quite accurate for the purpose of assessing treatment effects. Although there is uncertainty with each of the terms in the leaching equation, for the most part, uncertainty should be similar across a management interface (e.g., volatilization should be similar on post-pea and post-fallow subfields since the same N amount was applied). Therefore, we have more confidence in treatment differences than in absolute leaching amounts.

Although we only tested one treatment that was found likely to reduce deep percolation, replacement of fallow with any crop or converting from annual cropping to perennial cropping is expected to decrease deep percolation and nitrate leaching based on previous work. For example, a meta-analysis by Tonitto et al. (2006) found legume and non-legume cover crops both reduced nitrate leaching amounts compared to fallow, by 40 and 70%, respectively. In semiarid regions, perennial grass ecosystems have been found to retain >90% of added N in the long term (15 years) compared to only 30% in annually cropped ecosystems (Mobley et al. 2014).

Our results strongly indicate that fallowing can result in substantial nitrate leaching because mineralization is out of phase with N uptake, and deep percolation rates increase during and immediately after fallow. Our results further indicate that although continuously cropped systems can leach less than fallow-wheat systems, there was still substantial leaching loss from pea-wheat and continuous wheat. This suggests that conversion to perennials, especially in the most vulnerable locations, will likely be necessary to reduce nitrate loss and eventually meet the EPA drinking water standard in JRW groundwater. While somewhat uncertain, current climate models (Walsh et al. 2014) predict warmer, wetter winters and springs, and hotter, drier summers. These conditions are conducive to enhanced mineralization of organic matter, increased leaching in spring, and increased motivation toward fallowing as perceived drought protection. As a consequence, we see potential for further nitrate leaching to groundwater and potential soil degradation if management changes toward reduced fallow, cropping system diversification, and seeding perennials in the most vulnerable locations are not adopted.

Net revenue

The enterprise budget determined that pea-winter wheat net revenue (\$243 ± 39 ha^{-1}) was higher than fallow-winter wheat net revenue (\$160 ± 33 ha^{-1}) in 2013–2014 (John 2015). Conversely, the fertilizer AMPs had no effect on net revenue in either study year and replacing fallow with pea did not affect net revenue in 2012–2013 (John 2015). The differences in net revenue results between years for the fallow–pea comparison were likely a direct result of precipitation differences between years. Notably, 2012 winter wheat seeding conditions were much worse after pea than fallow due to a very dry September (<2 mm), whereas seeding conditions after pea and fallow were similar in 2013 due to a wetter than average September (ca. 75 mm). Lack of economic benefit from the fertilizer AMPs was possibly caused by the intensity of late May/early June rains in 2013 and drought in 2014 (making water, not N, limiting to yield), as discussed above, combined with additional costs of these AMPs.

Conclusion

This study provides empirical evidence that management practices that decrease both deep percolation and soil nitrate levels, such as fallow replacement, can increase the likelihood of reducing nitrate leaching, whereas fertilizer timing and source management practices may be less effective because they do not substantially decrease deep percolation. Nonetheless, fallow is still common in parts of the NGP, in part because replacing fallow is perceived to reduce economic returns, despite evidence that pea-wheat often produces higher net revenue than fallow-wheat (Zentner et al. 2001: Miller et al. 2015), consistent with our findings. The combined leaching and revenue results demonstrate that fallow replacement with pea can reduce leaching without decreasing profit when cropping system is considered as a whole. These results are highly relevant for local farmers who list maximization of yields as a top management priority when fertilizing with N, but for whom consideration of leaching has gained increased weight in management decisions during the period of this project (Jackson-Smith et al. 2016).

Our study results also demonstrated that leaching was highly dependent on the depth of gravel contact, indicating that it would be most beneficial to target fields and areas within fields that have the shallowest gravel contacts for management changes. Future nitrate leaching research in agricultural areas with shallow gravel contacts and shallow groundwater should focus on practices that reduce potential for deep percolation (e.g., fallow replacement, perennials) while maintaining or increasing economic returns to increase the adoption potential of these practices.

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