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Changes in runoff chemistry and soil fertility after multiple years of cattle winter bale feeding on annual cropland on the Canadian prairies

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A B S T R A C T

Feeding cattle on cropland through the winter months using pre-placed fodder bales (bale grazing or in-field bale feeding) has demonstrated economic benefits when compared to confined winter feeding, but very little research has been done to quantify the impacts of recurring cycles of in-field feeding on runoff water quality. In this study, two small watersheds (with and without winter bale grazing; WBG) located in the same annual cropped field in southern Manitoba, Canada were monitored from 2008 to 2016. Winter feeding in the WBG treatment occurred in the winters of 2008/09, 2010/11, 2012/13, and 2014/15. Export of N and P with the WBG treatment was also contrasted with the alternative practice of higher density confined feeding (CF). For WBG, soil fertility was improved, with higher soil test phosphorus (Olsen-P) and soil nitrate (NO_3^--N) following WBG. Exports of N and P with surface runoff were also higher in snowmelt following WBG when compared to the control watershed, primarily due to higher concentrations of particulate N and P, NH_4^+-N, and dissolved organic N. Higher P loss from the WBG watershed than from the control watershed persisted in the non-grazing years following each bale grazing. In contrast, concentrations of all forms of nitrogen in runoff consistently returned to levels observed prior to bale grazing following a winter without treatment. Annual runoff export of N and P per animal unit day (AUD) following WBG was comparable to that for two CF sites. Higher annual volumes of runoff were observed from the larger area and lower density bale feeding treatment, but with lower concentrations of N and P in comparison to feedlot sites. If WBG continues to be utilized to improve soil fertility and reduce manure transportation costs, the potential negative impacts on runoff water quality must be considered. Higher volumes of snowmelt runoff per AUD from WBG in comparison to CF sites will make runoff capture options more expensive to implement. WBG timing or siting changes that reduce runoff volume and make retention feasible or that decrease accumulation of P at the soil surface will reduce negative water quality impacts associated with WBG.

1. Introduction

In agricultural regions globally, phosphorus (P) and nitrogen (N) losses during runoff events are recognized as contributing to eutrophication of downstream aquatic ecosystems (Carpenter et al., 1998; Correll, 1998; Sharpley et al., 2013; Withers and Lord, 2002). Through most of the Northern Great Plains, runoff from cropland is the primary source of N and P exported with runoff (Bourne et al., 2002). Increasingly frequent cyanobacterial blooms in waterbodies of the region (McCullough et al., 2012; Schindler et al., 2012) increase impetus for the development of regionally appropriate agricultural management practices to decrease N and P runoff losses.

The practice of using winter bale grazing (WBG) to feed cattle, where fodder bales are placed throughout target fields for feeding during winter, is an alternative to confinement and feeding in the farmyard (Supplemental Fig. S1). It has gained popularity due to a number of agronomic benefits including reduced costs associated with manure and feed transport when compared to confined winter feeding (Jungnitsch, 2008; Kelln et al., 2011; Nayigihugu

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et al., 2007). Positive effects on soil fertility have also been observed (Flores and Tracy, 2012; Jungnitsch et al., 2011; Kelln et al., 2012). Despite these benefits of bale grazing, the potential for N and P loss with snowmelt runoff remains a largely unaddressed environmental problem. The western Canadian Prairies, typical of cold climate regions, are characterized by long snow-covered period in winter (up to 5 months per year), leading to accumulation of snow packs and repeated freezing-thawing cycles during spring snowmelt (van Bochove et al., 2001). Spring snowmelt runoff often accounts for a majority of annual runoff in the region and transports the majority of nutrient losses with runoff from agricultural land (Cade-Menun et al., 2013; Han et al., 2010; Liu et al., 2014b; Su et al., 2011). The materials deposited over a frozen soil surface or the snowpack during bale grazing (uneaten feed, straw bedding, urine, and dung) may act as significant nutrient sources in runoff water (Cade-Menun et al., 2013; Elliott, 2013; Roberson et al., 2007).

Uneaten feed, straw bedding, and animal wastes that remain on the soil surface following bale grazing can be regarded as similar to winter manure application. However, differences between dung deposited with grazing and manure as potential sources of P have been identified, with losses from dung generally being lower (Vadas et al., 2015). In many regions with temperate climates, the timing of manure application is strictly regulated to avoid application on snow covered or frozen land (Srinivasan et al., 2006), due to increased exportation risks during snowmelt (Klausner et al., 1976; Sharratt et al., 2000; Williams et al., 2011; Young and Mutchler, 1976). Frozen ground decreases soil infiltration, and low biological activity due to low temperature allows little utilization, decomposition, and transformation of applied N and P, which increases risk for transport with early spring runoff (Klausner et al., 1976; Sharratt et al., 2000; Young and Mutchler, 1976). Losses of N which occur during storage and transport from confined feeding operations to the field are eliminated with direct deposition in field. After deposition, low temperatures reduce ammonia volatilization losses and provide little chance to convert organic N, urea, and ammonia to nitrate (Smith et al., 2011). Moreover, dung, uneaten feed and straw bedding, which remain after grazing, not only improve soil quality, but also could be an important source of soluble nutrients due to freeze-thaw cycles during early winter and spring months (Liu et al., 2014a; Roberson et al., 2007).

Winter applied manure can impact the hydrological processes controlling snowmelt runoff through reducing albedo and promoting earlier thawing with increases in soil temperature (Kongoli and Bland, 2002; Williams et al., 2012a,b; Young and Mutchler, 1976), or by increasing insulation and retarding snowmelt (Kongoli and Bland, 2002). Regardless of impact on melt timing, manure lying beneath, in, or on the snowpack creates the potential for nutrient leaching to snowmelt runoff (Young and Mutchler, 1976). Further evaluation is required to identify whether similar problems are present in association with the residue material following bale grazing.

Overwintering cattle in confined feeding sites (CF; also referred to as feedlots) does allow for collection of manure for application to cropland during non-winter months followed by incorporation of that manure by tillage in order to reduce nutrient runoff losses with snowmelt in the following spring (Gilley et al., 2002). However, nutrient runoff from CF sites is also an environmental problem in many watersheds on the Canadian prairies where overwintering sites have historically been selected based on efficiency of surface drainage or proximity to the farmyard, but where less consideration has been given to prevention of runoff generation. Interception and storage of runoff from cattle overwintering facilities can effectively reduce nutrient losses with runoff (Li et al., 2011) and this practice is implemented by many producers where incentives or regulation encourage implementation; however, it is still common for uncontrolled runoff to leave CF sites used to overwinter smaller cow-calf herds on the Canadian Prairies.

Agronomic benefits associated with WBG have led to its adoption by cattle farmers. Preliminary research on pasture land in Saskatchewan found high concentrations of orthophosphate-P and ammonium-N in snowmelt runoff (Smith et al., 2011). To reduce the potential for N and P losses in runoff following WBG, the impact of the practice on hydrology, sediment and nutrient losses needs to be understood. There has been research on runoff nutrient losses from extended grazing and WBG on pastures (Sanderson et al., 2011; Vadas et al., 2015), but the potential benefits and impacts of WBG on cropland has not been studied, where tillage can be utilized to incorporate material deposited on the soil surface. It is also unclear what the impacts of multiple cycles of WBG at the same location are likely to be and how the impacts of WBG compare to CF sites. To fill these knowledge gaps, we studied bale grazing on annual cropland in western Canada between 2008 and 2016 covering three winters with in-field grazing. Our goals were (1) to evaluate the impacts of in-field WBG on N and P soil test levels and losses in surface runoff from annual cropland in western Canada, (2) to assess temporal trends associated with multiple years of WBG, and (3) to compare rates of N and P export with runoff from WBG to feedlot sites.

2. Materials and methods

2.1. Study area

For this study runoff was monitored from two small watersheds (WBG treatment – 2.86 ha and control – 2.80 ha) located within the same annually cropped field (98.38°W, 49.32°N). Runoff was also monitored from two CF sites used to overwinter cattle in those years with WBG treatment. The watersheds draining the two CF sites are 0.17 ha and 1.85 ha in area. All sites are located in the South Tobacco Creek (STC) Watershed, a sub-watershed of the Red River in southern Manitoba, Canada (Supplementary Fig. S2). The climate is semiarid and cold temperate, with an average annual precipitation of ~550 mm (of which 25–30% falls as snow), and an average annual temperature of ~3 °C (Environment Canada, 2015). Soils in the study field are of the Dezwood series and are Dark Grey Chernozems (Boralfic Borolls) formed on moderately to strongly calcareous glacial till overlaying fractured shale bedrock (Soil Classification Working Group, 1998).

2.2. Field and feeding site management

During the study period (2008–2016), both WBG treatment and control watersheds were managed identically (fertilizers, crops, tillage; Table 1) with the exception that in the WBG treated watershed cattle were fed in the early winter (November and December) in every second year. In general, the annual crop grown in the study field alternated between wheat and canola, fertilized with a mixed blend of N, P, and S applied at seeding side band, and supplemented with additional N applied in the spring. The watersheds were tilled consistently each spring and more infrequently in the fall to incorporate residue (Table 1). In upper slope locations tillage occurred in parallel with the flow of water to prevent spreading of residue from treatment to the control field. Number of cows fed in WBG years, bale type and number (alfalfa/brome grass and native grass), and duration of feeding (usually mid-November–mid-December) are presented in Table 2. Further details on type of cattle, body masses, and duration of feeding are noted for both WBG and CF sites in Supplementary Table S1.

Table 1
Summary of field management and fertilizer input.

Year	Crop	Season	Fertilizer application	Tillage	Tillage index*	Crop residue management	Yield (kg ha^{-1})
2007	Canola	Fall	84.1 kg ha^{-1} banded with cultivation	heavy duty cultivator (1 pass)	0.6	straw chopper	1961
2008	Wheat	Spring	11.2 kg ha^{-1} N banded with seeding 3.05 kg ha^{-1} P banded with seeding 11.2 kg ha^{-1} S banded with seeding	light duty cultivator (1 pass)	0.7		
		Fall		none	1	straw chopper	3493
2009*	Canola	Spring	5.6 kg ha^{-1} N banded with seeding 31.9 kg ha^{-1} P banded with seeding 22.4 kg ha^{-1} S banded with seeding 112.2 kg ha^{-1} N broadcast (101 kg ha^{-1} N in treatment watershed)	heavy duty cultivator (1 pass) then light duty cultivator (1 pass)	0.42		
		Fall		heavy duty cultivator (1 pass)	0.6	straw chopper	2802
2010	Wheat	Spring	5.6 kg ha^{-1} N banded with seeding 3.05 kg ha^{-1} P banded with seeding 11.2 kg ha^{-1} S banded with seeding	light duty cultivator (1 pass)	0.7		
		Fall		heavy duty cultivator (1 pass)	0.6	straw chopper	3363
2011*	Canola	Spring	89.7 kg ha^{-1} N banded with cultivation 16.8 kg ha^{-1} N banded with seeding 31.9 kg ha^{-1} P banded with seeding 22.4 kg ha^{-1} S banded with seeding	light duty cultivator (1 pass)	0.8		
		Fall		heavy duty cultivator (1 pass) and harrow (1 pass)	0.48	straw chopper	1401
2012	Wheat	Spring	95.4 kg ha^{-1} N banded with cultivation 6.7 kg ha^{-1} N banded with seeding 3.05 kg ha^{-1} P banded with seeding	light duty cultivator (1 pass)	0.7		
		Fall		none	1	baled	4035
2013*	Canola	Spring	101 kg ha^{-1} N banded with cultivation 11.2 kg ha^{-1} N banded with seeding 31.9 kg ha^{-1} P banded with seeding 16.8 kg ha^{-1} S banded with seeding	light duty cultivator with harrows (2 passes)	0.49		
		Fall		none	1	straw chopper	2522
2014	Wheat	Spring	95.4 kg ha^{-1} N banded with cultivation 11.2 kg ha^{-1} N banded with seeding 3.05 kg ha^{-1} P banded with seeding	light duty cultivator (1 pass)	0.7		
		Fall				straw chopper	3699
2015	Canola	Spring	112.2 kg ha^{-1} N banded with cultivation 11.2 kg ha^{-1} N banded with seeding 31.9 kg ha^{-1} P banded with seeding 16.8 kg ha^{-1} S banded with seeding	Anhydrous Rig (1 pass) light duty cultivator (1 pass)	0.56		
		Fall		heavy duty cultivator (1 pass)	0.60	straw chopper	2242

Tillage index*: the greater of tillage index, the less tillage intensity and the more residues saved. The calculations of tillage index refer to Khakbazan and Hamilton (2012).

Table 2
Management details for winter bale grazing (WBG) treatment watershed.

Year	Feeding Duration	Number & type of Cattle	Bales Fed	N input (kg)	P input (kg)
2008	Nov-12–Dec-12	62 dry cows & 1 calf	40 round alfalfa/brome (500–545 kg), 30 native grass (545–590 kg), 10 wheat straw bales (as bedding)	758.6	46.6
2010	Nov-23–Dec-22	63 cows for 15 days & 62 cows for 15 days	30 alfalfa (767 kg, 2009 production), 19 native grass (684 kg), 20 2nd cut alfalfa (567 kg, 2010), and 13 wheat straw bales (450 kg)	1013.5	85.6
2012	Nov-20–Dec-19	62 cows for 30 days	30 1st cut alfalfa (734 kg, 2012), 30 native grass (724 kg, 2012), 3 2nd cut alfalfa (572 kg, 2012), and 13 wheat straw (470 kg, 2012)	886.6	70.9
2014	Nov-15–Dec-15	62 cows for 30 days	31 bales of silage alfalfa (750 kg), 33 bales of native grass (726 kg), and 10 bales of straw (454 kg)	981.9	94

Runoff from CF sites was monitored in the same years as WBG treatment occurred (2009, 2011 and 2013). For both the CF sites and the WBG watershed the number of animal unit equivalents and days of feeding were utilized to calculate animal unit days (AUDs) as a metric of intensity. The convention of 1 AU per 454 kg of weight in cattle was utilized to make these calculations. In CF site 1 (located within a 1.85 ha watershed) a larger proportion of overwintering AUs were cows and for CF site 2 (located in a 0.17 ha watershed) a greater proportion of the herd was composed of yearlings being fattened for market (Supplementary Table S1). A weight of 363 kg over the course of the year was assumed for yearlings being fattened based on the mean of starting and ending weight estimates reported by the producers. A greater proportion of the feed ration for cattle being fattened was composed of grain rather than other forms of fodder; however, total inputs of straw, grain, and hay were not quantified at the two CF sites with inputs varying depending on weather and animal condition. Manure was removed from CF sites by the producer each fall.

2.3. Quantification of annual N and P inputs at cropland sites

Inputs of nutrients to the WBG watershed consisted of feed, bedding, and fertilizer. The masses of N and P imported into the WBG watershed in the bales and bedding (Supplementary Table S2) were estimated based on average bale weight, number and types of bales fed, and analysis of bale composition in each year. Each year a minimum of two bales of each type were weighed to calculate the average weight. At least one sample of each bale type in each year was sent for analysis to the Central Testing Laboratory ltd. (Winnipeg, MB) that utilized Association of Official Analytical Chemists (AOAC) standard methods for the determination of crude protein and P content (AOAC 935.13A, 968.08, and 990.03).

2.4. Soil and water sampling

Each fall, soil samples were collected at 16 sampling points along three transects covering the hillslope of each watershed (Supplementary Fig. S3). Samples were sectioned into 0–6 cm, 0–15 cm and 15–60 cm depths for Olsen-extractable soil P; and 0–15 cm and 15–60 cm for measurement of KCl extractable soil NO_2^- plus NO_3^--N (Soil NO_x^--N) (AgVise Laboratories, Northwood, ND). Additional soil samples were collected once during summer in 2011, 2013, 2014 and 2015.

For the control and WBG watersheds, event-based water sample collection at the outlet of each watershed was initiated during snowmelt runoff in 2008 and discharge monitoring began in the spring of 2009. The depth of water flowing over V-notch weirs installed at the watershed outlets was measured every 5 min with ultrasonic sensors (SR50, Campbell Scientific) and incremental changes in flow, as calculated based on weir geometry, were used to trigger autosamplers (Sigma 800SL). Depth was also measured manually during periods with flow to ensure accuracy of automated measurement. All water samples were analyzed using standard techniques for ammonia (NH_4^+-N), nitrate + nitrite (NO_x^-—N), total N (TN), total P (TP), total dissolved P (TDP), and mass of suspended sediment (TSS) as described in Li et al. (2011). These same methods were utilized for monitoring downstream of CF sites (Supplementary Fig. S1) where v-notch weirs were previously installed in support of a parallel project to measure effectiveness of downstream holding ponds. Temperature and snowfall data were provided from a nearby Environment Canada station (Miami-Thiessen, 98°15′W, 49°27′), and rainfall was monitored on site using a tipping bucket rain gauge (Supplementary Table S3).

2.5. Data preparation and statistical analyses

Water chemistry data were available from 2008 to 2016, but flow data were missing for 2008 for the control and WBG

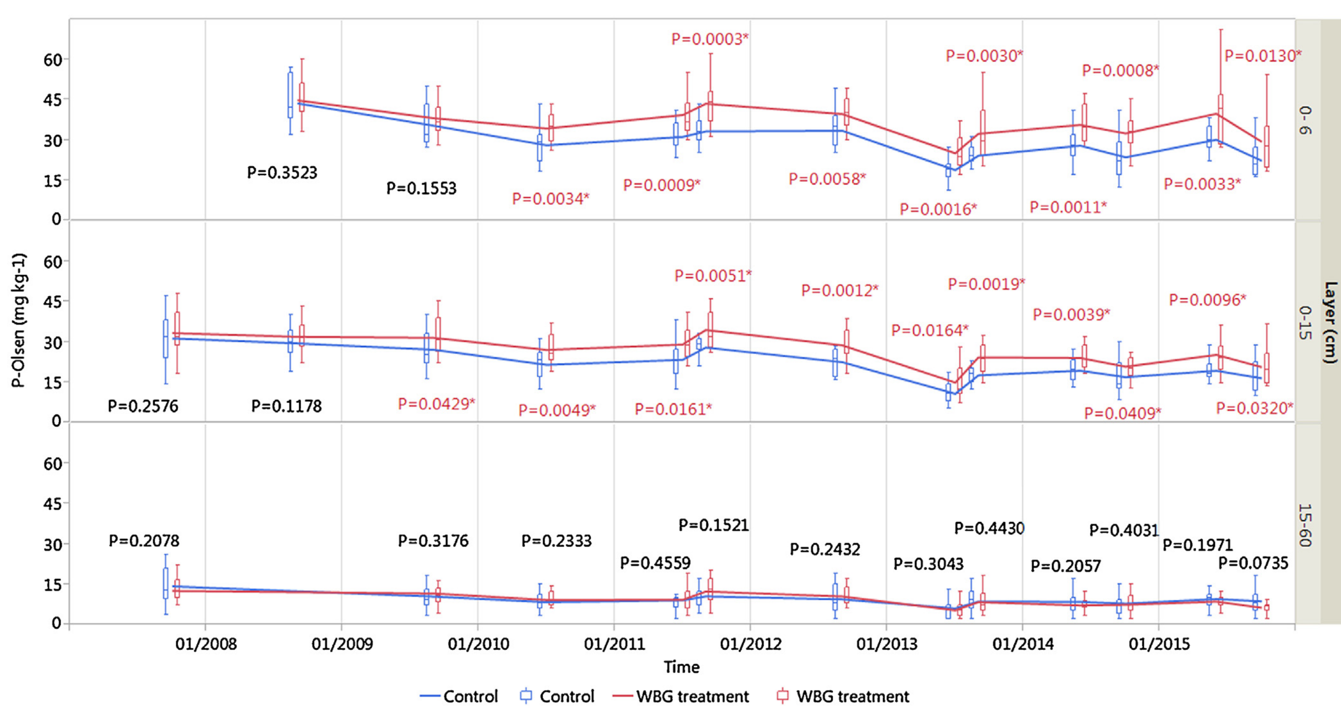

Fig. 1. Soil test Olsen-extractable P of the winter bale grazing watershed (WBG) and the control watershed. Lines connect the mean for each sampling date. For each sampling date, error bars show the range of data and box plots show 25th, 50th, and 75th percentiles of the data.

watersheds (prior to weir installation). Instrument malfunctions led to flow data loss for the WBG treatment watershed in 2014. For all small watersheds monitored, runoff events were separated by discontinuous flow (e.g., zero discharge over a few days) and hydrograph pattern. Data were grouped for calculation of flow and export annually, seasonally (snowmelt and rainfall) and by event. To calculate exports and flow-weighted mean concentrations (FWMCs), linear interpolation between sampling times was used to estimate concentrations at 5-min intervals (Tiessen et al., 2010). Sediment and nutrient exports were calculated as the product of 5-min flow volumes (L) and concentrations (mg L^{-1}) summed over the duration of each event, season, or year. FWMCs were calculated as export divided by flow volume.

All data was checked for normality prior to analysis and where transformation was necessary it is noted in the text; however, all data were back-transformed prior to reporting. T-tests were used to compare means of soil test N and P between the control watershed and the WBG treatment watershed for each soil depth and sampling time period, with significance level set at $a = 0.05$. Temporal trends in mean soil test N and P at all depths between control and WBG watersheds were examined on an annual time-step.

Despite a significant correlation between seasonal and annual water yields in the control and WBG watershed, variability in the water yield relationship was observed for individual runoff events of lower volume (Supplementary Fig. S4) and with differences in runoff timing. As a result, a clearly matched pair of measurements for each event could not always be defined. For this reason, ANCOVA analyses to evaluate differences in nutrient export between control and WBG watersheds on an event time step were based on differences in the linear relationship between FWMC or export and water yield in each watershed rather than making comparison for only the subset of the data where clearly matched events occurred.

Temporal trends in differences in snowmelt runoff FWMC N and P between watersheds were also examined on an annual time-step. For 2008 and 2014 FWMCs could not be calculated, so arithmetic mean concentrations of all samples are presented with the FWMCs. On dates where flow was recorded, the event-based FWMCs and arithmetic mean concentrations were highly correlated for all analytes (Supplementary Fig. S5). Sampling frequency and timing were based on flow conditions and for this reason it is assumed that mean concentration reflects FWMCs well in both 2008 and 2014. Arithmetic mean concentrations for all years were also utilized in nonparametric Wilcoxon/Kruskal–Wallis tests to compare between control and treatment and among years to provide an additional comparison of runoff nutrient concentration changes between the pre-treatment and post-treatment years.

3. Results and discussion

3.1. Soil test phosphorus and nitrogen

Prior to the first bale grazing (November of 2008), no statistically significant difference in extractable soil P (Olsen-P) was observed between the WBG treatment watershed and the control watershed at any depth (Fig. 1). After the first bale grazing, Olsen-P was significantly higher in the WBG watershed for the 0–15 cm sampling depth and remained higher over the course of the study. Significantly higher Olsen-P was also observed in the WBG watershed at the 0–6 cm depth beginning in 2010 and remaining higher throughout the study (Fig. 1). Olsen-P at 15–60 cm was not significantly different between treatments. Despite incorporation of surface residue with annual cultivation extractable P at the soil surface was increased in the WBG watershed, but no accumulation below 15 cm was detected.

Soil NO_x^--N was more variable than Olsen-P over the course of the study (Fig. 2), reflecting the greater mobility of NO_3^--N than P in cropland soils (Liang et al., 2014; Syswerda et al., 2012). In the pre-treatment years (2007 and 2008), soil NO_x^--N in the WBG watershed was less than or equal to that in the control watershed. However, from 2010 to the end of the study significantly more soil

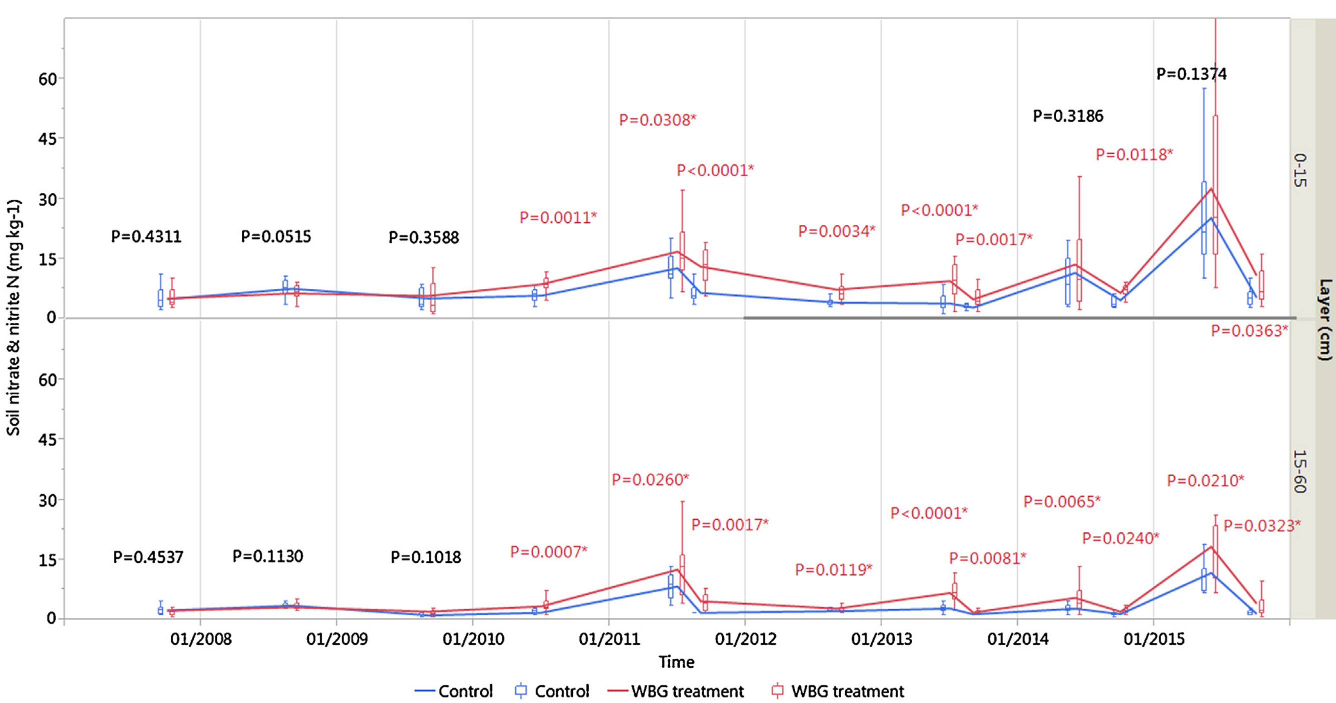

Fig. 2. Soil test N of the winter bale grazing watershed (WBG) and the control watershed. For each sampling date, error bars show the range of data and box plots show 25th, 50th, and 75th percentiles of the data. For the top layer soil test N from 2012 to 2016, only 6–15 cm depth soil rather 0–15 cm depth soil was sampled and measured.

NO_x^--N was observed at 15–60 cm depth in the WBG watershed. At the 0–15 cm depth, a similar increase was observed, with the exceptions of the June 2014 and 2015 sampling dates (Fig. 2). The time lag observed before a significant increase in the treatment watershed may reflect the time required for mineralization of organic N added through bale grazing (Chae and Tabatabai, 1986). Increases in NO_x^--N at the 15–60 cm depth demonstrates that there is downward movement of NO_x^--N beneath the WBG field and potential for nitrate contamination of groundwater.

3.2. Seasonal patterns of runoff from cropland with and without WBG

Inter-year precipitation variability was high during the study (Supplementary Table S3); but average snowfall (110 ± 40 mm) and rainfall (346 ± 147 mm) were close to the long-term averages of 131 mm for snow and 413 mm for rain (Environment Canada, 2015). Snowfall was about 24% of total annual precipitation, but snowmelt runoff comprised between 70% and 100% of total annual runoff and >66% of N and P exports (Supplementary Tables S4 and S5). Over the course of the study 8 years of snowmelt runoff was

Table 3
Nonparametric analysis (Wilcoxon/Kruskal–Walls Test, α=0.05) for nutrient (mg L^{-1}) and sediment (g L^{-1}) concentrations of grabbed samples during snowmelt periods. Data are means and standard deviations. Letters show significance levels, and the same letter indicates non-significant differences among years (the uppercase letter for the winter bale grazing (WBG) treatment watershed, and the lowercase letter for the control watershed). P-values indicate significance of differences between these two watersheds for each year and red text is used to highlight significance.

	WS*	2008	2009	2010	2011	2012¶	2013	2014	2015	2016
Sample number	WBG*	5	13	14	15	2	12	4	8	7
	C*	5	25	10	16	3	13	4	8	8
NH_4^+-N	WBG	0.52 ± 1.72 (BC)	8.01 ± 1.40 (A)	0.17 ± 1.38 (C)	3.88 ± 1.37 (A)	0.24 ± 2.36	4.28 ± 1.42 (A)	0.30 ± 1.83 (BC)	0.64 ± 1.54 (B)	0.60 ± 1.58 (B)
	C	0.42 ± 1.50 (ab)	0.49 ± 1.20 (a)	0.07 ± 1.33 (d)	0.25 ± 1.26 (b)	0.15 ± 1.69	0.10 ± 1.29 (cd)	0.04 ± 1.58 (d)	0.18 ± 1.38 (bc)	0.30 ± 1.38 (ab)
		0.8345	<0.0001*	0.0008*	0.0001*		<0.0001*	0.0304*	0.0406*	0.0092*
NO_x^--N	WBG	18.21 ± 1.82 (A)	0.036 ± 1.45 (D)	8.75 ± 1.43 (A)	0.15 ± 1.41 (C)	19.72 ± 2.57	1.48 ± 1.95 (B)	6.74 ± 1.95 (AB)	18.45 ± 1.60 (A)	18.48 ± 1.66 (A)
	C	16.24 ± 1.52 (a)	0.35 ± 1.21 (d)	9.35 ± 1.35 (ab)	4.42 ± 1.27 (bc)	17.87 ± 1.72	3.96 ± 1.30 (c)	7.66 ± 1.60 (abc)	22.28 ± 1.39 (a)	14.40 ± 1.39 (a)
		0.7526	<0.0001*	0.9068	0.0005*		0.0272*	0.4705	0.7132	0.1480
SuspN	WBG	0.38 ± 1.25 (C)	2.67 ± 1.15 (A)	0.55 ± 1.15 (C)	2.16 ± 1.14 (A)	0.78 ± 1.43	–	2.54 ± 1.29 (A)	1.03 ± 1.20 (B)	0.42 ± 1.21 (C)
	C	0.88 ± 1.20 (b)	0.81 ± 1.09 (b)	0.48 ± 1.14 (c)	0.75 ± 1.11 (b)	0.55 ± 1.27	–	3.37 ± 1.23 (a)	0.38 ± 1.16 (c)	0.50 ± 1.16 (c)
		0.0216*	<0.0001*	0.1514	0.0008*	0.3865	–	0.0421*	0.0039*	0.5244
TN	WBG	19.13 ± 1.25 (ABC)	23.46 ± 1.15 (AB)	10.00 ± 1.15 (D)	17.05 ± 1.14 (BC)	20.83 ± 1.43	–	11.21 ± 1.29 (CD)	29.21 ± 1.20 (A)	22.87 ± 1.21 (AB)
	C	17.74 ± 1.18 (b)	3.63 ± 1.08 (d)	9.75 ± 1.12 (c)	10.28 ± 1.10 (c)	19.17 ± 1.24	–	12.49 ± 1.20 (bc)	29.34 ± 1.14 (a)	18.34 ± 1.14 (b)
		0.8345	<0.0001*	0.9300	0.0034*			0.4705	0.8748	0.3854
SuspP	WBG	0.054 ± 1.47 (BC)	0.44 ± 1.27 (A)	0.09 ± 1.26 (B)	0.41 ± 1.25 (A)	0.13 ± 1.84	0.39 ± 1.28 (A)	0.27 ± 1.54 (A)	0.38 ± 1.38 (A)	0.035 ± 1.38 (C)
	C	0.14 ± 1.32 (ab)	0.14 ± 1.13 (a)	0.081 ± 1.22 (bc)	0.13 ± 1.17 (ab)	0.09 ± 1.43	0.06 ± 1.19 (c)	0.22 ± 1.36 (a)	0.05 ± 1.24 (c)	0.05 ± 1.32 (c)
		0.0601	<0.0001*	0.1598	0.0008*		<0.0001*	0.1939	0.0015*	0.9353
TP	WBG	0.84 ± 1.16 (D)	1.59 ± 1.09 (AB)	0.88 ± 1.09 (D)	1.56 ± 1.09 (AB)	1.29 ± 1.26	1.63 ± 1.10 (AB)	1.07 ± 1.18 (CD)	2.01 ± 1.12 (A)	1.36 ± 1.13 (BC)
	C	1.30 ± 1.15 (a)	0.82 ± 1.06 (bc)	0.80 ± 1.10 (bc)	0.91 ± 1.08 (b)	0.79 ± 1.20	0.84 ± 1.09 (bc)	0.71 ± 1.17 (bc)	0.65 ± 1.12 (c)	0.70 ± 1.12 (bc)
		0.0122*	<0.0001*	0.0177*	<0.0001*		0.0006*	0.0304*	0.0009*	0.0092*
TDP	WBG	0.79 ± 1.17 (CD)	1.03 ± 1.10 (BC)	0.77 ± 1.10 (D)	1.06 ± 1.09 (BC)	1.16 ± 1.27	1.14 ± 1.10 (B)	0.93 ± 1.19 (BCD)	1.64 ± 1.13 (A)	1.28 ± 1.14 (AB)
	C	1.07 ± 1.16 (a)	0.65 ± 1.07 (b)	0.72 ± 1.11 (b)	0.76 ± 1.09 (ab)	0.70 ± 1.22	0.78 ± 1.10 (ab)	0.59 ± 1.18 (b)	0.59 ± 1.13 (b)	0.67 ± 1.13 (b)
		0.0122*	0.0033*	0.0177*	0.0008*		0.0123*	0.0304*	0.0009*	0.0128*
TSS	WBG	0.012 ± 1.25 (BC)	0.035 ± 1.15 (A)	0.019 ± 1.14 (B)	0.035 ± 1.14 (A)	0.015 ± 1.42	0.043 ± 1.15 (A)	–	0.018 ± 1.21 (B)	0.011 ± 1.21 (C)
	C	0.058 ± 1.33 (a)	0.012 ± 1.13 (d)	0.025 ± 1.22 (bc)	0.052 ± 1.17 (a)	0.014 ± 1.44	0.017 ± 1.19 (bcd)	–	0.014 ± 1.25 (cd)	0.026 ± 1.25 (b)
		0.0216*	<0.0001*	0.2073	0.2856		0.0010*		0.1642	0.0371*

WS*: Watershed; WBG*: winter bale grazing treatment watershed; C*: control watershed.
*Significant.
– Missing data.
¶No Wilcoxon test for the data in 2012 due to low sample size.

monitored at each site and in 5 of these years rainfall runoff also occurred. Annual and spring water yield for the WBG watershed was slightly higher than for the control watershed in 4 of 6 years, but for 2 years (2015 and 2016) with low snow accumulation this trend was reversed (Supplementary Fig. S4). The tendency for higher water yield in the WBG watershed occurred in both grazing and non-grazing years, so the difference is likely a function of minor differences in watershed characteristics. Greater variability was observed in the relationship between water yields in the WBG and control watershed for matched runoff events (Supplementary Fig. S4c). Variability was greatest for matched rainfall runoff events. Rainfall runoff events were infrequent and did not occur in

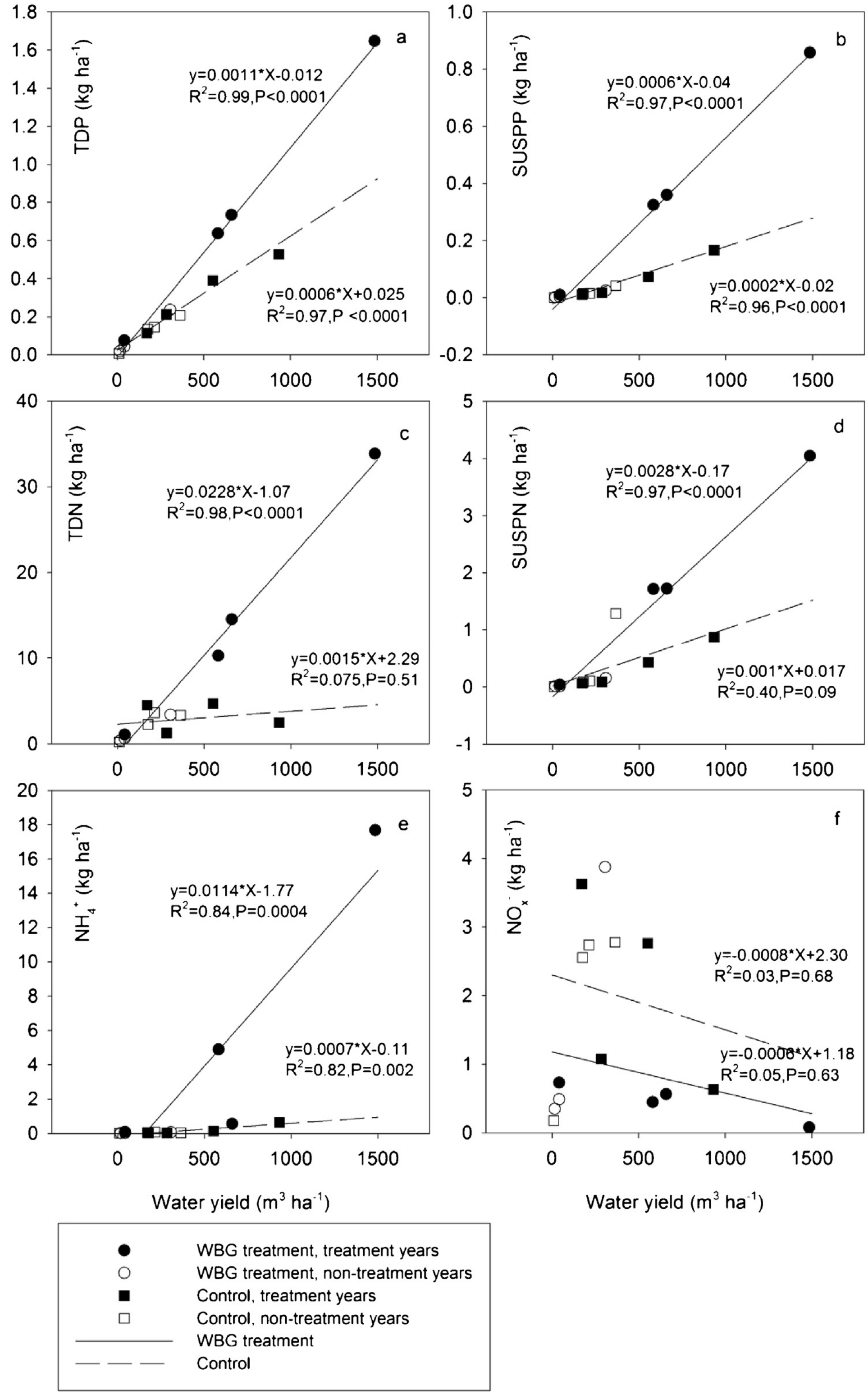

Fig. 3. Relationships between spring exports (kg ha^{-1}) and water yield (m^3 ha^{-1}) for winter bale grazing on cropland (WBG) treatment watershed and for the control watershed.

both watersheds in 2009, 2012 and 2014. Smaller rainfall runoff events in 2010 and 2011 did not always occur in both watersheds simultaneously.

3.3. Concentrations and exports of phosphorus following winter bale grazing

For both control and WBG treatment watersheds, average concentrations (Table 3) and flow weighted mean concentrations (Supplementary Fig. S6a) of TP with snowmelt were consistently well above regional guidelines suggested by Chambers et al. (2012) (0.10 mg L^{-1} TP) and Glozier et al. (2006) (0.26 mg L^{-1}) for prevention of downstream eutrophication for the prairie region.

Concentrations of TDP and TP were lower in the WBG watershed than in the control watershed in the pre-treatment year (2008), but following the first year of treatment TDP and TP concentrations were significantly higher and remained this way over the course of the study (Table 3). In all four bale grazing years, snowmelt and annual FWMCs of TDP, SuspP and TP in runoff from the WBG watershed were at least 1.5 times those observed in the control watershed (Supplementary Table S4).

TDP, SuspP and TP exports increased significantly with flow in snowmelt runoff in both WBG treatment and control watersheds (all $P < 0.0001$). The slope of the relationship between flow and export for the WBG treatment watershed was significantly greater than for the control watershed (ANCOVA, full model, slopes $P < 0.0001$), suggesting a disproportionate increase in dissolved and particle P exports from the treatment watershed in years with more snowfall (Fig. 3a and b). This trend appears to be driven largely by the elevated concentrations of all P forms observed for the WBG treatment watershed in years both with and without bale grazing (Fig. 4a and b).

Phosphorus was exported from both WBG and control watersheds primarily in a dissolved form (66%–89% annual average TDP/TP ratio); however, following bale grazing SuspP export increased significantly, and in all four WBG treatment years SuspP concentrations were significantly greater in the bale grazing watershed (Figs. 3 b and 5 b). Snowmelt FWMCs of TDP remained consistently elevated in the WBG watershed following an increase after the first bale grazing treatment, but SuspP was mainly elevated in treatment years (Fig. 5b). Despite a larger proportion of TP being in dissolved than in particulate form, in grazing treatment years the concentrations of TP in the WBG watershed were further elevated above the control due to an increase associated with SuspP from bale grazing (Fig. 5c).

The potential for higher P loss from the WBG watershed than from the control watershed persisted in the non-grazing years following each bale grazing. A pattern of increasing extractable P at the soil surface, but not at deeper depths was observed (Figs. 1 and 6) and indicates that the distribution of P in the soil profile was altered following WBG. This apparent stratification of soil P was observed despite frequent tillage in the WBG watershed to more thoroughly incorporate into the soil the material deposited with the WBG treatment.

Dung, urine, bedding, and uneaten feed are potential sources of dissolved P in runoff following bale grazing and these sources also appear to contribute to an increase and stratification of Olsen-P at the 0–6 cm depth that persists following bale grazing. Through monitoring of runoff from grazed pastures in Wisconsin and application of the Annual P Loss Estimator (APLE) model, Vadas et al. (2015) identify transport related factors (influencing annual runoff volume) as the primary control of annual P export with runoff from grazing sites. However, total dung excreted was also identified as a critical factor. The linear increases in TP export observed with water yield for the WBG watershed in the current study (Fig. 3) and elevation of TP export in years of greater deposition of dung, urine, and uneaten feed indicate that similar processes likely control potential for runoff export of P from WBG as at other grazed sites. Therefore the application of predictive models like APLE could improve planning and WBG designs.

More complete incorporation of WBG residues by tillage might reduce stratification and loss of accumulated soil P in runoff in non-bale grazing years; however, more intensive tillage could reduce the soil fertility benefits of WBG by burying P deeper than the rooting zone for seedlings of some crops or by increasing the potential for loss of SuspP through erosion of disturbed soils. For watersheds on the Canadian Prairies where repeated manure application have significantly elevated soil-P, strong correlations have been observed with FWM concentrations of TP and TDP in runoff (Little et al., 2007). Despite Olsen-P measurements falling within existing provincial fertility guidelines at the WBG study site in Manitoba (Manitoba Livestock Manure and Mortalities Regulation; 60 mg kg^{-1}), relatively high concentrations of P were observed in runoff from the WBG watershed in both treatment and non-treatment years. For this reason, any soil testing recommendations developed to prevent excessive runoff losses of P may need to be set lower than existing guidelines.

3.4. Concentrations and exports of nitrogen in runoff following winter bale grazing

For TN, NO_x^--N and NH_4^+-N, the concentrations measured in runoff from both WBG and control watersheds exceeded guidelines for aquatic ecosystems on the Canadian Prairies (>1.16, 0.28 and 0.12 mg L^{-1}, respectively; Glozier et al., 2006; CCME, 2002). In the pre-treatment year (2008), both the WBG and the control watershed had similar mean concentrations of NH_4^+-N, NO_x^--N and TN (Table 3). A dilution effects with increasing snowmelt was observed for NO_x^--N for both the WBG and control watersheds with a negative relationship existing between snowmelt FWMCs and water yield (Fig. 4f). Dilution was also observed for TDN and TN in the control watershed, but not in the treatment watershed due to high variability with the inclusion of both treatment and non-treatment years (Figs. 3 c and S6c).

Concentrations of NH_4^+-N increased significantly after the first bale grazing in the treatment watershed (Table 3). Although snowmelt FWMCs of NH_4^+-N significantly increase with water yield in both watersheds, the increase was disproportionately higher in the WBG watershed (Fig. 4e). In years following bale grazing, the WBG watershed had 17.4 (11.9 vs. 0.68 mg L^{-1} in 2009), 32.7 (8.42 vs. 0.26 mg L^{-1} in 2011), 8.8 (0.83 vs. 0.09 mg L^{-1} in 2013), and 6.7 (1.86 vs. 0.26 mg L^{-1} in 2015) times greater snowmelt/annual FWMCs of NH_4^+-N than the control watershed (Supplementary Table S4). In non-treatment years, NH_4^+-N concentration differences between the watersheds were considerably less, with the WBG watershed measuring 2.3, 1.3 and 1.3 times greater than the control in 2010, 2012 and 2016, respectively (Fig. 5g). Given that NH_4^+-N showed a tendency to increase with water yield; large differences were also observed for NH_4^+-N export in treatment years (Fig. 3e, Supplementary Table S5).

High NH_4^+-N concentrations in snowmelt runoff have previously been observed from pasture and cropland with manure application on snow (Cade-Menun et al., 2013; Jokela and Casler, 2011; Owens et al., 2011; Young and Mutchler, 1976) and from winter feeding sites on pasture (Smith et al., 2011), implicating dung or urine from overwintering cattle in the treatment watershed as the primary source of NH_4^+-N in runoff rather than fertilizer inputs, soil, or crop residue. Low temperature may be an important factor in minimizing ammonia volatilization, nitrification losses, and plant uptake (Gupta et al., 2004; Lauer et al., 1976) to maintain NH_4^+-N at the soil surface. For matched rainfall events in 2011, a pattern of decreasing FWMCs of NH_4^+-N was observed with time over the course of four rainfall runoff events occurring between the end of May and the end of June (9.2, 3.1, 0.38–0.13 mg L^{-1}). This could reflect a decrease in the NH_4^+-N source following flushing and/or transformation without rebuilding until the next bale grazing treatment, but increased replication is required to test this hypothesis.

Dung, bedding and uneaten feed are also a source of SuspN. In years with bale grazing snowmelt and annual FWMCs of SuspN in

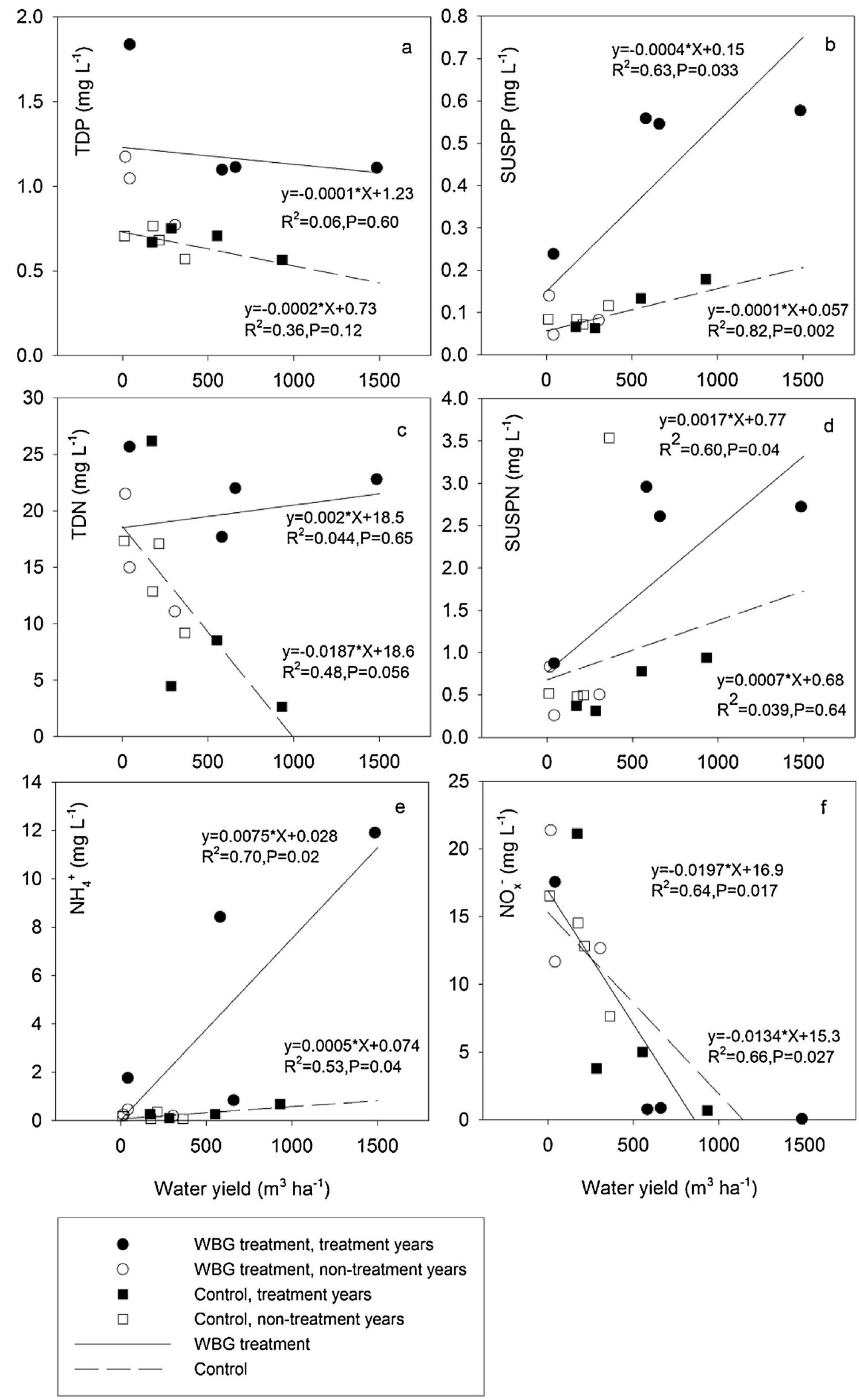

Fig. 4. Relationships between spring FWMCs (mg L^{-1}) and water yield (m^3 ha^{-1}) for winter bale grazing on cropland (WBG) treatment watershed and for the control watershed.

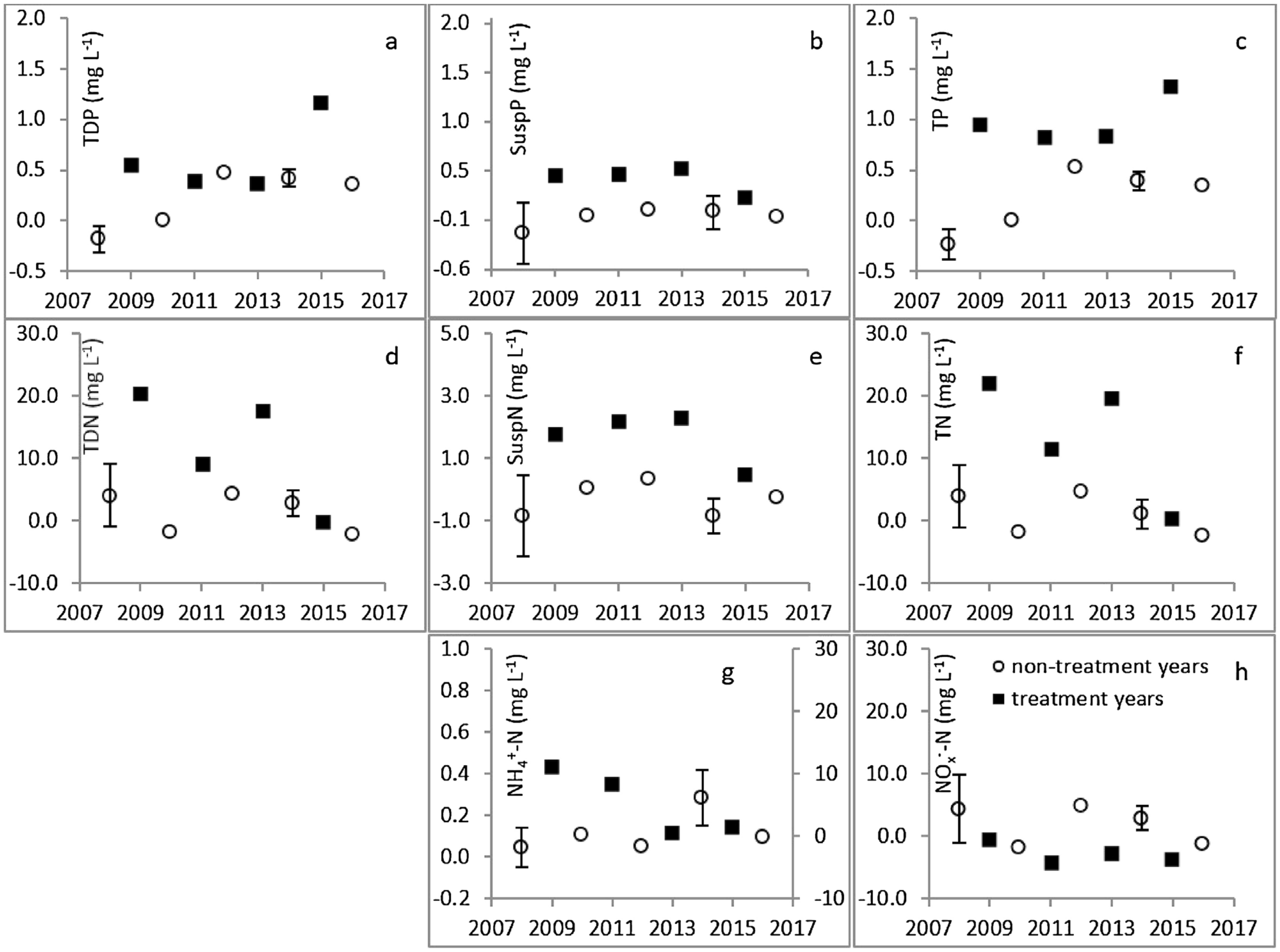

Fig. 5. Concentration differences between the winter bale grazing (WBG) treatment watershed and the control watershed for runoff nutrients. For NH_4^+-N, data in treatment years use second Y-axis on the right. Error bars are shown when arithmetic mean concentrations were used instead of flow weighted mean concentrations.

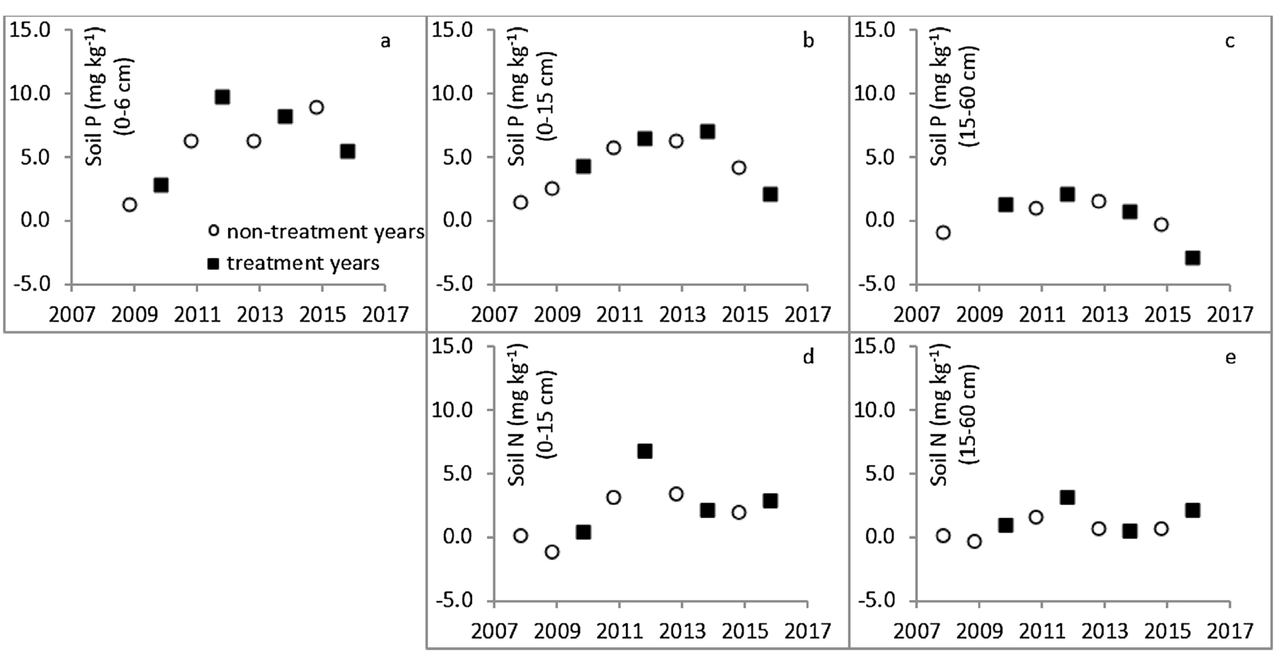

Fig. 6. Concentration differences between the winter bale grazing (WBG) treatment watershed and the control watershed for soil Olsen-extractable phosphorus and soil nitrogen.

runoff from the WBG watershed were significantly higher (Fig. 4d, Supplementary Table S4). Snowmelt and annual FWMCs of NO_x^--N in runoff from the WBG watershed were not significantly higher than the control watershed (Fig. 4f, Supplementary Table S4). Smith et al. (2011) also observed no significant increase in NO_x^--N concentration in snowmelt runoff after bale-grazing on a pasture site. Organic N, urea or NH_4^+-N are deposited with bale grazing and low winter temperatures are likely to slow their transformation to nitrate. In the presence of freshly added organic C, any nitrate that did form may be subject to microbial denitrification or immobilization (Taylor and Townsend, 2010).

Concentrations and snowmelt FWMCs of TN in runoff samples collected following bale grazing were significantly greater than the control watershed (Supplementary Fig. S6c and Table S4), but higher concentrations in the WBG watershed were not observed in non-treatment years (Fig. 5f). The higher TN observed following bale grazing appears to be largely a function of increased dissolved organic nitrogen (DON), NH_4^+-N, and SuspN. By minimizing NH_4^+-N volatilization in comparison to alternatives that require confined feeding, manure stockpiling, and manure spreading (Jungnitsch et al., 2011), in-field feeding systems likely increase the mass of N available for the following crop; however, this also increases the mass of N available for transport with runoff. Given the high bioavailability of NH_4^+-N and DON and their potential to impact downstream oxygen demand (Hooda et al., 2000; Lehman et al.,

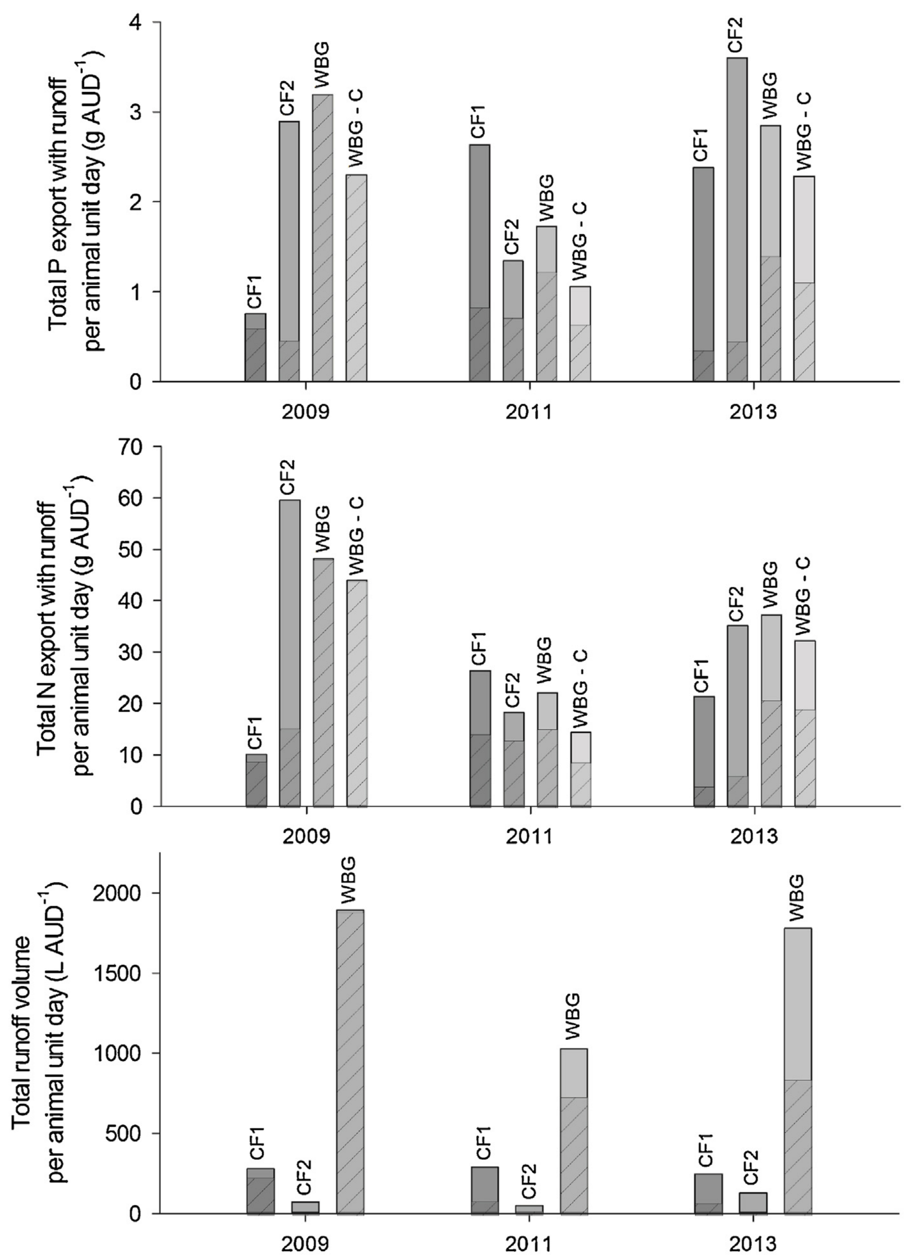

Fig. 7. Total P export, total N export, and total runoff volume per animal unit day (AUD) are shown for two confined feeding systems (CF1, CF2) and a lower density winter bale grazing on cropland site (WBG) located in the larger South Tobacco Creek watershed in Manitoba. For Total P export and Total N export for WBG values are also shown with export from a paired control site subtracted to provide a more conservative estimate of increases associated with WBG (WBG-C). Subtraction of background was completed using per unit area values and rescaled to the overall total based on watershed area. Bars heights in each plot show annual totals, while hashed sections of each bar indicate snowmelt totals.

2004), caution should be exercised when choosing bale grazing locations, so that direct runoff to aquatic ecosystems is avoided.

3.5. N and P export from winter bale grazing as compared to confined winter feeding sites

Rates of annual N and P export from CF sites were significantly higher per unit area than for the WBG watershed as a result of much larger watershed area at the WBG site and high concentrations in runoff from the higher density CF sites. However, when annual export rates are expressed per AUD of feeding the measured rates of export from all winter feeding systems are comparable (Fig. 7). Timing of N and P export with runoff differed significantly between feeding systems, with a much higher proportion of export from the CF sites occurring with rainfall runoff events (Fig. 7). Also, runoff from WBF was consistently of higher volume, but lower concentration in comparison to CF sites. High concentrations of N and P in runoff from CF sites are not surprising, given the differences in animal density and higher inputs per unit area of feed. Higher water yield per unit area is likely associated with compaction, presence of more impervious surfaces within CF sites, and lack of vegetation to use summer soil moisture.

Identification of environmental and management factors controlling runoff losses of N and P from CF sites is beyond the scope of the current manuscript which focusses on WBG. However, useful insight into what challenges must be considered in developing overwintering systems to reduce nutrient runoff can be gained from this initial comparison of neighbouring WBG and CF sites. Despite reductions in runoff concentration achieved with the lower density WBG system, larger volumes of runoff are produced per animal when the area provided for overwintering is increased. As a result, exports of N and P per AUD remains quite similar for the CF and WBG sites studied and any mitigation efforts involving capture and treatment of runoff will require a much larger storage volume for WBG. Any reduction in snowmelt runoff that can be achieved through management changes to WBG such as selection of sites with minimal snow accumulation and smaller upstream contributing area are likely to reduce annual export of N and P most significantly.

4. Conclusions

As expected, the WBG system improved soil fertility in our study watershed; however, the use of this management practice also increased N and P export with runoff. Future implementation of the practice must balance the economic and soil fertility benefits of in-field bale grazing with observed environmental impacts. This requires that the system be implemented with caution, avoiding sites above aquifers that are vulnerable to contamination, those sites where runoff has the greatest potential to reach surface waterbodies, and those sites through which larger volumes of runoff are likely to be transported. Given that accumulation of P was observed in the top 0–6 cm of the soil in the treatment watershed, one management option that may deserve further attention to help reduce P export would be a shift to more intensive tillage practices at cropland locations with WBG. This increase in tillage may more completely bury manure and feed residues accumulating near the soil surface. Soil testing to detect accumulation of excess P at WBG sties deserves further investigation and lengthening time between treatments should be considered, given the potential for accumulation of P at the soil surface and for increased export following repeated years of bale grazing.

In comparison to winter CF sites, export of N and P with runoff from WBG occurs primarily with snowmelt and is of lower concentration, but higher volume per overwintering animal. Preliminary comparison between CF and WBG systems indicates that the overall impact on runoff export per AUD is similar, but a unique set of challenges must be addressed if N and P runoff losses from WBG sites are to be reduced. Selection of WBG sites with lower snow accumulation and smaller upstream contributing area to minimize the volume of snowmelt from WBG may be an effective approach to reduce N and P losses. Any reductions in volume that are achieved will also make capture of runoff a more feasible option.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2017.02.003.

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