**Soil and water management practices: Opportunities to mitigate nutrient losses to surface waters in the Northern Great Plains**

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Soil and water management: Opportunities to mitigate nutrient losses to surface waters in the Northern Great Plains

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Abstract

The Northern Great Plains is a key region to global food production. It is also a region of water stress that includes poor water quality associated with high concentrations of nutrients. Agricultural nitrogen and phosphorus loads to surface waters need to be reduced, yet the unique characteristics of this environment create challenges. The biophysical reality of the Northern Great Plains is one where snowmelt is the major period of nutrient transport, and where nutrients are exported predominantly in dissolved form. This limits the efficacy of many BMPs commonly used in other regions and necessitates place-based solutions. We discuss soil and water management beneficial management practices (BMPs) through a regional lens – first understanding key aspects of hydrology and hydrochemistry affecting BMP efficacy, then discussing the merits of different BMPs for nutrient control. We recommend continued efforts to ‘keep water on the land’ via wetlands, and reservoirs. Adoption and expansion of reduced tillage and perennial forage may have contributed to current nutrient problems, but both practices have other environmental and agronomic benefits. The expansion of tile and surface drainage in the Northern Great Plains raises urgent questions about effects on nutrient export, and options to mitigate drainage effects. Riparian vegetation is unlikely to significantly aid in nutrient retention, but when viewed against an alternative of extending cultivation and fertilization to the waters’ edge, the continued support of buffer strip management and refinement of best practices (e.g., harvesting vegetation) is merited. While the hydrology of the Northern Great Plains creates many
challenges for mitigating nutrient losses, it also creates unique opportunities. For example, relocating winter bale-grazing to areas with low hydrologic connectivity should reduce loadings.

Managing nutrient applications must be at the center of efforts to mitigate eutrophication. In this region, ensuring nutrients are not applied during hydrologically sensitive periods, such as late autumn, on snow, or when soils are frozen, will yield benefits. Working to ensure nutrient inputs are balanced with crop demands is crucial in all landscapes. Ultimately, a targeted approach to BMP implementation is required, and this must consider the agronomic and economic context, but also the biophysical reality.

Keywords: beneficial management practice, agriculture, nutrient, Northern Great Plains, prairie, water quality
1 Introduction

The Northern Great Plains is an area of fertile soils and high agricultural productivity, considered part of the ‘bread basket’ of North America (Fig. 1). It includes agricultural regions of Canada’s three Prairie Provinces: Alberta, Saskatchewan, Manitoba, as well as North Dakota and parts of Montana, South Dakota, Minnesota and Iowa in the USA (Fig. 1). The region is semi-arid to sub-humid, flat and cold. On average, water deficits, where evapotranspiration exceeds precipitation, predominate on the Northern Great Plains and agricultural productivity is limited by moisture. Many streams in this region are temporary, flowing only in spring and following storms. The temporary nature of streams highlights the periodic and episodic nature of runoff in this region, and specific challenges to nutrient and flood control. Although sporadic, runoff typically has high concentrations of nutrients, in part due to limited dilution in a nutrient-rich landscape where agricultural sources have been associated with increased nutrient loads (e.g., Palliser Environmental Services Ltd and Alberta Agriculture and Rural Development 2008; Ryberg 2017; Schneider et al. 2018). Vast areas of the region are considered ‘non-contributing’ to larger watersheds under typical moisture conditions, with closed, non-dendritic drainage. The contributing area of a watershed is affected by climate, but also by land use, most notably agricultural drainage.

Throughout the region, lakes and rivers are affected by high nutrient concentrations, contributing to harmful cyanobacterial blooms, and loss of ecosystem services (Beaver et al. 2014; Kehoe et al. 2015; Maheaux et al. 2015). This is not a new issue, with evidence of poor water quality, even prior to agricultural development (Palliser and Spry 1968; Hall et al. 1999), due to naturally high nutrient concentrations in soils of this landscape (Barica 1987).

Nonetheless, there is recognition that human activity has contributed to degraded water quality (Ryberg 2017).
Nutrient export coefficients from agricultural lands in the region are typically higher than from areas with native vegetation (Timmons and Holt 1977; Jeje 2006; Donald et al. 2015). Moreover, field-scale export of P is related to soil test P values (Alberta Agriculture and Forestry 2014). There is increasing pressure to achieve nutrient load reductions from all anthropogenic sources, including agriculture (Schindler et al. 2012). The great challenge in terms of agricultural nutrient management is the sensitivity of downstream ecosystems. Aquatic ecosystems respond to nutrient additions in the $\mu$g L$^{-1}$ ($\mu$g kg$^{-1}$ water) range, while agricultural producers manage nutrient inputs in the kg ha$^{-1}$ (mg kg$^{-1}$ soil) range. Nutrient losses from farmland are often agronomically and economically insignificant, but can have large impacts on downstream ecosystems due to the sensitivity of lakes to nutrient addition. This is further compounded by the large land area to surface water ratio that characterizes lakes of the region. An example of this is the Assiniboine-Souris-Qu’Appelle watershed (in southern Saskatchewan, Manitoba, Canada and northern North Dakota, USA). This large watershed (~15 million ha) yields a relatively small amount of runoff water. Its P load of approximately 600 t of P per year from non-point sources (estimated from data in Jones and Armstrong 2001) translates into an average loss of only 0.04 kg ha$^{-1}$ P. However, median concentrations of P near the outlet for the Assiniboine River are in the range of 200 $\mu$g L$^{-1}$, which is in excess of recommended thresholds for aquatic ecosystems (Chambers et al. 2012).

While prairie lakes and rivers with their naturally high nutrient concentrations may have been considered relatively insensitive to change, evidence of worsening cyanobacterial blooms with increased nutrient loads suggests this assumption is incorrect (Schindler et al. 2012). Still, it is difficult to predict how nutrient load reductions will affect these naturally mesotrophic-eutrophic ecosystems. Overlaid on uncertainty regarding lake, stream and wetland responses is the issue of climate change, which is expected to increase the sensitivity of surface waters to
nutrient inputs (e.g., due to temperature increases; Paerl and Huisman 2008), and will also affect
nutrient loading with changing patterns of precipitation amount and form. It is likely that in
many regions, climate change may worsen the already serious effects of nutrient pollution from
human activity (e.g., Paerl and Huisman 2008; Crossman et al. 2013; Sinha et al. 2017).

Nutrient losses from agricultural lands to surface waters require the presence of both a
nutrient source (i.e., soils or plant residues) and a transport pathway. The likelihood of transport
increases when the source is in an easily mobilized form and has contact with water that is
flowing on the land surface. This runoff water depends on the volume of incident precipitation,
the permeability of the soil that partitions the incident water between infiltration and runoff, and
landscape features that distribute flowing water across the land surface. Dissolved nutrient forms
are easily mobilized with any runoff water but to release particulate nutrients the energy of the
runoff water must be sufficient to detach and entrain surface soil particles (water erosion).
Nutrient loading and concentrations in surface runoff depend on the relative magnitude and
interaction of sources and transport. In general, runoff volume is more important for nutrient
loads while nutrient source availability has a greater effect on nutrient concentrations. Within
agricultural landscapes, nutrient management can focus at the source (controlling inputs), and via
landscape interventions intended to control the transport of nutrients that are already in the
landscape. However, despite decades of work implementing BMPs, we still lack adequate
understanding of their efficacy in nutrient control among different landscapes and climates.
Uniqueness of place (sensu Beven 2000) limits transferability of research on BMPs among
landscapes with varied soils, hydrology, slope and other factors. High spatial and temporal
variation affect application of paired designs and before-after study designs for BMP assessment.
These challenges are particularly acute in regions such as the Northern Great Plains, which are
characterized by periodic runoff and high inter-annual variability. In many cases, exploration of
potential BMP effects has relied on model-based approaches, despite substantive challenges in modelling (structure, parameterization, and resulting uncertainty), and our often limited fundamental understanding of how, where, and why successful BMPs in nutrient management are transferable. A further challenge is that all BMPs have varied benefits and detriments – that span not just nutrients, but affect habitat, soils, hydrology and agricultural productivity (Gitau et al. 2005; Council of Canadian Academies 2013).

In the region, high spatio-temporal variation in flows, with notable importance of snowmelt nutrient export, creates challenges to nutrient control. Here, we start with a review of the physical and agronomic conditions, which make the Northern Great Plains unique, and, may limit the transferability of BMPs that have demonstrated success in nutrient control in other landscapes. We review the state of our knowledge among multiple management practices applied for agronomic and environmental reasons. We assess: (a) how these BMPs are likely to affect nutrient transport in the unique landscapes and cold climate that characterize the Northern Great Plains, (b) summarize studies that have quantified their efficacy, and identify potential trade-offs associated with their implementation. We discuss BMPs broadly, but note that management practices may be the more appropriate term, because net benefits in terms of one environmental goal are often matched by net detriments in terms of agricultural production, or other environmental goals. Our focus is on six categories of soil and water management BMPs: (1) tillage (or reduced tillage); (2) perennial forages, grasses and grazing; (3) vegetated riparian buffers, channels, and filter strips; (4) surface drainage; (5) subsurface drainage and (6) reservoirs, wetlands and holding ponds. Although agricultural nutrient management planning must be the basis of work to reduce nutrient loads, most of the challenges of managing nutrient application rates, sources, placement and timing are not particularly distinctive for this geographic region when compared to other regions with similar land use. Therefore, we briefly
summarize key aspects of nutrient management in this region and focus mainly on the effect of soil and water management BMPs on nutrient transport in the Northern Great Plains. Our focus is on synthesizing research where nutrient loss outcomes can be related directly back to management practices, typically at an edge-of-field or small watershed scale. Reductions in nutrient load at these scales may not reflect the loads observed at larger scales but they do represent reduced agricultural loadings into surface water systems. Nutrients may be transformed, recycled, stored, and lost affecting their downstream delivery. Their transport, both spatially and temporally, is beyond the scope of this review.

1.1 The Environment of the Northern Great Plains

There are three physiographic features of the Northern Great Plains that affect nutrient transport from agroecosystems. First, is the semi-arid to sub-humid climate, which typically has very low rainfall runoff. Second, cold winters and low rainfall runoff combine to make snowmelt runoff over frozen soils the dominant driver of nutrient transport. Typically snowmelt constitutes more than 75% of annual runoff (e.g., Nicholaichuk 1967; Gray and Landine 1988; Glozier et al. 2006; Little et al. 2007; Dumanski et al. 2015). Finally, most of the region is relatively flat or gently sloped, with low risk of soil erosion by water (Fig. 2).

Hydrology is a key determinant of nutrient exports, with spring flows dictated by processes which differ significantly from summer periods. Specifically, the development of a basal ice layer can significantly impact flowpaths during snowmelt, leading to periodic high runoff and low infiltration. Even in the absence of a basal ice layer, infiltration can be limited by water and ice content of soils at snowmelt, contributing to higher spring runoff, particularly if soil moisture was high in the previous fall. While a basal ice layer can impact soil-water interactions, the patchy nature of thaws, and slow runoff means that high nutrient concentrations can still be
observed in runoff, particularly as soil freezes and basal ice thaws (Quinton and Pomeroy 2006; Costa et al. 2017). Alternatively, where significant macropores exist, or in coarse soils, extensive infiltration can occur, retaining most of the snowmelt (Gray et al. 2001; Hayashi 2013). Solutes can move rapidly through soil without interacting with the matrix due to biopore formation (e.g. root channels and rodent burrows) and cracking of soils upon desiccation (Schuh et al. 1997). This type of preferential flow has been reported across the region (Perillo et al. 1999), with greater importance in areas of high clay content in soils (Dadfar et al. 2010).

The persistence of snow through the winter means it can be redistributed by wind across the landscape, with vegetation cover impacting where snow is trapped. For example, tall vegetation can hold snow in place. This leads to impacts on flow and flowpaths during snowmelt (Shook et al. 2015; Mahmood et al. 2017). Often, snowmelt in the Northern Great Plains is a slow process, occurring on the timescale of weeks. However, rapid events can occur, most notably when rain occurs before snowmelt is complete. Strong warm winds during the winter (referred to as Chinooks) also cause very rapid snowmelt in Montana and southern Alberta. These events can have important implications for peak flows, flood risk and nutrient transport. A large proportion of the landscape is non-contributing in any given year; however, the proportion that contributes to flow can be highly dynamic, driven by wet-dry climatic cycles characteristic of this region and associated changes in depressional water storage. Wet-dry cycles are a key factor in the regional climate, and are associated with alternating flood risk and drought risk (Mahmood et al. 2017), highlighting the need to understand the robustness of management practices to highly varied hydrological and climatic conditions.

Snowmelt processes dictate not only the volume of runoff, but can also affect the chemistry of runoff waters. Soil-water interactions during snowmelt may vary significantly from year to year, and in space. Basal ice layers alter meltwater runoff chemistry (Lilbaek and Pomeroy 2010).
and affect the degree to which soils and runoff water are in contact. The soil-water interaction zone that is thought to consist of the top 2 cm of soil under rainfall conditions (Ahuja and Lehman 1983) is likely less during snowmelt runoff over frozen soil. Contact time between soil and water can also be important in nutrient transport (Amarawansha et al. 2015), and will vary depending on characteristics of the snowmelt runoff event; however more biogeochemical research at low-temperature is required to understand how the rapidity of melt affects runoff chemistry. Potential snowmelt changes in runoff pH, and solutes such as calcium (Ca) and magnesium (Mg) may impact P retention in sediments (Nair et al. 1984; Klotz 1991). Direct nutrient inputs into the snowpack can also impact runoff chemistry. Finally, freeze-thaw cycles can impact microbial processes, water chemistry, and cycling and export of nutrients (Groffman et al. 2001; Bechmann et al. 2005; Matzner and Borken 2008; Messiga et al. 2010). Within the Northern Great Plains, the majority of nutrient transport is in dissolved forms (Table 1). This renders BMPs designed for particulate retention less effective in this landscape, and creates challenges in nutrient control.

Dryland cultivation of cereals and oilseeds covers most of the area (Fig. 3), with small but growing areas under irrigation. Irrigated areas for Northern Great Plains states and provinces ranged from less than 1% in North Dakota, Iowa, Manitoba and Saskatchewan to 3.5% of Alberta, and 11% in Montana (data from 2010 Canada and 2012 U.S.). Historically, cereals dominated in the drier areas such as Montana, Alberta and Saskatchewan while corn-soybean rotations were common in wetter and warmer regions like Minnesota and Iowa. Crop rotations have been relatively static in the past 30 years in Montana (70-75% cereals – mostly wheat, 20% forage) and in Iowa (~55% corn, ~35% soybeans), but elsewhere there has been a trend away from cereal production (Fig. 3). On the Canadian prairies, this trend has resulted in more diversified production with increasing area in oilseeds, pulses and soybeans and forages.
Production of corn, pulses and soybeans has also increased in the Dakotas and Minnesota. These changes towards more intensive agricultural production necessitate more intensive nutrient, soil and water management to prevent water contamination by nutrients.

Despite recent intensification, fertilizer use in agriculture in the Northern Great Plains appears to be generally in balance with crop production. Data from the International Plant Nutrition Institute (IPNI) suggest the N and P balances for the region are near neutral (IPNI 2012, 2014). However, there may be regional surpluses of P, where intensive livestock operations are concentrated (e.g., in SE Manitoba and SW Alberta), that represent potential priority areas for abatement. There are likely corresponding regional deficits for P in other regions. Soil test data indicate that on average 65% of soil samples tested less than 25 ppm Bray equivalent P in 2015 compared to 80% of samples in 2001. This represents a notable shift towards increased soil P, which may reflect at least in part, increased testing of manured fields. Regionally in 2015, more than 80% of samples from North Dakota and Saskatchewan were less than 25 ppm while the proportions for Alberta, Manitoba and Minnesota were closer to 60%, again indicative of important spatial variation. At a finer scale, a 2012 survey across a range of cropping systems in this region showed mean Olsen test P concentrations varied nearly 5-fold between fields (4.5-22 ppm; Wilson et al. 2016). This high degree of heterogeneity in nutrient pools between fields within the region, combined with evidence of a high degree of P saturation (e.g., fields where the Bray equivalent soil test P is greater than 50 ppm), highlights the potential of spatially targeted management to reduce nutrient runoff losses.

2 Tillage Practices
Historically, tillage has been used to prepare a seedbed, loosen compacted soil, incorporate residues and kill weeds (Davies and Payne 1988). Tillage practices cover a wide spectrum of soil disturbance from zero tillage through conservation tillage and rotational tillage to conventional tillage and subsoiling (Table 2). As its name suggests, some form of conservation tillage is generally recommended due to improvements in the protection of soil from wind and water erosion, conservation of soil moisture, and wildlife habitat.

Summerfallow, where no crop is grown in an attempt to conserve soil moisture in drier regions, can use tillage alone (conventional fallow), herbicides alone (chemical fallow) or a combination of both. Tilled fallow could have greater tillage intensity than cropped conventional tillage depending on the implements used and number of passes. A tillage-herbicide combination would involve intermediate tillage intensity and chemical fallow would have tillage intensity equivalent to zero tillage (Table 2). Organic production accounts for a small proportion of operations (e.g., less than 3% of operations within each of the Canadian Prairie Provinces in 2016; Statistics Canada, 2017) and relies on tillage and crop selection for weed control. As a result, organic fields are usually tilled more frequently than those managed using pesticides.

The effect of tillage on nutrient loss to surface water depends on the intensity of the practice and the environmental conditions (climate, topography, soil type) in which it is applied. Tillage practices vary throughout the Northern Great Plains due to differences in climate, tradition and policy, and are also changing with time. For example, in the Canadian portion of the Northern Great Plains, the area where direct seeding is practiced is increasing at the expense of conventional tillage (Clearwater et al. 2016). Conventional tillage decreased from 67% of the cropland area in 1991 to 16% in 2011 while direct seeding increased from 7 to 61% during the same period. The area under conservation tillage remained relatively constant throughout the
time period, between 25 and 35%. Summerfallow in the region has declined from 18% of total farmland area in 1981 to 4% in 2011.

2.1 Effects of tillage on hydrology and water quality

Tillage can affect water quality through six mechanisms: flattening of standing stubble, creating cracks in compacted soils, disrupting continuous pores, disrupting aggregates, mixing of soil and crop residues, and redistributing nutrient-rich soil in the landscape (Table 3). While some of these mechanisms affect water quality directly, most have an indirect effect through hydrology and the effect on water quality results from links between hydrology and water quality. While it is well-accepted that downstream nutrient loads during snowmelt- and rainfall-generated runoff events are largely controlled by the volume of runoff (Salvano et al. 2009; Liu et al. 2013; Rattan et al. 2017), nutrient concentrations may not be related to flow during snowmelt (Glozier et al. 2006; Han et al. 2010; Ali et al. 2017).

2.1.1 Flattening of standing stubble:

The flattening of standing stubble by tillage has a significant effect on water quality only when it occurs in the fall. Fall tillage is not normally part of a conservation tillage system, but it does occur when rotational tillage is practiced (Liu et al. 2014a). Conventional tillage in wetter regions and on many organic operations usually involves a fall tillage pass. In drier regions when conventional fallow is utilized, stubble remains standing during the first winter but is flattened in the spring prior to the second fall season.

Tall standing stubble can increase snow retention on a field by reducing the amount of snow that is blown off the field and subject to sublimation losses resulting in more moisture in the snowpack that accumulates on the field during winter (Pomeroy and Gray 1995). A single tillage pass is sufficient to negate the snow-trapping effect of standing stubble. In dry winters when
snow depth is less than the height of standing stubble, snow retention by the stubble has a significant effect on the snow water equivalent in the snowpack (Willis et al. 1969) but if incident snowfall accumulates to a depth in excess of the height of stubble the snow trapping effect is less significant (Tiessen et al. 2010). Because tall stubble results in greater snow water equivalent and more meltwater during snowmelt, the effect of flattening the stubble with tillage can potentially decrease runoff volumes and downstream nutrient loads, although this will depend on the landscape context and depressional storage. The flattening of stubble can have complex effects on hydrology because snow is redistributed by wind increasing sublimation losses and accumulation in depressions (Fang and Pomeroy 2008). Snow that accumulates in pothole wetlands and melts in situ may result in increased wetland water levels and groundwater recharge (given there is not sufficient opportunity for the meltwater to infiltrate in the upland landscape; Gray et al. 2001). However, this meltwater will not contribute to downstream flow unless the storage capacity of wetlands is exceeded (Shaw et al. 2012). Conversely, if snow accumulation occurs in stream channels or roadside ditches, downstream runoff could increase because the meltwater is delivered more directly into the stream (Pomeroy et al. 1998). Although total runoff volume could increase in this case, which is likely to increase nutrient loads, peak flows might not increase. This lack of effect on peak flows can result from a prolonged melt period with the accumulated snow melting more slowly than a shallower snowpack (Pomeroy et al. 2007).

2.1.2 Creating cracks in compacted soils:

In recently tilled fields, soil porosity is generally high in the surface soil and may allow greater infiltration of small rainfall events (Burwell and Larson 1969). However, during larger rainfall events, deeper infiltration and subsoil moisture recharge may be limited by the absence of continuous pores linking the highly porous topsoil with the subsoil. In compacted soils, deep-
ripping or subsoiling below the usual depth of cultivation has been shown to improve infiltration of rainfall-generated and snowmelt runoff to the subsoil (McConkey et al. 1990; Travis et al. 1991). In soils that do not crack naturally, the deep fractures created by subsoil tillage have been shown to improve soil water storage for several years and the cracks persist for up to 5 years in dry conditions (Grevers and Jong 1993). Subsoiling after the soil has frozen may increase the effectiveness of the practice (Pikul et al. 1996).

### 2.1.3 Disrupting continuous pores:

Unlike the deep tillage used in subsoiling of compacted soils, shallow surface tillage in conservation tillage systems can be sufficient to disrupt the development of continuous pore networks (Elliott et al. 2000). In contrast, the lack of soil disturbance in zero tillage and chemical fallow systems can allow the development of continuous pores and cracks that can increase infiltration during summer rainfall events and also potentially during snowmelt (Azooz and Arshad 1996; Elliott and Efetha 1999).

A continuous pore network that transmits water more efficiently through the soil profile can gradually develop in zero tillage and chemical fallow systems through processes including freezing and thawing, wetting and drying, and biological activity. This continuous pore network can ultimately improve infiltration rates, although this change is not immediate. In the interim, after cessation of tillage, but before the continuous pore network develops, infiltration may be reduced as denser soil initially results from elimination of tillage (Elliott et al. 2001).

Where continuous pore networks develop, increased infiltration on zero-till fields will reduce downstream runoff water volumes (Maulé and Reed 1993; Elliott and Efetha 1999). However, the net result of more effective porosity on runoff is more pronounced for rainfall events than during snowmelt runoff. This is because the soil moisture content rather than the pore network is the most important factor controlling infiltration into frozen soils (Granger et al. 1990).
1984). In chemical fallow fields, higher soil moisture than in cropped fields may reduce the
effect of the pore network (Pomeroy and Gray 1995). Greater moisture in fallow fields will
reduce infiltration into frozen soils and to a lesser extent into soils in general. In unfrozen soils
with similar pore networks, sorptivity and infiltration will decrease with increased soil moisture
but both the effectiveness of the pore network and the soil moisture content contribute to the
variation in infiltration rates observed among different soils (Azooz and Arshad 1996).

2.1.4 Disrupting soil aggregates:

In addition to disrupting soil pores, tillage also disrupts soil aggregates. This results in
smaller aggregate sizes that can be more prone to erosion and loss of particulate nutrients
(erosion is discussed in more detail in Section 2.1.5). The disruption also exposes soil organic
matter (SOM) to degradation and releases carbon and nutrients (Balesdent et al. 2000). Breaking
of the native prairie for crop production led to losses of as much as 50% of SOM in the first 30
years of cultivation (Janzen 2001). Reduced tillage practices including conservation and zero
tillage are reversing soil degradation, but any tillage pass can cause some aggregate disruption
(Larney et al. 1997; Six et al. 1999). From a water quality perspective, the release of nutrients
from SOM is a source of nutrients in runoff water. While nutrient release from SOM likely
occurs with every tillage pass, the effects are most evident in nitrate concentrations in snowmelt
runoff from soils that had been tilled the previous fall either after crop harvest or during fallow
(Elliott et al. 2001; Tiessen et al. 2010). Although breakdown of SOM will release nitrogen (N)
and P, N is present in greater quantities and its conversion to mobile forms is more rapid than
that of P. In annual cropping systems, nitrate released from SOM due to spring tillage might be
taken up, at least in part, by the growing crop. However, nitrate released in the fall after harvest
is available for transport in snowmelt runoff the following spring.

2.1.5 Mixing of soil and crop residues:
Conservation tillage can provide protection for the soil surface during rainfall events, thus reducing erosion and the transport of sediment and particulate nutrients in the runoff water (Balesdent et al. 2000). However, the mixing of surface soil and incorporation of plant residues that occurs with tillage can also affect the availability of nutrients for transport in runoff water.

Under conservation and zero tillage, crop residues, SOM and surface-applied nutrients can become concentrated in the surface layer of soil (Sharpley et al. 1992). This stratification of residues and nutrients is of particular concern for P, which is retained near the soil surface while N transformations allow for leaching into the soil profile (Liu et al. 2014a). Nutrient-rich residues in the runoff interaction zone that are subjected to freeze-thaw cycles become a source of dissolved nutrients in surface runoff (Hansen et al. 2000, 2002) and can increase downstream P loads (Messiga et al. 2010).

When rainfall-generated runoff dominates annual runoff, the surface protection afforded by the crop residues in conservation tillage fields acts to reduce downstream P loading by reducing erosion and the subsequent transport of sediment and particulate P (Bundy et al. 2001). However, during snowmelt runoff over frozen soils, more dissolved nutrients are transported than particulates. Therefore in regions where snowmelt is the primary process for recharge, conservation tillage can result in increased downstream P loads (Tiessen et al. 2010). In some ways conservation tillage appears to mimic natural systems such as native grasslands that are also stratified. However, the closed ecological system that limits nutrient exports from a native system (Timmons and Holt 1977) cannot be achieved in cropland where nutrients removed in harvested grains are replaced by fertilizers.

2.1.6 Redistributing nutrient-rich soils in the landscape:

Tillage erosion moves soil incrementally downslope with each pass. In the long-term, this results in nutrient-rich surface soil being delivered to areas of concentrated flow where the
nutrients are more likely to be transported in runoff water (Li et al. 2007). However, these effects of soil redistribution are difficult to distinguish from other processes that lead to deep, nutrient-rich soils in these landscape positions (Pennock et al. 1987), and as a result, the extent to which tillage erosion affects nutrient availability in the soil-water interaction zone is hard to quantify.

3 Perennial Forage, Grasslands and Grazing

In a manner similar to soil tillage, perennial forages and grasslands can affect the physical characteristics of soil, resulting in impacts on hydrology, as well as changing the nutrient supply to runoff water. There are a wide range of vegetation types (e.g., legumes, grasses, native species) and management (e.g., stand age, harvesting, grazing) that influence the effect of perennial vegetation on hydrology, soils and nutrient transport. Mixed farms usually include several years of forage production as part of their crop rotations, while on ranches, pastures and hayfields are usually only occasionally disturbed to improve productivity.

In the United States and Canada, programs promote the conversion of marginal agricultural cropland to perennial vegetation for soil conservation, and census data for the Canadian Prairies show that livestock numbers and the proportion of cropland that is in forages are increasing (Clearwater et al. 2016). Although both the United States and Canada have programs to promote perennial vegetation, there are important differences between them. The Conservation Reserve Program in the United States does not allow grazing or cutting of the vegetation, but lands in the Canadian Permanent Cover Program can be harvested and grazed.

3.1 Forage, grasslands and grazing – effects on hydrology, erosion, and transport of dissolved nutrients
In this analysis, we consider three categories of grassland vegetation (Table 4): forage in a rotation, long-term grassland, and grazing land; and explore how management practices affect water quality through effects on hydrology, erosion and transport of dissolved nutrients. Within each category, the unique prairie snowmelt situation and rainfall-generated runoff are reviewed.

### 3.1.1 Hydrology:

Grasslands affect hydrology through three mechanisms. First, the height of residues remaining in the fall can affect snow trapping and subsequently the volume of water in the snowpack at snowmelt. A second effect arises from the undisturbed root channels and improved soil structural stability that develop in grasslands and act to increase infiltration relative to annual cropland. Infiltration may also be increased through the third mechanism of lower soil moisture content due to the longer growing season and greater water use by grass stands (Entz et al. 2002).

In drier regions of the Northern Great Plains, tall standing stubble from annual crops has been shown to decrease the amount of snow blowing off exposed fields and increase the snowpack that accumulates during the winter months (Willis et al. 1969). The deeper snowpack increases meltwater and subsequently runoff over the frozen soil (Gray et al. 2001). A similar effect occurs when tall stands of grasses and forages are left on fields in the fall and trap snow to melt in situ (Fang et al. 2010), increasing runoff volumes and downstream nutrient loads.

Therefore, in drier conditions where tall stands may increase snow trapping, cutting forage late in the fall to leave a short stand to overwinter may decrease runoff, although risk of winterkill of the forages may increase.

Improved infiltration can occur under perennial vegetation as compared to annual crops. This may help offset the potential effect of increased snow trapping by perennial forages and grasslands (Bodhinayake and Cheng Si 2004). Even a relatively young forage stand may increase infiltration of rainfall by protecting the soil surface from sealing due to rainfall impact (Tomanek...
As the stands age, development of continuous pores, such as root channels and cracks, and improved soil structure and aggregate stability (Fuentes et al. 2004) will also reduce runoff. Young stands will likely only improve infiltration of rainfall-generated runoff (Liu et al. 2014b) but improved infiltration during snowmelt can occur as the stand ages as a result of soil structural development (van der Kamp et al. 2002). Improved snowmelt infiltration can also occur even in the absence of soil structural development. If perennial vegetation, and the longer season of water use results in drier surface soils going into winter than under annual crops, this reduced fall moisture would allow increased infiltration into frozen soils (Granger et al. 1984).

The broader landscape context also impacts the net effects of forage on runoff. Preventing snow from blowing off the field will likely increase runoff volumes at the field edge, if snowmelt infiltration is not enhanced (Liu et al. 2014b). However, when larger spatial scales are considered, this trapping of snow by tall forages limits snow redistribution into water channels and wetlands more than when wind-blown snow moves freely (Gray et al. 2001). Finally, if improved infiltration results in less runoff of snow, then runoff water volumes in streams and wetland water levels could decrease (van der Kamp et al. 1999).

When the effects of grazing perennial vegetation are also considered, there are some additional considerations with respect to hydrology. Summer stocking rates need to be carefully managed to avoid overgrazing. Overgrazing can negatively affect pasture and soil quality and hence reduce infiltration of rainfall compared to a healthy pasture (where the soil surface is structurally stable and protected by plants and a thatch layer; Naeth et al. 1990). Late fall grazing has a similar effect as a late haying operation and can reduce snow accumulation by reducing the height of grasses in pasture (Willms and Chanasyk 2006). This could have the beneficial effect of reducing runoff volumes in pastures in humid regions of the Northern Great Plains, but in
drier regions, snow accumulation is necessary for soil water recharge (Naeth and Chanasyk 1995) and/or preventing winterkill of forage in cold regions.

In-field cattle wintering is growing in popularity in western Canada because the in-field deposition of manure nutrients reduces the costs incurred for the transport and application of manure from a feedlot (Kelln et al. 2011). In the winter, in-field grazing is less extensive than during the snow-free period because livestock congregate around feed supplies, water and shelter. Siting of winter feeding areas in hydrologically disconnected locations is a key to the successful implementation of the practice because any runoff from these areas will have elevated nutrient concentrations (AAFC 2015). Infiltration is not usually affected within the feeding areas as long as the animals are only present when the soil is frozen, but if the surface soil thaws while the animals are present compaction and reduced infiltration will occur. Reduced albedo (light reflectance) in the feeding areas due to the presence of feed and feces in the snow results in an earlier melt than on un-stocked areas while compaction of snow can slow the rate of melt extending the duration of runoff period (Kongoli and Bland 2002).

3.1.2 Erosion and Particulate Nutrients:

Perennial vegetation provides year-round protection to the soil surface and can decrease erosion and transport of particulate nutrients in surface runoff (Renton et al. 2015). In addition, improvements in soil organic carbon and structural stability under perennial vegetation that may increase with stand age have also been reported (Naeth et al. 1991). The effect of surface soil protection on erosion will depend on the quality of the perennial vegetation stand. Additionally the effects of perennial vegetation relative to annual cropping will also depend on the residue management in the annual cropping system. Where the perennial vegetation is sparse, erosion and particulate nutrient transport may not be reduced by perennial vegetation when compared to reduced tillage with residue retention which provides surface protection throughout the year.
(Cade-Menun et al. 2013; Miller et al. 2013). Conversely, when the perennial vegetation provides good coverage and annual crop management includes fall and pre-seeding tillage, the concentration of particulates in runoff may be reduced under perennial vegetation (Liu et al. 2014b).

Grazing livestock on perennial vegetation may damage soil structure and increase erosivity if the stocking rate is too high (Renton et al. 2015), but well-managed pastures should retain the surface protection and soil structural stability benefits of ungrazed land (Chanasyk and Woytowich 1987). If pastures are managed to maintain adequate residue cover, sediment and P losses are the same as from ungrazed grasslands (Haan et al. 2006). Direct access of livestock to streams, however, can increase transport of particulate nutrients through increased streambank erosion (Marlow et al. 1987) due to trampling and direct deposition of feces in the stream (Ballard and Krueger 2005). Streambank erosion by livestock can be prevented by excluding livestock from the riparian areas by fencing (Miller et al. 2010) or reduced through the provision of an alternate water source that can reduce the time livestock spend accessing the stream (Rawluk et al. 2014). While erosion is generally not a problem during winter feeding, both feces and uneaten feed are sources of particulate nutrients (Chen et al. 2017).

### 3.1.3 Dissolved Nutrients:

Perennial vegetation can act as a major source of dissolved nutrients to surface runoff water, largely due to tissue nutrient release into the snowpack. As a result, conversion of annual cropland to perennial forage can result in no net change, or an increase in total nutrient transport, despite decreased erosion (Christensen and Kieta 2014; Liu et al. 2014b). The release of nutrients from vegetation is greatest during snowmelt because nutrients released due to cell rupture on freezing are retained in the snowpack and contribute to the dissolved nutrient load in snowmelt runoff (Elliott 2013; Kieta and Owens 2018). In an eight-year paired watershed study in southern
Manitoba where annual cropland was converted to alfalfa forage for three years (replicated in two pairings), total runoff losses of P in snowmelt from alfalfa forage land were 160% greater than those from annual crop land, due in large part to a 221% increase in losses of dissolved P (Liu et al. 2014b).

Some evidence suggests that nutrient release during freezing decreases as the age of the stand increases and would therefore be less in permanent cover than in forage seeded in a crop rotation. The quantity of dissolved nutrients stored in vegetation can also be reduced by a late cut of hay that decreases that amount of vegetation on the soil surface going into winter. As discussed previously, shorter vegetation allows for greater snow redistribution and accumulation of snow in stream channels and wetlands. These deep accumulations melt in situ and generally have lower concentrations of dissolved nutrients because contact between the snowpack and nutrient sources is less and because the concentrations of dissolved nutrients in meltwater are diluted by the accumulated water.

Another source of dissolved nutrients in perennial vegetation stands is the accumulation of P in the undisturbed surface soil (Liu et al. 2014b). This P stratification is similar to the effect discussed previously in the section on soil tillage. While undisturbed grasslands may develop a balance similar to native grasslands through time, newly-seeded forages and systems that are managed for production will not have the same ecological balance as natural systems. Nutrient management in perennial vegetation plays a role in determining the source of dissolved nutrients for transport. Soil fertility can significantly influence nutrient content of forages. For example, alfalfa tissue P has been observed to increase with plant available soil P (Schneider 2014). When soil nutrients are substantially greater in annually cropped land than under perennial vegetation, losses of dissolved nutrients can be correspondingly greater (Cade-Menun et al. 2013). Although perennial forages harvested for hay may often remove more nutrients than an annual crop,
forages often receive less fertilizer than annual crops and are less likely to have nutrient surpluses. However, when forages are fertilized to maintain production, appropriate nutrient management to avoid surpluses should be employed.

Detailed study of soil P forms under perennial forage fields with contrasting levels of P fertilizer input and Olsen soil test P indicates that while orthophosphates are present in greater abundance in soils under fertilized forage fields, the amount of mineralizable diesters may differ little between fertilized and un-fertilized fields (Schneider et al. 2016). Greater arbuscular mycorrhizal fungi colonization of alfalfa in unfertilized fields and no significant difference in yield based on soil test P (Schneider 2014; Schneider et al. 2015) indicate that maintaining relatively low levels of water extractable soil P under perennial forages may be possible without negatively affecting yield. The use of forage fields for disposal of manure nutrients can lead to accumulation of nutrients in soil and increased losses of dissolved nutrients in surface runoff and groundwater (Chang et al. 2005). Manure should be applied as a fertilizer in balance with forage removal following the principles outlined in the section on nutrient management (Section 8). Nutrient balances should also take into account N fixed from the atmosphere by legume forages such as alfalfa and clover (Chen et al. 2001).

During warm season grazing, livestock convert nutrients in the forage that they consume to digested nutrients in manure that are more readily available for transport in runoff (Haynes and Williams 1993). Stocking rate affects the ratio of digested to undigested nutrients with higher stocking rates resulting in potentially greater transport of nutrients in surface runoff. Although rotational grazing generally uses higher stocking rates to better utilize pastures, the practice could also be used to reduce dissolved nutrient loads in surface runoff through careful planning of livestock movement (Bailey and Brown 2011). Pastures that drain to streams and water bodies can be stocked in spring and late summer, but avoided at times when runoff-generating storms
are most likely and in fall when feces and urine that are deposited will be available for transport in snowmelt runoff.

The high stocking density employed during in-field winter feeding systems where feed is imported (e.g., bale-grazing) results in a large source of readily available dissolved nutrients in feces and urine for transport in snowmelt runoff (Smith et al. 2011) and repeated wintering on the same site can result in elevated soil nutrient concentrations that could also contribute dissolved nutrients to surface runoff (Chen et al. 2017). Some in-field winter feeding systems utilize stockpiled forage that was grown in the wintering field and do not import nutrients to the field. However, since it is the nutrient balance, the stocking density, and the availability of nutrients relative to the spring freshet that control nutrient transport in snowmelt, care must be taken in siting all winter feeding fields. The timing of the in-field winter grazing period relative to snowpack accumulation may also affect dissolved nutrient loads because nutrients incorporated in the snowpack may be transported more readily than those compacted in ice lenses at the base of the snowpack, but we are not aware of any research to support this hypothesis.

4 Surface Drainage

Drainage of agricultural land can involve a variety of practices, all with the intent of increasing productivity by moving water away from poorly drained sites. Given the low gradient and limited natural drainage in many areas of the Northern Great Plains and the economic benefits of draining to farmers, it is clear why artificial drainage has been pursued extensively. However, drainage can have significant environmental and societal consequences. Surface drainage – the dominant drainage practice in this region – has led to new hydrological connectivity in some
basins that would have naturally had only internal drainage. This connectivity is often associated with poor water quality, erosion, increased flows (and flooding), and loss of important wildlife habitat (Johnson et al. 2008; Blann et al. 2009; Schottler et al. 2013).

Debate and conflict over drainage has been ongoing in North America for more than a century (Blann et al. 2009). Drainage policies and their enforcement have varied, and continue to vary regionally. However, the changes in policy with time are most striking -- from early policies of cooperative work to facilitate surface drainage and maximize land under production, to more recent moves to arrest or reverse effects of past drainage, and manage excess water in light of increased flood risk (MacKenzie and Lieslar 2013; Prairie Habitat Joint Venture 2016; Breen et al. 2018).

Drainage practices include adding surface drainage (open ditches), straightening and deepening existing waterways to improve hydraulic capacity, diking to prevent water from over-spilling into fields, and pumping. They also include subsurface (tile) drainage (section 5), and infilling ephemeral wetlands to prevent ponding. Drainage practices can affect all classes of wetlands, but the greater ease of draining small shallow wetlands suggests that ephemeral, temporary and seasonal wetlands have undergone greater declines than permanent and semi-permanent wetlands (Miller et al. 2009). It is likely that these different classes of wetlands have varied effects on hydrology and nutrient export.

4.1 Effects of surface drainage on hydrology and nutrient transport:

Within the Northern Great Plains, despite the extensive surface drainage that has occurred (Dahl and Allord 1996), we lack even a basic understanding of the effects of surface drainage on discharge, and nutrient concentrations. The effects of drainage on discharge are complex and difficult to attribute to a single driver due to the complex structure of watersheds, highly variable...
climate and importance of memory effects in governing discharge (Skaggs et al. 1994; Dumanski et al. 2015; Ehsanzadeh et al. 2015). Nonetheless, there is concern that drainage can contribute to increased downstream flood risk and elevated nutrient export (Skaggs et al. 1994; Bower 2007; Schindler et al. 2012; Armstrong 2018).

Relatively little information exists on the effect of drainage on nutrient concentrations (Prairie Habitat Joint Venture 2016). There is a tendency for drained wetland soils to show increased nitrification, and a greater tendency for P desorption over time (Brown et al. 2017). High concentrations of nutrients in runoff are apparent in the few studies of drainage that exist (Brunet and Westbrook 2012; Armstrong 2018). In the Smith Creek basin (southern Saskatchewan), there is a relationship between drainage extent, and nutrient transport – indicating that a greater proportion of fully drained wetlands was associated with greater transport of nutrients (Armstrong 2018). These results reflect work in a wet period – perhaps indicating maximal impact. Functionally, while drainage is expected to have the effect of increasing nutrient loads, at least within wet periods, the magnitude of this effect will be spatially variable, and highly sensitive to hydrologic conditions. This makes the definition of mitigation requirements, if required, extremely challenging.

Direct drainage mitigation options fall into two main classes. The first group is practices to ‘keep water on the land’ – protecting and restoring existing wetlands, building new water storages, as well as efforts to prevent new drainage (see section 7 Small Reservoirs, Holding ponds and Wetlands). The second main class is altering the design and management of ditches and drains. Here we discuss controlled surface drainage, and other aspects of channel design and maintenance that may yield benefits to managing flows and nutrient export.

4.1.1 Controlled surface drainage to mitigate drainage impacts:
Controlled surface drainage is a drain (or channel) design feature that can help reduce both nutrient export and downstream flood risk associated with surface drainage via management of water levels within drains using infrastructure such as low-grade weirs and slotted board risers (Kröger et al. 2011). It is a practice with relatively low-cost infrastructure, allows for management of water that increases residence times (the average length of time water stays within a system), and decreases erosivity and downstream flood risk. In locations outside the Northern Great Plains it has been found to increase nutrient retention (Kröger et al. 2011; Baker et al. 2016), although increased P losses may occur where anaerobic conditions increase P mobility (Amarawansha et al. 2015; Jayarathne et al. 2016). In some regions of the Northern Great Plains (e.g., southern Saskatchewan; MacKenzie and Lieslar 2013) controlled surface drainage has been identified as a priority practice to reduce downstream flood risk.

While controlled drainage has shown nutrient retention benefits in diverse landscapes (Kröger et al. 2011; Skaggs et al. 2012), there are many questions regarding spatial variability in nutrient retention within ditches, and optimal ditch design and management to maximize nutrient retention while maintaining key hydrological and erosion control functions. As well, where snowmelt dominates hydrology, controlled surface drainage may have lesser effects when compared to other regions, due to the importance of residence time in controlling nutrient retention in ditches (Kröger et al. 2008, 2011; Baker et al. 2016). Given the prevalence of dissolved nutrients in nutrient transport (Corriveau et al. 2011), in-channel processes that store or remove dissolved nutrients, such as sorption, plant and algal uptake, and denitrification may be more important to understanding nutrient retention than sedimentation. However, the importance of stream channels as a source of sediments in some low-gradient areas (Koiter et al. 2013) suggests that controlling erosion within some parts of the region remains a management priority.

4.1.2 Other aspects of channel design and maintenance to mitigate drainage impacts:
Other aspects of channel design and maintenance may help manage hydrological and biogeochemical impacts of drainage. Restoration of in-stream wetlands and setting aside conservation easements may lead to benefits. Reconnecting rivers with their floodplains has benefits in terms of nitrogen retention, sediment transport, and reduced peak flows in some landscapes (Madramootoo et al. 2007) and engineering channel hydraulics and morphometry, for example constructing two-stage ditches where a floodplain - or bench- is integrated into the ditch design) to simulate more natural stream-structure may also help limit transport of phosphorus, nitrate, and suspended sediments (Mahl et al. 2015).

Performing channel maintenance via dredging or vegetation management may also alter the hydrological and chemical characteristics of ditches (Newbury and Gaboury 1993; Dollinger et al. 2015), although the types of impacts may differ in this region due to the importance of cold regions hydrological processes. Grassed waterways are sometimes associated with increased nutrient retention, in part due to slowing the water flows and increasing sedimentation. However, the lack of significant plant biomass during snowmelt, low particulate loads, and potential release of nutrients from vegetation suggest harvesting ditch vegetation may actually be preferable (see section 6 Vegetated Riparian Buffers, Channels, and Filter Strips). In the Northern Great Plains, the role of ditches in nutrient uptake is largely unknown, although a lack of retention has been shown for short ditches during snowmelt (Brunet and Westbrook 2012).

5 Subsurface (tile) drainage

Subsurface drainage, commonly referred to as tile drainage, is rapidly expanding in wet portions of the Northern Great Plains such as areas in the northern portion of Minnesota (Sands et al. 2012) and North Dakota (Scherer et al. 2015), as well as southern Manitoba (Cordeiro and Ranjan 2012; Satchithanantham et al. 2012). This expansion is associated with the current wet
cycle in climate, combined with high land prices, high crop values and regional issues of salinity (Cordeiro and Ranjan 2012; Sands et al. 2012; Kyle et al. 2013; Rahman et al. 2014; Scherer et al. 2015). Adoption of this practice is motivated by the agronomic benefits such as removal of excess water in the soil profile, salinity control, and timeliness of field operations (Bernstein and Francois 1973; Van Schilfgaarde 1974a, 1974b), which lead to improved crop growth and higher profitability of farm operations. Reduction of the moisture content of the upper soil layers (i.e., root zone) due to lowering of the water table also increases the bearing strength of the soil and trafficability (Young and Ligon 1972). It also favours gas exchange at the soil-atmosphere interface, enhances soil oxygenation, increases soil temperature, promotes soil structure, prevents nutrient losses through leaching and gaseous emissions, prevents crop diseases, and positively affects soil biology (Bower 1974; Van Schilfgaarde 1974b, 1974a; Hillel 1998; Sinclair et al. 2003).

While tile drainage design criteria are still evolving for this region, and could affect crop performance and nutrient export, recent investigations in Manitoba have shown promising agronomic results. Yield increases reported for potato ranged from 15 to 32% when tile drainage was combined with irrigation (Satchithanantham et al. 2012). Similarly, corn yield increases ranged from 2 to 22% for different combinations of irrigation and tile drainage (Cordeiro and Ranjan 2012). The agronomic benefits of tile drainage observed in Manitoba are in line with the improved crop performance that fostered tile adoption in more humid areas of Canada such as Ontario (Stonehouse 1995).

5.1 Effects of Subsurface (tile) drainage on hydrology and nutrient transport:

Despite extensive research carried out in the US Midwest (Randall and Mulla 2001; Dinnes et al. 2002; Randall and Goss 2008), the effects of subsurface drainage on nutrient transport are
not well characterized in the northern portion of the Northern Great Plains (i.e., northern MN and
ND, as well as the southern Canadian Prairies) in part because tile drainage is relatively new in
this region (Cordeiro and Ranjan 2012) and there is limited research on its effects in cold
climates (King et al. 2015). Tile drainage has been considered by some to be a hydrologic control
BMP (Scott et al. 1998). Increased infiltration promoted by tile drainage can reduce the
proportion of total outflow occurring as surface runoff, as demonstrated in several regions of
North America (Walter et al. 1979; Skaggs et al. 1994; Calhoun et al. 2002; Stillman et al. 2006;
Schilling and Helmers 2008). Given that surface runoff is one of the major causes of sediment
transport and is associated with nutrient transport (Sharpley et al. 1992, 2002; He et al. 1995;
Aksoy and Kavvas 2005; Koiter et al. 2013), reducing surface runoff could help control nutrient
export. However, tile drainage performance in the Northern Great Plains may not be the same as
in other regions of North America due to its cold climate. Soils may still be frozen during spring
runoff in cold-climate regions, and this can restrict infiltration (Gray et al. 2001; Shook and
Pomeroy 2012) and affect the hydrology of tile-drained fields (Luo et al. 2000).

Most tile outflow (>70%) occurs during snowmelt and the early growing season (March-
June) in the Northern Great Plains (Jin and Sands 2003; Satchithanantham and Ranjan 2015).
Frozen soils in spring can limit infiltration in the early stages of snow melt and the efficacy of
tile drainage in reducing runoff. Recent modelling investigations using the SWAT model in the
Upper Red River of the North Basin encompassing portions of South Dakota, North Dakota and
Minnesota suggest that more water could be moved from the basin to the rivers in the fall season
by tile drainage, thus creating more storage capacity in the soils (Rahman et al. 2014). However,
it appears that increases in soil storage capacity had negligible effects on monthly flow volumes
in the following spring (Rahman et al. 2014), possibly due to frozen soils.
Preferential flow has an important influence on nutrient export from tile-drained annual cropland and perennial forages in the United States and more humid parts of Canada (Beauchemin et al. 1998; Shipitalo and Gibbs 2000; Simard et al. 2000; Jamieson et al. 2002, 2003; Hoorman and Shipitalo 2006; Ball Coelho et al. 2007; Lapen et al. 2008; Eastman et al. 2010; King et al. 2015). However, this pathway is not well characterized in the northern portion of the Northern Great Plains within the tile-drainage context, despite evidence that water is flushed down rapidly via cracks and fissures in the root zone during snowmelt in cracked heavy textured soils in the Canadian Prairies (Joshi and Maulé 1999). For example, in Manitoba, where tile drainage is quickly expanding (Cordeiro and Ranjan 2012), more than 40% of farmlands had moderate to very high likelihood of crack flow (Dadfar et al. 2010). A similar situation would be expected in the clay-rich areas of North Dakota and Minnesota, in which tile drainage is also expanding (Jia et al. 2012; Rahman et al. 2014).

In these northern regions, there has been limited assessment of the effects of tile drainage on nutrient export, but evidence from other regions demonstrating very large contributions of tile-drained fields to watershed export (Kleinman et al. 2015) suggests that management of drainage waters may be important in controlling agricultural nutrient export. Existing data suggest large exports of nutrients can occur from tile-drained fields in the Northern Great Plains. Rates of nitrate export from tile drains are primarily driven by fertilizer application rates, with tile drainage design (i.e., depth and spacing of tile drains) as the second most important factor (Davis et al. 2000; Sands et al. 2008). Work in southern Manitoba has shown nitrate loads are particularly high from recent tile-drain installations during higher than normal precipitation (Cordeiro et al. 2014; Satchithanantham et al. 2014). Nitrate loads from the recently drained fields were larger than most values reported for Canada and the United States (Randall and Goss 2008) and were comparable to the amount of N fertilizer applied to corn (Cordeiro and Ranjan...
2012) and potato (Satchithanantham et al. 2012), respectively. It is likely that the extremely wet
conditions during this study (i.e., 57% greater May-Sept precipitation than the 30-year average;
Cordeiro and Ranjan 2012) led to a rapid flush of nitrate that had accumulated in the soil profile
through these newly installed drainage systems. High rates of nitrate flushing through newly
installed tile drainage systems have been reported previously for farmland in Europe
(Hodgkinson et al. 2002). It is also possible that specific farm practices (e.g., manure
application), as well as the recent history of fertilizer-intensive crops (Cordeiro et al. 2014) have
influenced this initially elevated flush. Thus, these high outputs may not represent
general conditions in the Northern Great Plains. Lower export was reported in the following
year, bringing values to within the ranges observed elsewhere in Canada (Cordeiro et al. 2014).

While the above results also suggested the important role of wet conditions in hastening
higher N export, P export coefficients from these Manitoba sites (Cordeiro et al. 2014;
Satchithanantham et al. 2014) were within the ranges reported in the literature for eastern Canada
(e.g., Goulet et al. 2006). Nonetheless, given the sensitivity of aquatic ecosystems to P
concentrations at μg L⁻¹ levels (Schindler 1977), P export associated with tile drainage here
(Cordeiro et al. 2014; Satchithanantham et al. 2014), and elsewhere (Hergert et al. 1981b, 1981a;
Jia et al. 2012), can have significant environmental effects.

5.2 Mitigation of subsurface drain nutrient losses through controlled drainage,
bioreactors and physico-chemical treatments

Controlled tile drainage, where there is a restriction on drainage water leaving the field,
can reduce nutrient export and keep the water table at a shallower depth more easily accessed by
crops at times when their water demand is high (Crabbe et al. 2012; Gilham et al. 1979; Skaggs
et al. 2012; Sunohara et al. 2015). The practice has been proven to be quite effective in
controlling nutrient export in North America (e.g., Adeuya et al. 2012 [Indiana]; Helmers et al. 2012 [SE Iowa]; Sunohara et al. 2015 [Ontario]) and research into the practice on the Northern Great Plains has also shown substantial reductions in nitrate and P loads (Lalonde et al. 1996; Stacey et al. 2010; Cordeiro et al. 2014; Satchithanantham et al. 2014). However, unexplained increases in flow-weighted mean P concentrations have been reported in some instances (Stacey et al. 2010; Satchithanantham et al. 2014), suggesting that this practice needs further investigation in the region.

One ‘end of pipe’ solution to high nutrient export from drained lands is the use of bioreactors and physico-chemical treatments for nutrient removal. Woodchip bioreactors are the most tested technology, where simple structures intercept drainage waters and facilitate nutrient removal through addition of a carbon source to stimulate biological denitrification that converts nitrate to nitrogen gas. Physico-chemical treatments placed after bioreactors have also been proposed to improve nutrient removal using a wide variety of materials (Jaynes et al. 2008; Buda et al. 2012; Hussain et al. 2014; Hua et al. 2016; Roser et al. 2018). For example, Salo et al. (2015) demonstrated that a woodchip bioreactor system with iron media as post-treatment removed nitrate and phosphate from subsurface drainage. While bioreactors are typically applied for tile drains, they can also be used in surface drainage (Pfannerstill et al. 2016; Pluer et al. 2016). Bioreactor technology has not been directly assessed in Northern Great Plains landscapes, but given that bioreactors tend to be most effective under warm conditions and at low-flow volumes (Pfannerstill et al. 2016), they may not be ideally suited to snowmelt dominated drainage. However, with adequate effluent storage capacity or additional chemical amendments (e.g., physico-chemical treatments such as P sorbing materials) and sufficient retention time to enable substantial nutrient removal (Bock et al. 2015; Addy et al. 2016; Hoover et al. 2016), the relatively low cost, and small footprint (Moorman et al. 2015) of bioreactors and chemical
nutrient removal technologies may make them an appealing nutrient management BMP in this landscape.

6 Vegetated Riparian Buffers, Channels, and Filter Strips

Many conservation organizations in the Northern Great Plains promote the management of vegetation in and adjacent to surface water flowpaths (referred to in this section collectively as buffers). A wide-range of environmental benefits can result from removing erosion-prone pathways of concentrated surface water flow (often referred to as gullies) and near-to-stream cropland from production. These effects include: prevention of bank or channel erosion (Sweeney et al. 2004; Wood 2004), removal of sediment from overland flow (Mander et al. 1997; Owens et al. 2007; Kröger et al. 2013), carbon fixation and storage (Guyette et al. 2002; Tufekcioglu et al. 2003), acting as islands of habitat or corridors for wildlife in agricultural landscapes (Chen et al. 1999), and regulation of stream temperature (Brosofske et al. 1997).

Perennial vegetation can impact nutrient transport and retention on the Northern Great Plains via several mechanisms and as noted in the preceding sections of this review, most movement of N and P with runoff in the region occurs with snowmelt in a dissolved form. The mechanisms by which vegetated buffers in cold climates influence phosphorus dynamics are outlined in detail in a recent review by Kieta et al. (2018), where the authors note that vegetation can stimulate dissolved N and P retention via plant uptake and translocation of nutrients from runoff, soil water, and groundwater into plant tissue or microbial biomass. Retention may also occur with sorption of dissolved nutrients to soils, by preventing erosional transport of particle-bound nutrients, by producing detritus fueling microbial cycling (e.g., denitrification), and by altering infiltration rates or flow paths (Dosskey et al. 2010). Most research into the effect of
riparian buffers on N and P export has focussed on regions dominated by rainfall-runoff, with much less research occurring at locations where snowmelt runoff predominates (Kieta et al. 2018). In Finland, forested riparian buffers retained a moderate proportion (16%) of added $^{32}$P during snowmelt (Vaananen et al. 2006), with the greatest portion of this retention occurring in the soil. This pattern highlights a significant challenge in using buffers to retain nutrients from snowmelt runoff; that vegetation in early spring is likely to be present only in a dead or dormant state and may act as a source of dissolved nutrients (as described previously).

In addition to observations of low rates of P retention during snowmelt, a number of challenges specific to the Northern Great Plains are likely to limit the efficacy of buffers in retaining N and P from snowmelt runoff. To evaluate the potential effect of riparian buffers on nutrient transport in the Northern Great Plains, important modifications must be made to the dominant conceptual model of how buffers are likely to function within agricultural watersheds. The unique landscape and climatic features of the northern Great Plains create significant differences in the spatial extent, mechanism, and timing of interaction between buffers when compared to warmer regions.

### 6.1 Factors affecting the efficacy of Vegetated Riparian Buffers, Channels, and Filter Strips in the Northern Great Plains

Historically studies on the efficacy of buffers have focused on hillslope processes, erosional transport, and sheet flow through buffers. However, in the Northern Great Plains these processes are of lesser importance due to the presence of frozen soils and vegetation, concentrated flow paths (channelized rather than evenly dispersed flow), prevalence of flooding, and dissolved nutrient transport. The impacts of frozen soils and vegetation on riparian buffer hydrology and biogeochemistry are similar to those mentioned earlier in the manuscript for
upland soils (reduced rate of infiltration with prevalence of surface flow and release from vegetation following freeze thaw cycles) and have been reviewed in detail with reference to vegetated buffer phosphorus dynamics in cold regions by Kieta et al. (2018). The following section will focus primarily the other three key factors.

**6.1.1 Concentrated flow paths:**

Under low and moderate flow conditions, most runoff travels quickly through buffers as channelized rather than sheet flow, reacting with only a small area of the overall buffer (Stewart et al. 2010). Detailed analysis of water movement during snowmelt in an agricultural watershed located in Poland identified dominant flow paths during melt to be along preferential flow structures (Banaszuk et al. 2013). In the Midwestern United States, most runoff has been observed to breach buffers through concentrated flow paths (Knight et al. 2010), indicating broader prevalence of this challenge beyond the Northern Great Plains. Further research is needed to identify whether changes in width or vegetation type may increase the potential for buffers to disperse concentrated flow over a larger surface area; however, grassed buffers expanding the width of existing treed buffers in the midwestern United States were shown to increase dispersion of rainfall induced runoff (Knight et al. 2010).

**6.1.2 Flooding:**

High peak flow volumes and wide floodplains with low slope characterize many river systems in the Northern Great Plains. As a result, in higher order streams and rivers, riparian vegetation is frequently inundated by flood water (Stewart et al. 2010). This inundation means in years with high flow, the frequency, duration, and spatial extent of contact between floodwaters and riparian vegetation and soils is often greater than for more localized overland flow that is likely to occur through channels of preferential flow. Irrespective of the driver of inundation, within watersheds of the Red River, flooding is common and concentrations of both dissolved
and particulate P tend to increase with flooding (McCullough et al. 2012). Experimental flooding of a range of alkaline agricultural soils typical of the Northern Great Plains showed increased risk of P release to floodwater following increases in pore water and ponded water P concentrations as anaerobic conditions developed during the course of simulated flooding (Amarawansha et al. 2015; Jayarathne et al. 2016). As Amarawansha et al. (2015) note in their discussion of the implications of this result, the research focussed only on disturbed soils with controlled environmental conditions. Potential for P release from soils of non-cultivated, riparian buffers with perennial vegetation remains understudied in the region, particularly in the context of snowmelt-related flooding. Further research is required to develop recommendations for management of riparian vegetation (spatial extent, species selection, harvest effects) in frequently flooded areas.

**6.1.3 Potential for dissolved nutrient release and transport:**

In surveys of changes in runoff P concentration with flow of snowmelt through riparian buffers in the Northern Great Plains, observed responses have varied widely (Kieta et al. 2018), with no change in concentration or increases have being observed to occur with equal (Sheppard et al. 2006) or greater (Habibiandehkordi et al. 2017) frequency to reductions in concentration. A recent Northern Great Plains study by Satchithanantham et al. (2018) evaluated the hydrological and biogeochemical factors controlling dissolved N and P uptake rates with flow of runoff through riparian buffers with both snowmelt and rainfall. The results presented by Satchithanantham et al. (2018) indicate that sorption-desorption with soil is the primary control on dissolved P uptake and release regardless of season or riparian buffer vegetation type, which suggests that the high degree of variation in concentration changes that have been observed with previous studies reflects variation in the degree to which P saturation has occurred in riparian soils throughout the region. Less research has occurred on the factors controlling N retention
and transformation in riparian buffers in the Northern Great Plains, but paired comparisons of nitrate uptake completed by Satchithanantham et al. (2018) during both summer and snowmelt indicate greater potential for uptake in riparian buffers in the summer, suggesting greater biological control of N removal rates as compared to P, and the potential importance of temperature on denitrification and biological N uptake rate.

While sediment-bound nutrient retention remains an important benefit of buffers in northern environments (Uusi-Kämppä et al. 2000; Miller et al. 2016), particulate-associated nutrients are a small proportion of overall nutrient loads in systems with relatively flat topography and frozen soils at the time of runoff. Also of concern is the potential for soil-bound nutrients to be released into the dissolved phase following accumulation (Uusi-Kämppä et al. 2000). The potential for the release of retained nutrients is illustrated in a variety of agricultural systems where retention of sediment-bound P can be followed by a shift from P sink to net source of dissolved P during runoff events (Hoffmann et al. 2009; Dodd and Sharpley 2015; Satchithanantham et al. 2018). Studies of riparian efficiency in retaining P from runoff have focussed primarily on rainfall runoff-dominated geographies, but review of these studies illustrates that the potential for P desorption following retention is a relatively widespread problem (Kröger et al. 2013; Dodd and Sharpley 2015). The potential for N buildup may be less concerning due to permanent removal to N$_2$ following denitrification (Mayer et al. 2007), although similar increases in stream N concentrations (primarily as nitrates) have been illustrated where conditions favour the buildup of a leachable pool of N, release from riparian soils, and transport to the stream (Duncan et al. 2015).

6.2 Managing Vegetated Riparian Buffers, Channels, and Filter Strips
It is often assumed that riparian characteristics that reduce velocity of flow (high density vegetation, surface roughness, and lower slope) are likely to be beneficial in decreasing both particulate and soluble P transport in runoff from upland agricultural sources. Given the dormancy of vegetation during snowmelt, those hydrological conditions that lead to greater contact between runoff water and the soil matrix (increasing infiltration rate, reducing velocity) seem likely to be factors that will define the potential for retention of N and P from runoff. However, increasing hydrologic retention time where there has been a legacy of P loading to riparian buffers may lead to mobilization of a large and leachable pool in the soil or overlying vegetation (Kröger et al. 2013; Dodd and Sharpley 2015; Satchithanantham et al. 2018).

Similarly, where high rates of nitrate loading to riparian areas accompany agricultural runoff, this N source may cause higher rates of microbial activity and emission of greenhouse gases, particularly N$_2$O.

Research conducted in regions other than the Northern Great Plains tends to indicate greater retention with larger buffer widths. In a meta-analysis of riparian buffer effectiveness for reduction of N loads that primarily focussed on the United States Midwest, Mayer et al. (2007) identified those buffers with a width > 50 m as more consistently reducing loads than those of narrower width. In Finland, the percentage of P retained in buffer zones was observed to increase with width, but this pattern was primarily due to retention of sediment-bound P (Uusi-Kämppä et al. 2000). Even where buffers can be designed to effectively retain N and P, the potential for buffers to shift from sink to source with retention and accumulation illustrates that in absence of concurrent reductions of runoff loss from adjacent agricultural land, capacity of downstream approaches to mitigation are likely to be overwhelmed. This challenge does not negate the fact that reductions of N, P and sediment transport have frequently been observed with flow through well-managed buffers of perennial vegetation (Uusi-Kämppä et al. 2000). Establishment of
perennial vegetation along streams and in areas of concentrated runoff from agricultural land still
can be expected to have a positive effect on water quality in the Northern Great Plains when
considering the alternatives of extending cultivation and fertilization to the water’s edge and into
erosion-prone flow paths.

Harvest of hay or biomass from riparian areas and vegetated buffers removes N and P that
has accumulated in vegetation during the course of its growth. Long-term (10 year) harvesting
experiments have demonstrated reductions in the potential for P loss with snowmelt runoff and
reductions in the accumulation of extractable P in surface soil in grassed buffers in Finland when
compared to unharvested control buffers (Uusi-Kämppä 2005). This result indicates that harvest
may act to reduce N and P leaching from vegetation during the snowmelt following harvest and
act as a pathway for removal of P that has accumulated in the soil for a longer period. However,
the effect of timing and frequency of harvest and potential trade-offs with other environmental
benefits of non-harvested wooded buffers (e.g., carbon storage and prevention of channel
erosion) should be further studied on the Northern Great Plains (Kieta et al. 2018). In addition,
the cost effectiveness of vegetation harvest from buffers will be influenced by the increasing
scale of farms and farm equipment. Operation of larger equipment requires increased minimum
buffer width for harvest with common farm equipment and the operation of larger equipment
becomes increasingly problematic as the complexity of field boundaries are increased. Allocation
of vegetative N and P to growth by grazing animals may also represent a mechanism through
which removal may be possible. However, the potential for increased losses with erosion
associated with animal movements and the potential for release from feces and urine deposited
while grazing are important considerations, given that improvements in water quality have been
observed with the relocation of grazing and bedding sites away from riparian areas (Olson et al.
2011).
There is growing interest in ‘keeping water on the land’ (Kanu 2013; McEwan 2013) as a means of mitigating flooding in wet years, maximizing benefits from scarce water in dry years, and attaining environmental benefits in terms of water quality and habitat. Within the Northern Great Plains, on-farm water storage ponds, known as dugouts, are common, and have been for almost a century (Walker 1945) with the water being used as a drinking source, for spraying and to irrigate. Worsening problems of degraded water quality (Schindler et al. 2012), ongoing wetland drainage (Cortus et al. 2011), and costly flooding (Wheater and Gober 2013; Gober and Wheater 2014) have led to widespread interest in reservoirs, holding ponds and wetlands for water storage and nutrient retention.

7.1 Small reservoirs

Small, constructed water bodies such as reservoirs and farm dugouts are common in agricultural lands, covering approximately 77,000 km$^2$ globally, with the area expanding annually (Downing et al. 2006). Although they may be most common in wet regions (Downing et al. 2006), there are many small (< 5 ha) and large reservoirs (e.g., Lake Diefenbaker at 393 km$^2$; Sadeghian et al. 2015) throughout the Northern Great Plains, with growing interest in expanding the coverage of small reservoirs. Landowners in the South Tobacco Creek Watershed in southern Manitoba have made extensive use of reservoirs less than 5 ha in size to help reduce peak flows, and prevent flooding and erosion. An ancillary benefit of these reservoirs is their role in sediment and nutrient removal. Intensively studied reservoirs (two reservoirs 2-3.1 ha in size) showed sediment retention of 49-83% (Tiessen et al. 2011). The proportion of nutrients retained...
was more modest, but remained significant, with average annual N retention varying from 15-20% between the two reservoirs and annual average P retention was between 9 and 12%.

Important to their suitability for the Northern Great Plains, the major form of nutrients retained were dissolved. Reservoirs were most effective at retention in summer rainfall runoff (Tiessen et al. 2011).

7.1.1 Mechanisms of nutrient removal in small reservoirs:

Mechanisms of nutrient removal in small reservoirs include sedimentation of eroded materials and particulate organic material, microbial removal of N via denitrification, and sorption or chemical precipitation of P in sediments (Reddy et al. 1999; Bridgham et al. 2001; Harrison et al. 2009; Galuschik 2015). Nutrient removal is regulated by the residence time (or the average time water remains within a system) with maximal removal facilitated by longer residence times (Paul 2003; Fairchild and Velinsky 2006). While sedimentation can remove the inflowing suspended solids, including sediment-bound nutrients, sedimentation can also limit the lifespan of a reservoir by contributing to infilling (Vörösmarty et al. 2003; Beusen et al. 2005; Powers et al. 2013). Reservoir construction may also result in changes in downstream channels, and bed and bank erosion can be enhanced as the water reacquires its particulate load (Williams and Wollman 1984). Within South Tobacco Creek, rates of sedimentation are not sufficiently high to impact reservoir lifespans (estimated time to infill of 360-1850 years; Tiessen et al. 2011), although this is a concern in many other regions, and is an important consideration when considering the suitability of this BMP.

Denitrification is the permanent removal of nitrate via reduction to N-gases (N\textsubscript{2} and N\textsubscript{2}O). This pathway can be very important in ponds, reservoirs, and wetlands. Nitrogen removal via denitrification tends to be directly related to residence time, with denitrification rates governed by the availability of substrates (nitrate and organic carbon), anoxic conditions, and other factors...
(Seitzinger et al. 2006; Baulch et al. 2011). In the Tobacco Creek area, reservoirs were shown to
have higher rates of denitrification in the open water period than natural stream pools, which
may relate to higher sediment organic carbon, and lower dissolved oxygen in the reservoirs
(Gooding and Baulch 2017). Rates of denitrification were also related to nitrate concentrations
but beyond an apparent threshold of ~ 1.4 mg NO$_3^-$ N L$^{-1}$ in reservoirs, denitrification was
nitrate-saturated, meaning rates no longer increased with increasing nitrate concentrations. This
threshold is higher than was observed in stream pools, but still represents an important constraint
on this N removal process (Gooding and Baulch 2017). It may be feasible to manage these
systems to optimize denitrification. For example, nitrate saturation commonly occurs when
organic carbon becomes limiting, suggesting managing riparian vegetation as an organic carbon
source may help support higher denitrification rates (Stanley et al. 2012). However, these types
of options should be considered in the context of other ecosystem services of these reservoirs,
most notably P retention.

Reservoirs are typically important sinks of P – through retention of P in the sediments
(Jossette et al. 1999; Donald et al. 2015). Sedimentation of particulate P contained within eroded
materials is one important mechanism for P retention. However, in the Northern Great Plains,
where dissolved P export is high, mechanisms which retain dissolved P must also be important
for this BMP to be successful. Biotic uptake (plant and algal removal of dissolved P, followed by
sedimentation) and sorption of dissolved P to suspended solids can be important interactions
within the water column; however, sorption of water column P to benthic sediments and
precipitation reactions may be the most important (Froelich 1988; McDowell et al. 2003).

Phosphorus may be adsorbed or coprecipitated with Ca and Mg-containing minerals.
Binding with Fe and Al oxides can also be important, particularly in regions with non-calcareous
soils (Boström et al. 1988; Olila and Reddy 1993). Within the Northern Great Plains, calcareous
soils are common, and the glacial history of the region has influenced concentrations of Ca and Mg in surface till and groundwater, imposing potentially important spatial gradients. Specifically, higher levels of Ca in the north and east of the Northern Great Plains (Anderson and Cerkowniak 2010), may contribute to important regional variability in phosphate retention via sorption and precipitation. Even within a 1000-km$^2$ catchment, almost 4-fold variation in Ca-concentration was observed in reservoirs (Galuschik 2015). These water column calcium concentrations were strongly negatively related to equilibrium P concentrations (EPC; defined as the concentration of water column P in equilibrium with sediments), suggesting that this variation in Ca may be an important determinant of reservoir P retention, and merits consideration within reservoir siting. In a study that paralleled the denitrification work cited previously (Gooding and Baulch 2017), EPC was measured across 8 reservoirs in the Tobacco Creek watershed. In 65% of measurements, sediments were removing P from the water column while sediments acted as a source in 22% of measurements. In the remaining cases, they were near equilibrium (Galuschik 2015). Negative relationships between dissolved organic carbon (DOC) and EPC suggest that there may be trade-offs in terms of reservoir design to maximize N retention (via denitrification, which tends to be stimulated by DOC), and P retention (where higher DOC is associated with lower EPC). However, the role of DOC in influencing P interactions with soils and sediments is an area where more work is required.

### 7.1.2 Risk of phosphorus remobilization in reservoirs

Although reservoirs typically retain P over annual time scales, the question of whether some of the stored P will be remobilized is likely the most important question with regards to the use of reservoirs as BMPs. Within the Northern Great Plains, very high rates of P release from sediments have been observed in lakes (Orihel et al. 2017), suggesting research is required to assess whether changes in environmental conditions (e.g., low oxygen, changing pH, etc) may
influence P binding and mobilization in these reservoirs. Importantly – EPC is variable in time, with an average 4.6-fold change within reservoirs over a four month period (Galuschik 2015). Assessing the potential for winter anoxia and pH change to stimulate P release from reservoir sediments is an important area for future work. In addition, the potential for nutrient release following reservoir construction associated with inundation of terrestrial materials merits consideration (Ostrofsky 1978).

7.2 Managing feedlot runoff in holding ponds

Management and treatment of feedlot nutrient runoff is a key consideration in light of the potential importance of feedlots in nutrient export (Howarth et al. 2002), and the very high dilution of these runoff waters required to protect aquatic life (Miller et al. 2004). Catchment-scale research where multiple BMPs were implemented in South Tobacco Creek (Southern Manitoba) demonstrated both that significant reduction in nutrient loads was feasible on relatively short timescales, and that installation of a holding pond to manage runoff from a small on-farm beef cattle feedlot was responsible for a large proportion of reductions in nutrient export (Li et al. 2011). The holding pond retained all runoff from the feedlot, which was then used as irrigation water. Holding ponds can be differentiated from lagoons and treatment wetlands in that their purpose is the temporary storage of water, rather than nutrient removal (Dickey and Vanderholm 1977). Despite this, significant nutrient removal can occur during storage, as a result of ammonia volatilization, and potentially also denitrification, if oxic conditions allow generation of nitrate or nitrite (Dickey and Vanderholm 1977).

The chemistry of holding ponds can vary seasonally as a result of microbial processing and chemical precipitation and therefore the agronomic benefits of this nutrient source depend on the timing of its use for irrigation (Dickey and Vanderholm 1977). There are very clear benefits to
managing feedlot runoff, but environmental trade-offs do merit mention, including risk of contamination of surface water and groundwater if containment is insufficient (Prichard et al. 2005).

Pond design must take into account the feedlot hydrology. Like many feedlots, the South Tobacco Creek feedlot contributed a larger proportion of runoff than would be anticipated based on its area, presumably as a result of low infiltration rates, and limited water storage on the site. This effect was most apparent in summer runoff, with differences in infiltration between the feedlot and other areas of the catchment in winter likely dampened by the presence of frozen soils through parts of the catchment (Li et al. 2011), and may also have been influenced by shallow groundwater transport through the drainage system. While work in reservoirs and wetlands suggests that nutrient retention may be increased with longer residence times, management recommendations for holding ponds suggest that ponds be emptied after runoff events to avoid exceeding their storage capacity (Gilbertson et al. 1980). This is important both for snowmelt runoff, and rainfall runoff, with rainfall runoff occurring more frequently in some areas of the region (Miller et al. 2004).

There are numerous design considerations that merit consideration for holding ponds, and feedlots as a whole, and regulations vary across jurisdictions (OMAFRA 2003). Key considerations are ensuring: effective management of water from outside the feedlot area, appropriate feedlot drainage, sufficient pond storage capacity (and appropriate management including emptying to ensure capacity for additional runoff), siting the pond to avoid contamination of adjacent water bodies, preventing seepage, and appropriate techniques for land application (Gilbertson et al. 1980; Ham and DeSutter 1999; Alberta Agriculture and Forestry 2012). Managing, or relocating snow may also merit consideration. In this region, one major challenge can be the need to empty holding ponds in the fall to create storage capacity for spring
7.3 Natural, semi-natural and constructed wetlands

The Northern Great Plains have undergone extensive wetland loss, and while policy is moving towards arresting or reversing this loss (e.g., Johnson et al. 2008; Government of Alberta 2013; Saskatchewan Water Security Agency and Agency 2015; Division and Water Stewardship Division 2018), active drainage is still occurring (Cortus et al. 2011). While the aggregate effects of drainage on water quality are not known, wetland drainage efficiency and extent is an important determinant of nutrient export (Brunet and Westbrook 2012; Armstrong 2018), suggesting that drainage contributes to nutrient mobilization or loss of nutrient sinks in the landscape. A key question for wetland policy and wetland valuation is the role of wetlands in nutrient storage and processing – an area in which very little work has been done in the Northern Great Plains. Some regional studies suggest significant retention may occur in wetlands (White et al. 2000; White and Bayley 2001; Anderson et al. 2013), but these benefits are not always observed (Ontkean et al. 2003) or consistent with results of other cold regions (Braskerud et al. 2005). The mechanisms by which wetlands can improve water quality are similar to processes in reservoirs, and include physical processes (primarily sedimentation but also filtration and volatilization), chemical processes (precipitation, complexation, adsorption) and biological processes (plant, algal and bacterial uptake; Fisher and Acreman 2004; Chouinard et al. 2015).

7.3.1 Natural and semi-natural wetlands vary in their potential to attenuate nutrient export and key mechanisms

Studies of nutrient processing, and nutrient removal efficacy are relatively rare within this region. A northern prairie marsh receiving municipal and agro-industrial wastewater has been
estimated to store 60% of P inputs in its sediments (White et al. 2000), and can remove 87% of inflowing ammonia and 80% of nitrate (White and Bayley 2001). However, results suggest that the sorption capacity for P may be exhausted over time; hence continued high P removal will not be sustainable (White et al. 2000). The calcareous sediments were cited as one reason that P retention was so effective in this wetland (White and Bayley 2001), and emphasizes an earlier call for coordinated research to understand the key factors that determine P retention, given work to date demonstrates a high degree of site specificity (Braskerud et al. 2005). In a southern Alberta wetland that was enhanced for wildlife habitat, significant nutrient retention was not demonstrated; however, other benefits including decreasing total suspended solids and fecal coliforms were shown (Ontkean et al. 2003).

### 7.3.2 Constructed wetlands also show variable nutrient retention capacity

Constructed wetlands are engineered systems that include a variety of designs, some of which may overlap with those previously discussed for reservoirs and holding ponds. They may include sedimentation basins, infiltration basins, surface/subsurface flow matrices, and vegetative filters (Cronk 1996; Vymazal 2007). They may be present in low lying areas where wetlands once naturally occurred, or constructed in areas higher in the landscape. This siting will impact the residence time, and the proportion of a catchment’s nutrients that pass through the wetland. While operating costs are often low, these systems require monitoring, water management, and may require control of exotic plants and burrowing animals. Construction costs can be substantial, with subsurface flow wetlands costing more than surface flow systems, but requiring less land (Rousseau et al. 2008).

Constructed wetlands can be effective at P retention in cold regions (Jenssen et al. 1993), although the retention rates, and proportion of P retained can vary widely (Braskerud et al. 2005; Chouinard et al. 2015). Total P (TP) may be effectively retained (TP retention varied from -6
[net release] to 88% [retention]) and while dissolved P retention is common, significant net release may also occur (-33 to 89%; Koskiaho et al. 2003; Braskerud et al. 2005). Nitrogen retention also varies widely (-7 to 40% for TN, -8 to 38% for NO$_3$-N, and -50 to 57% for NH$_4$-N; Koskiaho et al. 2003).

A constructed wetland on a cattle feedlot near Riverton MB (3-ha 800-head feedlot operation near Lake Winnipeg) operated during the growing season (May-October) showed strong capacity to remove ammonia (~97% when influent concentrations were high) which was believed to reflect strong nitrification capacity. However the system showed limited efficacy for P removal (0-14%; Pries and McGarry 2001). This system included a runoff collection system, settling pond, holding pond, and treatment wetland that had a two month residence time during a relatively dry period. Total suspended solid retention was 60-76%, while fecal coliform removal was also high (68-94%; Pries and McGarry 2001). A second operation constructed and monitored at approximately the same time showed similar efficacy. This treatment wetland at a 1800-head feedlot near Lake Manitoba removed 81-93% of ammonia, 57-71% of TSS, and 52-70% of fecal coliforms and also showed higher P removal capacity at 27-41%.

**8 Nutrient Management Practices**

The relatively dry, cold climate and nearly level topography of the Northern Great Plains means that traditional soil and water management practices such as conservation tillage, perennial forages, and vegetated buffer strips are not always effective for reducing nutrient losses in this region (Figs. 4, 5). Sound nutrient management practices are especially important in this region and must be the foundation of efforts to mitigate nutrient export from agricultural lands (Fig. 6) because of the challenges for intercepting nutrients. Simply put, soil and water management
BMPs cannot fully mitigate the effects of poor nutrient management practices. Sound nutrient management is crucial.

8.1 Special importance of the timing of application

Selecting and using the right rate, source, placement and timing of fertilizer and manure nutrients (known widely as the 4Rs) are essential components of all sound agricultural nutrient management programs. Throughout the world, the principles for sound nutrient management are relatively similar; therefore, the BMPs associated with these aspects of nutrient management in the Northern Great Plains are not unique. However, the timing of nutrient application is a particularly important issue for the Northern Great Plains. Specifically, late fall and winter applications of manure or fertilizer should be avoided, due to the large proportion of runoff that occurs during snowmelt in this region.

The problem of nutrient loss from winter applications of manure was identified many years ago. In Manitoba, Schulte et al. (1979) reported that total N measured in surface runoff from plots receiving winter manure applications was equivalent to 12% of the N applied in manure. Studies in Manitoba also showed that application of manure onto frozen or snow-covered soil also results in large runoff losses of P (Green 1996). In Saskatchewan, Maulé and Elliott (2006) measured large losses of ammonium N (5.9 kg N ha$^{-1}$) in snowmelt runoff from a field where manure was applied onto the snow. American researchers have found similarly large nutrient losses. For example, in the state of New York, Klausner et al. (1976) found that solid dairy manure application during thawing periods resulted in significant P loss to surface water bodies. In Minnesota, Young and Mutchler (1976) found that up to 16% of orthophosphate was lost during spring runoff when manure was applied to frozen soil. In the same study, losses of less than 4% of P were observed when the manures were incorporated following application into the
soil in the fall. In summary, applying manure outside the winter period, when it can be
incorporated or injected underneath the soil surface, markedly reduces nutrient losses in runoff
relative to surface applications (Daverede et al. 2004; Tarkalson and Mikkelsen 2004; Little et al.
2005). Winter application of fertilizer or manure is prohibited in Manitoba and weather-related
regulations for manure application exist in South Dakota and Iowa. However, voluntary
guidelines regarding winter spreading of manure remain the norm in the region (Liu et al. 2018).
While synthetic fertilizer is rarely applied during the winter, mainly due to economic and
agronomic concerns about losses in runoff, in other areas of the Northern Great Plains livestock
manure is sometimes applied during winter because some livestock operations do not have
sufficient capacity for storing manure. This suggests there is potential for reduced nutrient
export with increased manure storage, and with regulation.

8.2 Benefits of rotational P-based application of manure

Changing the practice of basing manure applications on N requirements could also
decrease P export. N-based application of manure often leads to elevated soil test phosphorus
(e.g., Eghball and Power 1999; Miller et al. 2011), hence a shift to P-based application can have
rapid benefits in reducing P runoff. However, this can be a costly change for feedlot operators
(Lory et al. 2004; Miller et al. 2011). Rotating fields where N-based manure application rates
are applied may limit P build-up (Karimi et al. 2018) and risk of P export. For example, in
southern Alberta, using P-based manure application every 1-3 years led to 50-94% reductions in
P runoff, with triennial P-based application showing similar benefit to the annual implementation
of P-based manure application (Miller et al. 2011). This may represent a more cost-effective
option than transitioning to P-based manure application in all years.
8.3 Critical source area management

While the spring freshet is a key period for nutrient transport, as illustrated by the high proportion of nutrients transported during snowmelt (Table 1), the concept of hotspots of nutrient transport also merits attention. These areas are often referred to as critical source areas, where high nutrients, and high risk of transport of those nutrients (or hydrologic connectivity) co-occur in space. Targeted work to mitigate the effects of critical source areas can yield rapid benefits at some spatial scales, in some regions (Sharpley et al. 2011), hence critical source areas may merit special consideration in the design of programs to manage agricultural runoff (Palliser Environmental Services Ltd and Alberta Agriculture and Rural Development 2008; Sharpley et al. 2011). Given the very large range in hydrologic connectivity in the region, and evidence of high spatial variability in soil phosphorus in some areas (Wilson et al. 2016), disproportionate benefit could result if critical source areas are targeted via efforts to manage excess nutrients or efforts to limit hydrologic connectivity (e.g., gating ditches, reservoir construction). However, as in other regions, instream buffering, and legacy phosphorus can mean that benefits achieved at smaller scales (e.g., edge of field) are not observed at the watershed scale (Sharpley et al. 2011). In addition, existing tools for identifying critical source areas may require revision for snowmelt-dominated regions and regions where dissolved nutrients dominate loads. Overall, it appears that additional efforts to foster sound nutrient management practices in areas of high hydrologic connectivity may be strategically beneficial. While the idea of critical source areas has been under discussion for decades (e.g., Pionke et al. 2000) in other regions, there are currently (as of July 25, 2018) no peer-reviewed papers catalogued in Web of Science referring to critical source areas within the Northern Great Plains or the constituent states and provinces, suggesting that the concept may not have been fully explored in science, policy, or management of this region.

Substantive efforts are currently underway to mitigate elevated nutrient loads, yet considerable gaps in our understanding remain. Here we highlight strategic future research to help inform management, and provide a suite of current management recommendations (see also summary in Table 5).

9.1 Tillage practices

The effects of tillage are complex, and no single tillage option will resolve downstream nutrient loading issues. An understanding of the various systems (Tables 2, 3) can be used to select the most appropriate practice for a particular situation. On sloping lands where erosion and particulate nutrient transport are issues, conservation or zero tillage is a good option. However, on much of the Northern Great Plains, where dissolved nutrient transport is more significant than erosion of particulates, increased P transport due to stratification of P in the surface soil of conservation and zero tillage fields is an important concern. Rotational tillage (intermittent use of tillage to mix the soil) can provide a solution but it should be recognized that the tillage will eliminate any infiltration benefits of zero tillage, and fall tillage will increase nitrate and particulate nutrient transport during snowmelt (Liu et al. 2014a). Rotational tillage that utilizes a spring tillage pass may be an option to reduce P transport with less impact on nitrate and particulates in snowmelt. Research into nutrient placement for conservation tillage may also provide an alternative management to reduce nutrients available for transport. The recent decline in summerfallow acreage has been positive for surface water quality and where drier conditions deem it necessary flexible rotations should be employed so that producers can seed a crop when soil moisture reserves are thought sufficient to produce an economic yield (Zentner et al. 1993).
Finally, tillage practices and other practices such as fallow can alter the risk of wind erosion (e.g., Merrill et al. 1999). Wind-eroded particles can be significant contributors to nutrient loading via atmospheric deposition. For example, within the large, shallow Lake Winnipeg, atmospheric deposition of P is estimated to represent 7% of P sources (TN deposition is 11% of loads; Environment Canada and Manitoba Water Stewardship, June 2011). Given atmospheric deposition estimates can be highly uncertain (Ahn and James 2001) and may result from both localized sources and long-distance transport, future work aimed at understanding sources and magnitude of atmospheric deposition will be useful in building a more complete understanding of transport pathways, and options to reduce nutrient loading.

9.2 Perennial forage, grasslands, and grazing

The net effect of perennial vegetation on water quality relative to annual cropping will depend on the relative contributions of snowmelt and rainfall-generated runoff. Perennial vegetation (especially long-term cover) will likely reduce nutrient transport during rainfall runoff events but during snowmelt, dissolved nutrient transport will likely increase. Forages and grasses have been shown to increase nutrient contributions to snowmelt in field and lab studies but the effects of stand composition, age and management require further investigation. If snowmelt infiltration improves in long-term forage stands, the resulting reduction in runoff volume should be expected to reduce nutrient loads. Further research is required to assess the net effect of declining nutrient source and increasing snowmelt infiltration that occur as a vegetation stand ages. Sustainable stocking rates, rotational grazing and relocation of livestock away from streambank areas are the major management practices that can be used to reduce the effect of grazing livestock on nutrient loading (Olson et al. 2011). Due to concerns regarding elevated
loads of dissolved and particulate nutrients, in-field winter feeding should be confined to areas not expected to contribute to flow the following spring.

Finally, tillage practices and other practices such as fallow can alter the risk of wind erosion (e.g., Merrill et al. 1999). Wind-eroded particles can be significant contributors to nutrient loading via atmospheric deposition. For example, within the large, shallow Lake Winnipeg, atmospheric deposition of P is estimated to represent 7% of P sources (TN deposition is 11% of loads; Environment Canada and Manitoba Water Stewardship, June 2011). Given atmospheric deposition estimates can be highly uncertain (Ahn and James 2001), future work aimed at understanding the magnitude of this source will be useful in building a more complete understanding of transport pathways, and options to reduce nutrient loading.

9.3 Surface drainage

Although surface drainage is extremely widespread in the Northern Great Plains, and has been practiced for more than a century (Palliser and Spry 1968), its hydrological and biogeochemical effects remain poorly characterized. Options to help mitigate nutrient export from drained land, and the hydrological effects of drainage are needed. Controlled drainage, channel design, and channel maintenance may yield some benefits – but require study within the region. Fundamentally, the first step is to investigate the effects of drainage on nutrient export under a wide range of weather conditions to fully characterize the response (Spence et al. 2018). There is also the need to better understand differences in drainage practices – from infilling wetlands, to draining different classes of wetlands, and different scales of drainage projects – with the goal of understanding how these practices differ in their impacts on both hydrology, and hydrochemistry of catchments.
9.4 Subsurface drainage

The spring hydrology of tile drains throughout cold regions and controls on P export have received very little study (King et al. 2015), but given the importance of snowmelt to hydrology and nutrient export in this region, this time of the year is critical. In addition, to understand how expansion of tile drainage affects exports of both N and P, we need to compare exports from tile-drained fields to past practices, such as surface drainage. A shift from surface drainage to tile drainage will likely lead to greater N loss, and less P loss – affecting the stoichiometry of nutrient export, with potential implications to downstream ecosystems – an area of considerable importance to management (Baulch 2013). Finally, work among different soil types, crops/forages, tillage practices, and nutrient management practices is required to understand and anticipate changes as this practice becomes more widespread. Longer-term studies are necessary to understand whether effects associated with new drains persist through time, and how drains function through both wet and dry climate cycles.

Expansion of tile drainage practices brings with it the need to understand management options to mitigate nutrient export. Management practices to minimize the effect of preferential flow on nutrient export (e.g., tillage) is a pressing area for work to help mitigate nutrient effects of drainage. In addition, effective nutrient management, and options to trap nutrients (Miller et al. 2012) from tile-drained land deserve further investigation as agricultural practices and policies co-develop. Controlled drainage, bioreactors and physico-chemical amendments are promising – but require study, and modification to suit this cold region where snowmelt dominates nutrient export, and may decrease residence time within these treatments.

9.5 Vegetated Riparian Buffers, Channels, and Filter Strips
Buffers can be highly effective in reducing erosion and transport of particulate-bound nutrients and will in many cases retain dissolved nutrients from runoff, but over time retention also leads to increases in the concentration of P available for release to runoff. Without reductions in the rate of loading from upstream sources, riparian buffers that effectively retain P are likely to become saturated and shift to a net source of P to downstream aquatic systems. The potential for transformation of N to a gaseous form via denitrification exists, but uptake and transformation rates are typically lowest with snowmelt and net release of N to snowmelt is commonly observed during snowmelt. Research to design buffers that will effectively reduce nutrient transport from agricultural fields to surface water on the Northern Great Plains must consider the key factors that influence the potential for uptake and release, including frozen soils and vegetation, concentrated flows, flooding, and dissolved nutrient transport.

The relatively small surface area that comes into contact with runoff through concentrated flowpaths (as compared to more dispersed sheet flow) is likely to lead to more rapid saturation of P retention capacity of soils that contact runoff water. For this reason fixed width buffers may not be the most effective way to disperse and promote nutrient removal from concentrated flows travelling to the edge of field in preferential flow paths. Variable width designs that fit the landscape may be more successful. Additionally, there are indications that removal of biomass from riparian areas may extend P retention effectiveness over time where rates of removal result in drawdown of leachable soil P, but the timing and method of removal requires further investigation. Mechanical harvesting is possible in larger, regularly shaped buffers but removal by grazing may be more practical in uneven terrain.

For riparian buffers to effectively prevent N from entering surface waters those factors controlling both uptake and transformation must be considered. Rates of microbial and plant uptake are likely to be highest during warmer months where rainfall runoff occurs.
Denitrification rates may also be higher where temperatures are higher and soils are not frozen. For this reason, timing of nitrogen application (fall vs. spring) may be an important consideration in defining the potential for downstream riparian buffers to intercept any losses with runoff.

9.6 Small reservoirs, holding ponds and wetlands

9.6.1 Small reservoirs

Initial implementation of headwater dams as a BMP was predicated on hydrological benefits, not nutrient benefits. Although a reduction in peak flows has been demonstrated locally in small subcatchments (Tiessen et al. 2011), the limited water storage capacity of reservoirs at the scale of the larger watershed means reductions in peak flows at the watershed outlet may be small (Liu et al. 2014c). This highlights key trade-offs in the design of these BMPs. Deep reservoirs have greater storage capacity per unit area, and the potential for higher nutrient retention via sedimentation. However, their more limited sediment-water interactions due to their greater depths can limit denitrification and sediment P sorption/desorption reactions. Deep reservoirs have differing vulnerability for P release as a result of pH change and anoxia. This is because they have a deeper water column that attenuates the impact of sediment respiration – the major process driving anoxia risk, which also influences pH change.

Differences in reservoir design lead to different costs in terms of the area of land lost to cultivation per unit volume of water stored. For example, deep reservoirs may be more space-efficient, but could be more costly to build. Construction costs at the time of building of the Tobacco Creek dams were estimated between $15,000 and $26,000 for a ~50 year lifespan (AAFC 2012). Work to date suggests low rates of infilling; hence sedimentation is unlikely to be a constraint on reservoir lifespans, although concerns regarding management of legacy dams in many regions, means questions of infrastructure sustainability require consideration.

9.6.2 Feedlot runoff holding ponds

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Given the extreme inter-annual variability in runoff observed in the Northern Great Plains, compounded by questions regarding changing hydrology and more efficient rainfall runoff generation in some areas (Dumanski et al. 2015), there are key questions regarding the appropriate design capacity for holding ponds in light of climatic and hydrological change. Another key area for research is effective utilization of the water collected in the ponds to prevent nutrients being released back into surface waters when the ponds are emptied. Treatment systems and effluent irrigation are both options but add to the costs incurred by the livestock producer.

9.6.3 Wetlands

As noted previously, a key priority is quantifying the impact of drainage on hydrology and nutrient transport. A key forward-looking goal is understanding the magnitude and controls driving variable nutrient retention across different wetlands. For example, greater protection of wetlands with high nutrient removal capacity could be advocated with research supporting identification of the hydrological attributes, and soils associated with high nutrient retention. Siting decisions for constructed wetlands would likewise be informed by understanding regional differences in the capacity of wetlands soils and sediments to retain phosphorus. Characteristics such as wetland surface area, water residence time, as well as wetland age have been found to affect nutrient retention dynamics in constructed wetlands (Koskiaho et al. 2003; Braskerud et al. 2005), suggesting similar insights for natural wetlands could be used to prioritize for conservation measures.

Within cold regions, nutrient retention can be more effective in summer, reflective both of greater biological activity, but also changed flows, with potentially shorter residence times in winter where ice significantly reduces the water volume but continued inflows occur (White and Bayley 2001). This suggests that design and restoration/modification of wetlands for nutrient
retention in cold climates may need to account for seasonal conditions (Jenssen et al. 1993). Like reservoirs, the infilling rate in wetlands may be important, as can changes in retention capacity over time. Additional research is required to better understand the potential for nutrient mobilization in restored wetlands (e.g., Aldous et al. 2007; Ardón et al. 2010) and assess how soils can be managed or amendments applied to avoid or minimize these effects (Koskiaho et al. 2003; Ballantine and Tanner 2010). Removal of nutrients from wetlands through harvesting of biomass for fuel production (Berry et al. 2017) is also a promising avenue for further investigation.

**9.7 Nutrient management practices**

Balancing agronomic productivity and the need to protect water quality is a key challenge. In the Northern Great Plains landscape, managing the source of nutrients may be even more fundamental than in other regions, due to challenges of trapping dissolved nutrients during snowmelt. As a result, ensuring sound nutrient management, particularly in areas of high hydrologic connectivity where nutrient surpluses could result in hotspots of nutrient transport, is crucial. Initiatives such as preventing late fall and winter spreading of manure, and implementing P-based manure application on a rotational basis can yield significant benefits, with relatively low costs, and little or no impact on agricultural productivity. Likewise, managing livestock density, and ensuring livestock are managed with respect to sensitive areas, and sensitive hydrologic periods will yield benefits. However, more work is required to identify best practices for managing nutrients in reduced tillage systems and tile drained systems.

**10 The Challenge Ahead**
Some of the conditions that make the Northern Great Plains an area of high agricultural productivity also make this region vulnerable to degraded water quality. Although point sources have contributed to nutrient issues in some areas (Leavitt et al. 2006), agricultural activity is an important source of nutrients in the landscape. It is now more than 4 decades since the publication of pivotal research aimed at understanding the drivers of eutrophication (Schindler 1974). Despite this long-standing understanding of the need to control P inputs to surface waters, we see continued degradation of water quality in many regions, and few successes in controlling nutrients at the landscape scale in the Northern Great Plains.

One barrier to progress has been our inadequate understanding of place-based solutions. Solutions must be considered through lenses of regional hydrology, soils, agricultural practice, and the sensitivity of lakes and rivers. Trade-offs exist at every level; between nutrient concentrations and loads (e.g., dilution effects of practices that increase runoff volume), between dissolved and particulate nutrients (e.g., minimizing tillage and establishing perennials), between N and P (e.g., rotational fall tillage), between surface and ground waters, between water quality and other environmental attributes (e.g., habitat and greenhouse gas emissions), and between environmental and agronomic/economic goals. The latter three are beyond the scope of this review but our descriptions of the soil and water managements BMPs address surface water quality trade-offs and indicate the need to match the practice to the place and water quality concern. On much of the Northern Great Plains where snowmelt runoff exceeds rainfall-generated runoff and erosion is often relatively insignificant, practices to control particulate transport that minimize tillage and establish perennial vegetation can concentrate nutrient sources at the soil surface and increase dissolved nutrient transport. Drainage provides an example of a trade-off between N and P loss to surface water with surface drainage favouring P
loss while tile drains can provide a pathway for N loss to surface water. Rotational fall tillage may also decrease P in runoff water while increasing N transport. Therefore, every individual situation requires identification of the key environmental risks, then careful, informed selection of locally relevant soil and water management practices that maximize benefits and minimize trade-offs, using the ‘right strategy, right place’ principle (Dodd and Sharpley 2016).

Understanding the changing climate to ‘future-proof’ management options is also crucial (Crossman et al. 2013), for example, considering how a shift from snowmelt-dominated runoff to greater rainfall runoff may affect the efficacy of different practices for nutrient control.

Currently, the prognosis for surface water quality in the region is poor. The many shallow lakes in the region have periodically high hydrological and biogeochemical connectivity to their landscapes. And they have large watersheds, with effective drainage areas continuing to increase as agricultural drainage continues. Legacy nutrients in some parts of the landscape (e.g., associated with long-term application of manure in excess of crop demand), combined with structural changes in aquatic ecosystems and nutrient buildup in sediments, mean that there will be lags in ecosystem recovery even once effective practices are implemented. Rivers of the region are also highly degraded and climate change is increasing the sensitivity of surface waters to nutrients via increasing water temperatures affecting cyanobacterial bloom risk (Paerl and Huisman 2008).

Despite current and future challenges, there is growing evidence that short-term improvement in water quality can be achieved. One promising example in the region is the successes resulting from cattle management BMPs and improved nutrient management (Alberta Agriculture and Forestry 2014). We highlight six outstanding research and management needs to help facilitate more work to improved water quality in the Northern Great Plains. Additional science and management needs were noted in Section 9 and Table 5:
Tillage practices

1. Rapid adoption of reduced tillage practices demonstrate that large scale, rapid changes in practice are possible, but also highlight the need for careful consideration of the varied environmental costs and benefits of changing practice. *We need to identify options to mitigate elevated nutrient export associated with reduced tillage.* For example research should assess potential benefits of rotational tillage with a spring tillage pass and of altered nutrient management practices in reduced tillage systems.

Perennial forage, grasslands and grazing

2. *The net effects of conversion of annual cropland to forage should be considered in the context of specific landscapes and fields.* This is due to trade-offs between the benefit of erosion protection afforded by forages, the release of soluble nutrients from vegetative material, and changes in soil nutrient pools through time.

3. *For cattle farms, in-field winter feeding can lead to hotspots of nutrient export, and should be isolated to hydrologically inactive areas.*

Surface drainage, tile drainage and ‘keeping water on the land’

4. *Drainage in the region has been extensive, yet the effect of drainage on nutrient export has received little study.* There is the urgent need to quantify the importance of drainage to nutrient exports, and potential mitigation options. Specifically, research is required to understand how surface drainage, and tile drainage affect nutrient transport as compared to previously undrained fields, and strategic options to maintain water and nutrient retention capacity in the landscape. BMPs to ‘keep water on the land’ (Kanu 2013; McEwan 2013) can help in nutrient control, maintaining or replacing water storage that had been lost. Siting of these BMPs, which include wetlands and reservoirs, may be optimized by better understanding the importance of substrate chemistry to P retention.
Surface drainage, and options to control drainage waters, and modify ditch design require more study. Tile drains in the region shows very high nutrient export, yet it is not yet clear whether this is an installation effect, or reflects longer-term risk.

**Vegetated riparian buffers, channels, and filter strips**

5. *Vegetation management in riparian areas, buffer strips, and erodible channels is unlikely to remain effective for P removal over time if high rates of loading exist, but may be more effective for N removal, and during rainfall events.* When compared to cultivating to the river’s edge (and through floodplains) ecological benefits are likely and potential benefits for nutrient management remain important to consider.

**Nutrient management practices**

6. *Improvements to nutrient management practices are required.* Similar to other regions of the world, the correct rate, source and placement of manure and fertilizer is important. In the Northern Great Plains, timing nutrient applications to avoid or minimize the risk of losing nutrients during snowmelt is particularly important. Specifically, minimizing or eliminating nutrient application in late fall or during winter is important given the high risk of direct transfer of nutrients to snowmelt and the limitations of many traditional practices for intercepting nutrients in snowmelt-dominated landscapes. Periodic high summer runoff and anticipated future shifts in climate emphasize the importance of robust nutrient management across seasons, and evidence of nutrient accumulation in some areas of high livestock density suggest that livestock and manure management may merit further attention. Research is required to understand how nutrient management can be adapted for specific practices, such as reduced tillage, forage, and tile drainage to help reduce nutrient runoff. Exploring P-based manure application, at least on a rotational basis, is merited. Finally, targeting additional nutrient management efforts in existing
critical source areas, and other areas of high hydrological connectivity may yield disproportionate benefits.

Across this region, where water quality degradation is so common, yet agriculture is such an important part of the economy and an important contributor to global food security, a wholesale reassessment of the prioritization of BMPs is in order. Agricultural inputs are measured in the kg per hectare, yet lakes respond to nutrient inputs at microgram per liter levels – which has contributed to the ‘grand challenge’ of solving N (National Academy of Engineering and Engineering 2008) and P issues. This challenge becomes even greater when pressures to increase food production to meet growing demand are considered, and other environmental and agronomic costs, and benefits are factored in, such as managing soil carbon, greenhouse gas emissions, and wildlife habitat. Within the Northern Great Plains, there are unique challenges and opportunities associated with mitigating agricultural nutrient loading. In light of the relative water scarcity in the region, protecting water quality remains an important priority, hence place-based options for nutrient mitigation are required, along with a targeted approach to their implementation.
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Williams and Wollman 1984


Table 1: Proportion of nutrient export from dissolved species during snowmelt within areas of the Northern Great Plains

<table>
<thead>
<tr>
<th>Location</th>
<th>Phosphorus</th>
<th>Nitrogen</th>
<th>Citation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Red River Valley, Manitoba, Canada</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4 catchments, southwestern Manitoba.</td>
<td>82 to 87</td>
<td>94 to 96</td>
<td>Corriveau et al. 2011.</td>
</tr>
<tr>
<td>Madill subwatershed, South Tobacco Creek</td>
<td>73</td>
<td>86</td>
<td>Li et al. 2011</td>
</tr>
<tr>
<td>Steppler subwatershed, South Tobacco Creek</td>
<td>90</td>
<td>87</td>
<td>Li et al. 2011</td>
</tr>
<tr>
<td>Paired subwatersheds, South Tobacco Creek</td>
<td>55 to 94</td>
<td>65 to 94</td>
<td>Tiessen et al. 2010</td>
</tr>
<tr>
<td>Alberta, Canada</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Subbasins of Crowfoot Creek subwatershed</td>
<td>65 to 93</td>
<td></td>
<td>Ontkean et al. 2005</td>
</tr>
<tr>
<td>Field-scale microwatersheds</td>
<td>5 to &gt;99</td>
<td>Little et al. 2006</td>
<td></td>
</tr>
<tr>
<td>-----------------------------</td>
<td>---------</td>
<td>-------------------</td>
<td></td>
</tr>
<tr>
<td>Minnesota, USA</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Small runoff plots</td>
<td>75%</td>
<td>Hansen et al. 2000</td>
<td></td>
</tr>
</tbody>
</table>

Table 2. Summary of different tillage practices employed on the Northern Great Plains (see section 2 for detailed discussion).

<table>
<thead>
<tr>
<th>Practice</th>
<th>Definition</th>
<th>Surface Residue Remaining‡,†</th>
<th>Scope</th>
</tr>
</thead>
<tbody>
<tr>
<td>Zero Tillage</td>
<td>Direct seeding into undisturbed stubble or sod using low disturbance implement</td>
<td>80-90% residue cover remains after seeding.</td>
<td>Small proportion of producers practice true zero tillage using a low disturbance implement but many direct seed.</td>
</tr>
<tr>
<td>Conservation Tillage</td>
<td>Tillage that retains most of the crop residue on the soil surface – may include direct seeding with a high disturbance implement</td>
<td>50-80% residue cover remains after seeding.</td>
<td>Most common practice in general. More popular in drier parts of the region.</td>
</tr>
<tr>
<td>Rotational Tillage</td>
<td>Fall tillage pass practiced once every 2 years to mix residues into soil in a direct seeding system</td>
<td>60-80% residue cover going into winter of the tillage year, 100% cover going into winter in the other year.</td>
<td>Introduced to counter stratification under zero and conservation tillage without reverting to a conventional tillage system.</td>
</tr>
<tr>
<td>Conventional Tillage</td>
<td>Most of the crop residues are incorporated into the soil through tillage, seeding and planting</td>
<td>20-50% residue cover remains after seeding.</td>
<td>Varies throughout region, usually more intensive in wetter areas where it usually includes at least one fall.</td>
</tr>
<tr>
<td>Method</td>
<td>Description</td>
<td>Residue Cover</td>
<td>Notes</td>
</tr>
<tr>
<td>------------------------</td>
<td>-----------------------------------------------------------------------------</td>
<td>---------------</td>
<td>------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Fall Tillage</td>
<td>Part of conventional tillage in wetter areas – soil is tilled and standing stubble knocked down prior to winter</td>
<td>60-80%</td>
<td>Less common in drier areas where standing stubble is retained to trap snow.</td>
</tr>
<tr>
<td>Chemical Fallow</td>
<td>No crop is seeded to conserve soil moisture and vegetation control on the undisturbed soil is by herbicides. Several applications are required during the growing season.</td>
<td>80-100%</td>
<td>Used in combination with zero tillage or on conservation till land. Mainly in south west Saskatchewan and southern Alberta.</td>
</tr>
<tr>
<td>Conventional Fallow</td>
<td>No crop is seeded to conserve soil moisture and vegetation control on the undisturbed soil is by tillage. Several passes are required during the growing season.</td>
<td>&lt;30%</td>
<td>Used in combination with conventional tillage. Mainly in south west Saskatchewan and southern Alberta.</td>
</tr>
<tr>
<td>Subsoiling or Deep Ripping</td>
<td>Soil is tilled below the normal tillage depth using specialized implement with residue can be relatively undisturbed.</td>
<td>Depends on implement used</td>
<td>Used mainly on compacted soils especially Solonetzic.</td>
</tr>
</tbody>
</table>
The amount of residue on the surface varies greatly by crop.

Reduction in residue cover based on number and type of field operations (Renard et al. 1997).

More residues remain if herbicides are used in combination with tillage for weed control.
Table 3. Summary of the impacts of tillage on hydrology and nutrient export (see section 2 for detailed discussion)

<table>
<thead>
<tr>
<th>Tillage Effect</th>
<th>Tillage Intensity to Produce Effect</th>
<th>Influence during Snowmelt</th>
<th>Influence during Rainfall</th>
</tr>
</thead>
<tbody>
<tr>
<td>Standing stubble flattened</td>
<td>Single pass is sufficient to cause an effect</td>
<td>Snow trapping and runoff volume reduced</td>
<td></td>
</tr>
<tr>
<td>Cracks are created</td>
<td>Deep tillage implement needed. May have to be repeated in 3 to 5 years.</td>
<td>Infiltration improved in compacted soils.</td>
<td>Infiltration improved in compacted soils.</td>
</tr>
<tr>
<td>Continuous pores are disrupted</td>
<td>Single pass may impact pore continuity. Several years of low disturbance can be required to create continuous pores.</td>
<td>Infiltration may be improved under long-term zero tillage.</td>
<td>Small events may infiltrate more easily into recently tilled soil.</td>
</tr>
<tr>
<td>Aggregates are disrupted</td>
<td>Single tillage pass can release C and N</td>
<td>Fall tillage pass increases nitrate concentrations during snowmelt</td>
<td>Nitrate released is taken up by plants or leached below soil-runoff interaction zone by first precipitation</td>
</tr>
<tr>
<td>Residues are mixed through tillage depth</td>
<td>Mixing increases with number of passes and intensity of implement</td>
<td>Stratification of P at soil surface and loss of dissolved P in runoff are reduced</td>
<td>Stratification of P at soil surface and loss of dissolved P in runoff are reduced. Surface protection is removed leading to increased particulate losses during high intensity events.</td>
</tr>
<tr>
<td>----------------------------------------</td>
<td>---------------------------------------------------------------</td>
<td>---------------------------------------------------------------------------------</td>
<td>----------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Surface roughness increases</td>
<td>Roughness depends on tillage implement used.</td>
<td>Runoff may be slowed allowing for greater infiltration.</td>
<td>Runoff may be slowed allowing for greater infiltration.</td>
</tr>
<tr>
<td>Tillage erosion gradually moves surface soil into areas of concentrated flow</td>
<td>Some movement occurs with every pass but distance depends on degree of disturbance</td>
<td>More nutrients are available for transport but interaction with runoff water is low during snowmelt</td>
<td>More nutrients are available for transport and loss in runoff can increase.</td>
</tr>
</tbody>
</table>
Table 4. Categories of grassland and management considerations for managing hydrology, erosion, and dissolved nutrients (see section 3 for detailed discussion)

<table>
<thead>
<tr>
<th>Management practice</th>
<th>Hydrology</th>
<th>Erosion &amp; Particulates</th>
<th>Dissolved Nutrients</th>
</tr>
</thead>
<tbody>
<tr>
<td>Perennial Forage in Rotation</td>
<td><strong>Snowmelt:</strong> Height of last cut may impact snow-trapping and redistribution. Infiltration rate minimally affected.</td>
<td><strong>Snowmelt:</strong> Forage protects the soil surface from erosion, but snowmelt is generally non-erosive.</td>
<td><strong>Snowmelt:</strong> Vegetation is a nutrient source, but can be reduced by late fall harvesting as hay. Lack of disturbance leads to nutrient stratification at surface. Nitrogen fertilizer input may be reduced where legumes are cultivated.</td>
</tr>
<tr>
<td><strong>Rainfall:</strong></td>
<td>Infiltration may be improved by surface protection, root channels &amp; water use.</td>
<td><strong>Rainfall:</strong> Surface protected from spring rains.</td>
<td><strong>Rainfall:</strong> Nutrient cycling impacted by changes in N fertilizer input and N-fixation in legumes.</td>
</tr>
<tr>
<td>Permanent Cover</td>
<td><strong>Snowmelt:</strong> Height of last cut may impact</td>
<td><strong>Snowmelt:</strong> Forage protects the soil surface from erosion, but</td>
<td><strong>Snowmelt:</strong> Vegetation is a nutrient source but can be reduced by</td>
</tr>
</tbody>
</table>

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<table>
<thead>
<tr>
<th><strong>snow-melt</strong></th>
<th><strong>snow-trapping.</strong></th>
<th><strong>late fall harvesting as hay.</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Infiltration may be improved due to soil structural improvement, water use and cracking.</td>
<td>snowmelt is generally non-erosive.</td>
<td>Soil and vegetation nutrients may be depleted as stand ages.</td>
</tr>
<tr>
<td>Nutrients stratify at soil surface.</td>
<td></td>
<td>Nutrients stratify at soil surface.</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Rainfall:</strong></th>
<th><strong>Rainfall:</strong></th>
<th><strong>Rainfall:</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Infiltration may be improved by surface protection, soil structural improvement, root channels &amp; water use.</td>
<td>Surface protected from spring rains. Improved soil structural stability reduces erosion.</td>
<td>Nutrient cycling impacted by N-fixation in legumes.</td>
</tr>
<tr>
<td>Nutrient balance easier to achieve for grazed lands?</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Grazing-land</strong></th>
<th><strong>Winter:</strong></th>
<th><strong>Winter:</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Feed and feces are source of particulate nutrients.</td>
<td>Concentrated nutrient source transported in runoff.</td>
<td>Timing of grazing relative to snowpack accumulation requires consideration.</td>
</tr>
<tr>
<td>Soil nutrients increase in grazed area if feed is</td>
<td>Maximize feed utilization to limit source.</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Soil nutrients increase in grazed area if feed is</td>
</tr>
<tr>
<td>Summer:</td>
<td>Summer:</td>
<td>Summer:</td>
</tr>
<tr>
<td>------------------------------------</td>
<td>----------------------------------------------</td>
<td>----------------------------------------------</td>
</tr>
<tr>
<td>Overgrazing may reduce infiltration, so stocking rate is important</td>
<td>Fencing cattle out of water courses reduces in-stream deposition &amp; streambank erosion. Overgrazing may damage soil structure and increase erosivity</td>
<td>Stocking rate impacts digested to undigested nutrient source ratio. Rotational grazing can better utilize pastures and keep livestock away from sensitive areas at critical times.</td>
</tr>
</tbody>
</table>
Table 5: Summary of land use practices with the potential to impact nutrient export, outstanding research needs, and recommendations for practice

<table>
<thead>
<tr>
<th>Practices</th>
<th>Key impacts of practice in the Northern Great Plains</th>
<th>Recommendations for research</th>
<th>...and practice.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduced Tillage</td>
<td>Reduced tillage controls erosion but increases export of dissolved P.</td>
<td>Can rotational tillage reduce nutrient export as compared to zero tillage and conservation tillage? How can altered nutrient management practices reduce nutrient export in low tillage systems?</td>
<td>Other environmental and agronomic benefits support continued low till practices, but with need to address issues of elevated nutrient export.</td>
</tr>
<tr>
<td>Winter bale grazing</td>
<td>Can lead to hotspots of nutrient export.</td>
<td>Restrict to hydrologically inactive (non-contributing) areas.</td>
<td></td>
</tr>
<tr>
<td>Forages and pastures</td>
<td>Conversion of annual cropland to forage generally reduces erosion but can initially increase supply of dissolved nutrients to runoff water.</td>
<td>How do hydrology and biogeochemistry impact nutrient transport from a range of different forage and pasture types? Do infiltration rates improve under perennial vegetation relative to annual cropping? How can increased dissolved nutrient release from forages during snowmelt be mitigated?</td>
<td>Ensure appropriate stocking density, Restrict livestock access to streams via fencing or provide alternative water supplies. Rotational grazing to move livestock from sensitive areas at critical periods.</td>
</tr>
<tr>
<td>Tile drainage</td>
<td>High export possible immediately after installation. (long-term effects not known)</td>
<td>Will high export from new drains be sustained in the long term? What are the controls on P loss in</td>
<td>Caution in approving new drainage programs given potential for high export (more research urgently required, given agronomic and climate adaptation.</td>
</tr>
<tr>
<td>Method</td>
<td>Notes</td>
<td></td>
<td></td>
</tr>
<tr>
<td>-------------------------------</td>
<td>----------------------------------------------------------------------</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ditches and surface drainage</td>
<td>Largely unstudied. Evidence of increased nutrient export associated with surface drainage in wet periods.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Reservoirs</td>
<td>Can lead to nutrient retention.</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

- Can controlled drainage, bioreactors or P sorbing materials mitigate export?
- How do tile drainage exports compare to surface drainage and no drainage scenarios and will changing flowpaths impact the N:P ration of nutrients exported?
- Can preferential flow be minimized?
- What changes to nutrient management are required in tile-drained fields?
- Can ditch design be optimized for nutrient removal despite episodic flows?
- Can controlled drainage lead to retention of dissolved nutrients?
- How can vegetation in ditches best be managed to maximize nutrient retention?
- Establish as water resource, flood/drought mitigation tool, and in nutrient retention.
<table>
<thead>
<tr>
<th></th>
<th>Divert concentrated runoff from stream systems</th>
<th>What can be done to economically optimize the use of water collected in holding ponds?</th>
<th>Careful management of holding ponds as a key point to reduce nutrient export from rural point sources.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Holding Ponds</td>
<td>Impacts on nutrient retention not known.</td>
<td>Do prairie wetlands retain nutrients or just transform them?</td>
<td>Protect and restore existing wetlands due to potential for hydrological benefits and other ecosystem services.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>How permanent is nutrient removal?</td>
<td>Targeted use of constructed wetlands to support nutrient mitigation with careful consideration of siting options.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>How many wetlands is enough?</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>How does wetland class related to effects of drainage and restoration on nutrient export?</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>What design options are best suited for constructed wetlands in this landscape?</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Can constructed wetlands, holding pond design, and feedlots be optimized to minimize nutrient runoff in this low gradient, snowmelt-dominated landscape?</td>
<td></td>
</tr>
<tr>
<td>Wetlands</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Riparian vegetation</td>
<td>Unlikely to remain effective for P removal over time if high rates of loading exist, but may be preferable over cultivation to the waters’ edge, more effective for N removal, and during rainfall events.</td>
<td>Can changes in buffer width or vegetation type disperse concentrated flows?</td>
<td>Continue encouraging based on habitat benefits, but do not rely upon for nutrient mitigation where high rates of loading from upland sources remain. Effectiveness for nutrient removal is largely a function of upland management and history of loading. Soil testing to identify degree of phosphorus saturation will aid in the selection of sites where restoration is most likely to result in retention or where mitigation efforts such as</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Can practices such as biomass harvest to remove P from buffer soils increase effectiveness for phosphorus retention over time?</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>What are the mechanisms</td>
<td></td>
</tr>
</tbody>
</table>
controlling seasonality of N uptake rates with flow through buffers and can management of vegetation type increase rates of removal?

| Nutrient management | While our review is focused on soil and water management practices, we view nutrient management as foundational. The timing of nutrient application (avoiding hydrologically sensitive periods, such as late autumn and on snow or frozen soils), and hydrologically sensitive areas (such as large, seasonally inundated areas and areas of high hydrologic connectivity) is critical. Widespread support of soil testing, and more stringent guidelines, particularly in terms of phosphorus, are required. For example, implementation of annual, biennial, or triannual P-based manure application should benefit in areas with hydrologic connectivity. More data about regional nutrient balances (fertilizer and manure application, livestock density) at finer spatial scales will help in understanding implications of decisions, such as approval of new livestock facilities and consequent manure handling. Nutrient management should be viewed as a keystone activity which underlies and interacts with all soil and water management activities discussed here. | biomass harvest may be required. |
Figure 1: The Northern Great Plains region is defined using level 3 ecoregions (including the Lake Agassiz Plain, Northern Glaciated Plain, Northwestern Glaciated Plain, Northwestern Great Plain and Western Corn Belt). The northern boundary is defined by the upper limit of these ecoregions, while the southern boundary is approximated based on the March 0°C isotherm USGS 2011; data are from 1950-2000). March snow water equivalent (SWE) is also plotted, showing north-south variation in March snow accumulation (Armstrong et al. 2005).

Figure 2: Factors affecting nutrient transport in the Northern Great Plains.

Figure 3: Crops grown in northern Great Plain states and provinces in 1981/82 and 2011/12. Data are from whole states rather than the Northern Great Plains portions. Data are from USDA Census of Agriculture and Clearwater et al., 2016.

Figure 4. Effects of tillage and perennial vegetation on nutrient mobilization and transport on the Northern Great Plains.

Figure 5: Summary of efficacy of vegetated riparian buffers, channels, and filter strips; drainage; and reservoirs, wetlands and holding ponds for nutrient control in the Northern Great Plains noting key research needs.

Figure 6: Recommendations for research and implementation of management practices to mitigate nutrient export.
a) Enlarged Northern Great Plains Ecoregion

1) Lake Manitoba & Lake Agassiz Plain
2) Aspen Parkland/N. Glaciated Plains
3) Northwestern Glaciated Plains
4) Northwestern Great Plains
5) Western Corn Belt Plains

March SWE (mm)
- 0–10
- 10–25
- 25–50
- 50–100
- 100–200
- >200
Large proportion of landscape is naturally non-contributing in an average year.

Snowmelt dominates runoff in the relatively flat relief resulting in overal flow risk of water erosion of soils.

Snow accumulation and runoff volume are affected by vegetation/vegetation.

Ponded water allows increased water-soil/residue interactions.

Vegetation is dormant and wildfire beconis key hydrological periods.

Widespread wetland drainage.

Most nutrients transported in dissolved form.
Subsoiling can improve infiltration into cultivated soils

Cultivation breaks soil aggregates releasing nutrients for transport

Tillage can disrupt continuous pores that can develop through time in undisturbed soils

Grazing animals can reduce infiltration and mobilize nutrients

Nutrients are released on freezing of crop residues and vegetation

Vegetation and crop residues protect soil from erosion

Height of vegetation or residue influences snow catch and snow redistribution

Nutrients stratified near the surface of undisturbed soil are available for transport
Summary of efficacy of vegetated riparian buffers, channels, and filter strips; drainage; and reservoirs, wetlands and holding ponds for nutrient control in the Northern Great Plains noting key research needs.
<table>
<thead>
<tr>
<th>PRACTICE</th>
<th>RESEARCH and MANAGEMENT PRIORITIES</th>
</tr>
</thead>
<tbody>
<tr>
<td>Winter Bale Grazing</td>
<td>Management: Only in conjunction with transport control</td>
</tr>
<tr>
<td>Winter Manure Application</td>
<td>Management: policy change to prevent or limit practice</td>
</tr>
<tr>
<td>Conservation Tillage</td>
<td>Research: Management options to control stratification</td>
</tr>
<tr>
<td>Conversion to Perennial Forage</td>
<td>Research: Harvest management to reduce nutrient source</td>
</tr>
<tr>
<td>Riparian Buffer</td>
<td>Research: Strategies for optimal siting and management.</td>
</tr>
<tr>
<td>Drainage</td>
<td>Research: Management options to control outflow</td>
</tr>
<tr>
<td>Small Reservoirs and Surface Water Storage</td>
<td>Management: Locate and manage for nutrient removal</td>
</tr>
<tr>
<td>4Rs – Nutrient Management</td>
<td>Management: Incentives and outreach</td>
</tr>
<tr>
<td>Wetland Restoration</td>
<td>Research and Management: priorities for restoration and policy options for restoration and protection</td>
</tr>
</tbody>
</table>

Recommendations for research and implementation of management practices to mitigate nutrient export.

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