Beneficial Management Practices (BMPs) for Reducing Greenhouse Gas (GHG) Emissions in Prairie Agriculture

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Contents

Summary

The beneficial management practices (BMPs) outlined in this report describe promising practices for mitigating greenhouse gas (GHG) emissions in the agricultural sector of the Canadian Prairies. We describe how the BMP is implemented, its potential for GHG mitigation by reducing GHG emissions or enhancing storage of soil organic carbon (SOC), current and potential adoption, barriers to adoption, co-benefits and trade-offs of implementing the BMP as well as knowledge gaps that exist. We first describe practices in conventional annual cash-crop production systems (sections 1-7). We then describe BMPs relevant for livestock and pasture systems (sections 8-10). Third, we discuss landscape-scale changes in the agricultural sector to enhance soil carbon sequestration, carbon in woody biomass and improving natural systems (sections 11-15) that are not directly part of the cropping system – changes that typically pertain to the marginal lands on the edges of fields, around wetlands or watercourses and incorporating trees within traditionally low-vegetation dominated areas of the farm. We then describe systematic changes (sections 16-18) where several practices can be implemented together to improve either the cropping or livestock systems. Finally, we discuss several BMPs still in the theoretical stage (sections 19-25) that lack information on mitigation potential or adoption across the Prairies, providing a brief description of the practice and the knowledge gaps that exist in the region with the aim to identify potential future research.

Scope of the Study

This report is developed under the assumption that the overarching structure of Canadian agriculture will remain as it is with grain, livestock and production of other commodities being the focus of agricultural lands. Demand for such commodities is assumed to be similar in the future as it is today. While the BMPs covered are not limited to the farm or field boundary (e.g., bioenergy with carbon capture and storage), they do concentrate almost exclusively on land-based management practices farmers can act on. Land-based practices within livestock systems are included (e.g., grazing ruminants, manure application) but the impact of adoption of those BMPs on the GHG emissions from the whole of livestock production are not included. BMPs related to storage and handling losses of agricultural products and other sources of food waste are not covered because they are not land-based BMPs.

It was also assumed that there will be no leakage resulting from the adoption of the BMPs that could reduce overall production. Although some of the BMPs could change the proportion of certain crops or livestock being produced, this study does not account for changes in diet or market shifts, nor does it consider sector-wide policy changes that could alter the production of agricultural goods, such as border carbon tariffs. There is no consideration of policies that might transfer production to or from the Prairies based on regional advantages in GHG intensity. Although a discussion on the role of exports and other underlying topics relating to GHG mitigation from the Canadian food system are important, interesting, and useful, such strategies not in the scope of this report.

Synthesis of Land Management BMPs

There is a large potential to decrease net GHG emissions from Prairie agriculture (i.e., reduction of N_2O, CH_4, CO_2 and increases in soil carbon) while providing more sustainable food and fibre production. All the BMPs outlined in this report could coexist on the landscape. Some BMPs are competitors for land area (e.g., set-aside marginal land versus cropland, organic versus conventional), but none were fundamentally antagonistic within a land use such that the producer is forced to choose between one BMP over another. Therefore, approaches aimed at optimizing the adoption of multiple BMPs across the agricultural landscape are needed.

There are several major themes across the BMPs discussed in the report. Within the first section of the report, BMPs generally have an abundance of information regarding the co-benefits of implementing each management practice. Almost all BMPs provide valuable economic or environmental co-benefits, which is likely to improve the adoption of BMPs in agroecosystems. In contrast, the GHG impact (whether GHG emissions, or total impact considering SOC as well), is not as well documented than other criteria we reviewed. The impact of BMPs on N_2O emissions is usually more of a knowledge gap than the impacts on SOC, with the trade-off, however, that effects on soil carbon must be monitored over decades after BMP implementation while effects on N_2O are within a growing season. This lack of GHG impact from several BMPs in combination with the lack of Prairie or Canada-wide adoption of many practices is a critical barrier to assessing the potential mitigation of BMPs. Given this challenge, GHG mitigation of Prairie-wide agriculture would be greatly benefitted through improved activity data collection of management practice adoption, including the systems they are adopted within. We also found that specificity and consensus within the literature occurred for BMPs on specific land management changes, which did not occur for agricultural systems like regenerative agriculture, or pasture systems. This is interesting because, although there is detailed information about specific BMPs that exist within a system (e.g., cover cropping, nutrient management, no tillage), the impact of systematic approaches is not clear due to their complexity.

As detailed in the BMP Summary Table 37, the potential mitigation of BMPs is based on the potential area of adoption and the emission factor per area. Therefore, the greatest potential for emissions reductions and removals are through BMPs that can be newly adopted over large areas of the Prairies. Based on the summary table, the BMPs with a high GHG mitigation potential (>5 $MtCO_2e$ per year in 2030) include cover cropping, reducing deforestation to agriculture, avoided conversion of grasslands, pasture and haylands, as well as crop residue bioenergy with carbon capture and storage. BMPs with a medium GHG mitigation potential (2-5 $MtCO_2e$ per year in 2030) include intercropping, optimized nutrient management, wetland conservation and restoration as well as crop residue bioenergy systems. Many of the other BMPs outlined may have a GHG mitigation potential are largely adopted already (no tillage, avoidance of residue burning), or are only available to be adopted in a small area (rotational grazing, conservation of shelterbelts).

Generally, increased external support and targeted research is necessary to address the agronomic, technological and knowledge gaps described in this report. In addition, more work is needed to address and overcome the several critical barriers for BMP adoption outlined within the report, many of which include social and agronomic barriers, which cannot be overcome through research alone.

Impacts across BMPs

Generally, most BMPs are compatible with those improving the agricultural landscape and with other BMPs. However, some BMPs reinforce the benefit of another BMP and, in some cases, one BMP has a trade-off with another BMP. These interactions are outlined in Table 1.

Many of the instances of potential BMP reinforcing and trade-offs relate to the impact of BMPs on crop residue. The crop residue retained is beneficial to soil C sequestration and soil building as crop residue that is anchored and retained on the surface reduces erosion risk. However, the retained residue can create conditions that incentivize some residue burning and tillage. Cover crops and reduced tillage are excellent when there is inadequate residue to control erosion but can aggravate management when there is abundant residue. The BMPs of biochar and bioenergy were based on using crop residues as feedstock, therefore have trade-offs with any residue retention BMP and reinforcing the beneficial BMPs for improving low residue retention. Clearly, there needs to be a crop residue BMP strategy that involves the co-application of crop-residue-relevant BMPs to meet GHG reduction goals while being consistent with soil health goals.

Farming systems that integrate livestock with cropping also have several positive and negative impacts. While generally the integrated crop-livestock system benefits many land-based BMPs, there can be trade-off with increased livestock emissions. The BMPs that affect the perennial herbage for cattle feed, rotational grazing and avoided conversion of land producing the perennial herbage to cropland, share this major trade-off if the adoption of those BMPs induces an increase or a smaller reduction in the cattle herd compared with what would have happened without the BMP adoption, The trade-off is due to increased GHG emissions from livestock, primarily as methane, from what would have happened without the BMP adoption. The trade-off severely reduces, and could even negate, the GHG benefits of the BMP. Consequently, implementing these BMPs needs to consider the overall impact on the cattle GHG emissions to assess the net GHG impact.

In addition, the regenerative and organic farming systems are more than any single BMP and, rather, a system of BMPs that are reinforcing. Adoption strategies should be based on the outcome for a system of multiple land-based BMPs (that also consider directly affected livestock emissions) even when the analysis is done for a single BMP.

Table 1: Interactions among the land-based BMPs outlined in the report. Reinforcing refers to BMPs that improve or utilize another BMP. Trade-offs refer to the use of a BMP negatively impacting or making it unavailable for a famer to adopt another practice. General impacts include systems where the BMP is often adopted.

No.	BMP abbrev.	BMP	BMPs reinforced	BMPs with Trade-offs	General impacts
1	RedTill	Reduced Tillage	Biochar, BioNRG (mitigate erosion from residue removal); RedErsn (reduces erosion)	Org/Regen (challenges for reduced tillage in organic systems); MaxRes (residue challenges for reducing tillage)	Org/Regen (Regen ag includes)
2	CoverC	Cover Crops	Biochar, BioNRG (mitigate erosion and C loss from residue removal); RedErsn (reduces erosion); IntLstck (grazing increases economic value of CoverC); Nman (reduces leaching and runoff N losses)	InterC (challenges to have interseeded cover crops in a competitive intercropping system); MaxRes (cover crops can aggravate problems with excessive residue)	Org/Regen (Regen ag includes)
3	InterC	Intercropping	Diverse (diversity); Nman (improves N management); IntPest (pest, disease management)	CoverC (challenges to have intercropping and also have interseeded cover crops)	Org/Regen (Regen ag includes)
4	Legume	Increased legume crops	Diverse (diversity); Nman (improves N management)	Biochar, BioNRG (less value to remove legume residue); IntPest (increased legume pest); MaxRes (legumes can produce less crop residue than cereals or oilseeds)	Org/Regen (legume important to organic ag)
5	RedBurn	Reduced field burning of crop residues	Biochar, BioNRG (alternate residue removal)		Org/Regen (Regen ag includes)
6	Nman	Improved N management	Legume (reduces N additions over rotation); InterC (more efficient N use); CoverC (reduces leaching and runoff N losses)		
7	Biochar	Biochar addition to soil	RedTill(benefit from reduced tillage); RedBurn (less need to burn residue if harvested); MaxRes (more residue to harvest while leaving more root C); Nman? (reduces N_2O emissions);	Org/Regen (Harvesting crop residue for biochar is inconsistent with organic and regenerative agriculture)	

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			OrgAmend (mitigate residue removal soil impacts)		
8	OrgAmend	Increasing Organic Amendments Applied to Agricultural Lands	IntLstck (manure source); Biochar, BioNRG (mitigate impact of crop residue removal) Improved N management		
9	RotGraz	Rotational Grazing	IntLstck (need for pasture)	SavPeren (more efficient pasture decreases area needed for pasture)	Org/Regen (Regen ag includes)
10	RotPeren	Rotation of Annual Crop with Perennial Forages	RedErsn (reduces erosion); IntLstck (forage for livestock); Rebuild (rebuild soil); IntPest (break for annual crop pests)		Org/Regen (Regen ag includes); livestock GHG emissions must be considered
11	Trees	Increase and Manage Trees in Working Agricultural Landscapes	IntLstck (shade for grazing livestock)		Part of regenerative landscape
12	NoDefor	Reduce Deforestation to Agriculture	Wetland (reduce wetland drainage in trees)		Part of regenerative landscape
13	Sbelt	Reduce loss of Woody Biomass in Agriculture (Avoided Conversion of Shelterbelts)	RedErsn (reduces erosion)		Part of regenerative landscape
14	SavPeren	Avoided Conversion of Grassland, Pasture and Hayland	RedErsn (reduces erosion); IntLstck (forage for livestock); Rebuild (rebuild soil)	RotGraz (reduces area needed for pasture); RotPeren (reduce need for long-term perennials)	Part of regenerative landscape; livestock GHG emissions must be considered
15	Wetland	Wetland Conservation and Restoration	IntLstck (water, pasture and forage in drought)		Part of regenerative landscape; wetland emissions must be considered
16	BioNRG	Crop Residue Bioenergy and Carbon Capture and Storage	RedBurn (less need to burn residue if harvested); MaxRes (more residue to harvest while leaving more root C); OrgAmend (mitigate residue removal soil impacts);	Org/Regen (Harvesting crop residue for bioenergy is inconsistent with organic and regenerative agriculture)	

			RedTill (reduce erosion risk with residue removal)		
17	Org/Regen	Organic and Regenerative Ag Systems	Most other BMPs are reinforced, especially increasing plant and livestock integration, increased plant diversity, cover cropping, increased legumes, increased organic amendments	RedTill(challenges for reduced tillage in organic systems); Biochar, BioNRG (crop residue harvest)	Many practices are part of Regen and organic agriculture, so these systems reinforce application of multiple BMPs
18	IntLstck	Integrated Crop-Livestock Systems	OrgAmend (manure source); RotPeren, SavPeren (forage for livestock); RotGraz (better pasture quality and quantity)		Org/Regen (Regen ag includes); livestock GHG emissions must be considered
19	IntPest	Integrated Pest Management	InterC, Diverse (crop biodiversity component)	Legume (legume pests increased)	Org/Regen (Regen ag includes)
20	MaxRes	Maximize Crop Residue Production	Biochar, BioNRG (more residue to harvest while leaving more root C)	RedTill(residue challenges for reducing tillage); Legume (legumes can produce less crop residue than cereals or oilseeds); CoverC (cover crops can aggravate problems with excessive residue)	
21	Rebuild	Rebuilding degraded agricultural land through targeted Regenerative Ag practices	RedErsn (rebuilding requires reducing erosion); RotPeren, SavPeren (perennial rebuild soil)	Biochar, BioNRG (residue removal hurts soil rebuilding)	Part of regenerative landscape
22	SetAsid	Conversion of marginal cropland to permanent cover – Land set aside	Rebuild (manages erosion)	NoDefor (could incentivize deforestation due to market forces from set aside)	Part of regenerative landscape
23	RedErsn	Reduce soil erosion in areas of high risk	Rebuild (rebuilding requires reducing erosion); RedTill RotPeren, Sbelt, SavPeren, SetAsid (reduces erosion); CoverC (reduces erosion)	Legume (legumes can produce less crop persistent residue than cereals or oilseeds, particularly under tilled systems)	Part of regenerative landscape
24	Monitor	Monitoring practice adoption, soil health, vegetation condition	All		Supports all BMPs, no trade-offs

		to identify opportunities			
25	Diverse	Increase Diversity of Plant Species on Agriculture Land	IntPest (pest, disease management); Org/Regen (diversity is essential)		Org/Regen (Regen ag includes)

Identified Beneficial Management Practices for Improving the Greenhouse Gas Intensity of Prairie Agriculture

BMPs Pertaining to Annual Cash-Crop Production

BMPs in this section focus on improvements to annual cash cropping systems which have the potential to reduce GHG emissions. The practices seek to increase soil carbon sequestration, improve nitrogen management, and generally improve soil health. The mechanism for the improvements follow many of the guidelines outlined in the regenerative Ag principles, such as reducing soil disturbance, increasing plant diversity, and maximizing crop cover and total biomass. Biological N fixation through increased legume crops and adopting the 4R® Nutrient Stewardship principles are also covered.

1. Reduced Tillage

Description

By reducing tillage intensity, soil disturbance is also reduced. This reduced disturbance slows decomposition of SOC, thereby increasing the amount of SOC. Tillage practices are divided into intensive tillage (IT, most residue soil incorporated), reduced tillage (RT, most residue on the surface) and no-till seeding without preceding tillage (NT). Indeed, an increase in SOC from reduction in tillage practices (either from IT to RT or NT, or from RT to NT) between 1990 to 2017 accounted for 5.7 M t CO_2e/yr of SOC sequestration in Canada (ECCC, 2019).

Concerns about soil erosion and soil moisture conservation have long made tillage reduction a priority in the Prairies. The tillage type with the most soil disturbance, the moldboard plow, was essentially replaced with lower-disturbance tillage implements like the cultivator by the early 1950s. Throughout the 1960s and 1980s, the intensity of tillage in both IT and RT systems were decreasing with fewer tillage passes and more use of herbicides for weed control. Farm adoption of NT on the Prairies started in the mid 1970s and that adoption began to accelerate rapidly around 1990 when a combination of better seeding equipment, lower-priced glyphosate, and more experience with appropriate agronomic practices for NT coalesced.

Effect of reduced tillage on GHG emissions

SOC

Liang et al. (2020) provided a summary of all research for the Prairies on the effect of NT adoption on SOC. For the Prairies, they found that the average rate of C increase per year was 0.58 t C/ha for coarse-textured soils, 0.30 t C/ha for medium textured soils, and 0.43 t C/ha for fine-textured soils. These rates are considerably higher than the estimates used in Canada's national inventory report (NIR) where rates vary from 0.14 t C/ha for medium textured soils in Black soil zone to 0.06 t C/ha for coarse textured soils in the Brown soil zone (ECCC, 2022). The NIR values are lower for several reasons. The research values are for continuous NT while those in the NIR were per year of NT in systems of discontinuous NT (McConkey et al. 2014). Also, the NT definition used for the NIR had some tillage 1 out of every 10 years. Perfect NT is difficult to achieve because tillage is used for rut removal, strategic control of herbicide-resistant weeks, seedbed preparation after perennial forages, and incorporation of solid manures or biosolids (Wilson, 2019). Whereas research experiments minimize soil disturbance, there can be additional soil disturbance on NT farm fields. Banding fertilizer in a separate operation than direct seeding adds soil disturbance but has always been considered consistent with NT. Heavy spring tooth harrows spread the surface crop residue and disturb the upper one cm or so of the soil but is considered an accepted NT practice. Vertical tillage with low-angle discs has gained popularity to cut the residue and do some residue incorporation into the soil to improve direct seeding performance under high-residue situations. It is widely considered by producers as consistent with a NT system (Wilson, 2019). Hence the SOC sequestration rates from research experiments where soil disturbance is more stringently minimized may overestimate the potential SOC gains from practical NT application where than can be much more soil disturbance.

The Prairie soil carbon balance project monitored SOC change over 90 commercial farm fields that had converted to NT in 1997 for 21 years (McConkey et al. 2020). The change in SOC (due to both change in tillage and reduction in fallow) were very similar to what would have been estimated using NIR methods.

N_2O and other fluxes

Liang et al. (2020b) summarized the effects of tillage system on N_2O emissions. They found that conservation tillage (RT or NT) reduced direct N_2O emission by an average of 27% compared with intensive tillage on the Prairies.

Drever et al. (2021) estimated reductions in fossil fuel use were 0.025 Mg CO_2e/ha for IT to RT transitions and 0.015 Mg CO_2e/ha for RT to NTl transitions on the Prairies.

Agriculture is unique in that it can quickly change the surface cover and thereby the land surface albedo (radiation reflectance) over huge areas. This albedo effect has important impacts on radiative forcing. Liu et al. (2022) showed that the increase in NT over 1990 to 2019 on the Prairies reduced the radiative forcing due to its larger albedo (greater radiation reflection). This albedo effect would be comparable to the radiative forcing effect of a 180 Mt CO_2e reduction in emissions. The effect of increased albedo was about twice the effect of the cumulative C sink in CO_2e units that was produced by those tillage system changes over the same period (from summing C sink values from NIR over that period). Other changes to surface cover are also important such as they documented that canola has higher albedo than wheat. Changes in albedo are not yet considered in national GHG accounting but their effects can be as important as direct GHG emissions/removals from agricultural land management.

Current adoption

The best available values for adoption are from the Census of Agriculture (Statistics Canada, 2022) (Table 2).

Table 2: Percentage of intensive tillage (IT), reduced tillage (RT), and no-till (NT) in the Prairies in 2011, 2016, and 2021. Source: Statistics Canada, Census of Agriculture.

	2011			2016			2021		
Province	IT	RT	NT	IT	RT	NT	IT	RT	NT
AB	12.8	22.1	65.1	11.6	19.0	69.4	9.2	16.1	74.7
MB	36.3	37.6	26.2	39.0	38.3	22.7	29.0	41.6	29.5
SK	11.1	20.6	68.3	8.0	18.9	73.1	4.9	17.5	77.6

Potential adoption and GHG benefits

For the provinces of Alberta and Saskatchewan, Drever et al. (2021) estimated that by 2030 it is feasible to further increase NT adoption to 85% (from 80% in 2016) within the drier Brown and Dark Brown soil zones and to 70% (from 64% in 2016) in the cooler and wetter Black, Dark Gray, and Gray soil zones. These increases represent 1.3 M ha of additional NT areas. In Alberta and Saskatchewan, they also assumed that half of the area of IT would be converted to reduced tillage. For Manitoba, they concluded that true NT production was limited but there was an opportunity to convert at least half the land in IT to RT as the latter tillage system can be adapted to cool, moist seeding conditions better than NT while mitigating much of the soil erosion and moisture loss problems of IT. Drever et al. (2021) estimated potential mitigation from the above 2030 adoption (Table 3). The 2021 Census of Agriculture, which was not available for the Drever et al. (2021) study, shows that the potential 2030 adoption of NT in Manitoba was underestimated. The NT and RT adoption in Alberta and Saskatchewan have about reached the 2030 values although the other potential tillage reductions are consistent with 2021 tillage system adoption.

Table 3: Potential tillage system adoption and mitigation (Drever et al. 2021)

Province	No-Till Adoption (%)	Reduced Tillage Adoption (%)	Mitigation (Mt CO_2e/yr)
Alberta	75.3	18.9	0.2
Saskatchewan	78.4	17.6	0.24
Manitoba	25	55.5	0.25

Barriers to adoption

The increasing prevalence of herbicide resistant weeds is a barrier as some tillage may be needed for weed control.

Crop yields have been steadily increasing in Canada. This situation increases the challenge to better manage increased crop residue for optimum NT seeding in high-yield regions such as the Red River Valley in Manitoba. A particular challenge with high amounts of crop residue is less timely seeding due to delays for the seedbed to dry and warm up sufficiently for good crop establishment.

Increased production of cover crops could further challenge residue management for NT seeding and thereby reduce its adoption.

Co-benefits

Major co-benefit is reduced soil erosion.

Another important co-benefit is better moisture conservation that is important in drought years. The improved moisture conservation from conservation tillage also made continuous cropping without frequent fallow more more feasible. Reducing fallow improved soil health. Biodiversity is increased by reducing the amount of surface and soil disturbance. Many animals prefer undisturbed soil and many live in and feed on crop residues. Increases in soil organic matter content promote greater microbial biomass and potentially, microbial resilience and diversity.

Many crops that are now widely grown on the Prairies are poorly suited to production under IT. These include pulse crops that do not produce enough residue for satisfactory soil erosion control under IT, broadleaf crops whose emerging seedlings are irreparably damaged by wind erosion, and small seeded crops like canola that require shallow seeding into moist seedbed that is best achieved with NT. Consequently, the diversity of crops produced on the Prairies is fostered by conservation tillage.

Trade-offs

By replacing weed control by tillage with herbicides, NT and RT can increase the quantityof herbicides that are applied for weed control. Increased use could increase their losses in runoff into streams and harm some aquatic organisms. Herbicides are carefully regulated to reduce this latter harm. Increased herbicide use also reduces some of the emission savings from less tillage with conservatin tillage. Other than for organic production, IT production on the Prairies also relies heavily on herbicides.

Knowledge Gaps

GHG emissions and removals of NT as applied on farms needs to be better quantified to have more accurate estimate of the amount and uncertainty of GHG impacts.

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2. Cover Crops

Description

A cover crop refers to a crop grown when the land would otherwise not have living vegetation and that is in addition to normal production of cash crops that are harvested for grain or forage.

Shoulder-season cover crops are grown after cash-crop harvest and/or before cash-crop seeding. Interseeded, also called undersown, cover crop is when a shoulder-season cover crop is planted within the cash crop. This planting could be when the cash crop is planted or later after the cash crop has emerged. In the 2019 Prairie cover crop survey (Morrison, 2021), 37% of cover crops are seeded with the cash crop and 17% were broadcast into the cash crop. The cover crop species is chosen so it grows primarily below the cash-crop canopy, so it does not compete much with the cash crop or interfere with cash crop grain harvest. Most of the cover crop growth happens after cash-crop maturity and harvest. Alternatively, a shoulder-season cash crop can be planted after cash crop harvest. In the 2019 Prairie cover crop survey, 28% of cover crops were seeded after cash-crop harvest. A winter cover crop continues to grow the next spring before the next cash crop is grown. It is also possible to seed a cover crop into frozen soil, so germination and emergence is the next spring with the cover crop growth terminated before the next cash crop.

A full season cover crop is grown during the normal growth period of a cash crop. Some farmers deliberately fallow the land by not growing a crop in the normal growing season. A cover crop grown on this fallow provides a valuable soil amendment from the plant growth that otherwise would not have happened. Such crops may also be called green manure crops. Some growers are primarily interested in the various soil and agronomic benefits of the green manure crop and may grow this type of cover crop when, otherwise, they would grow a cash crop.

Forages established within or immediately after a cash crop are not considered cover crops in this study when the forage grows for one or more subsequent growing seasons. This practice of interseeding forages with a cash crop, often called companion cropping, is considered an established practice for forage establishment. Forage crops provide many soil and environmental benefits, but these are attributed to forage production over years, not to the companion crop during the establishment year.

An intercrop, when two or more crop types are grown together but all harvested for grain, is not considered to be a cover crop. Winter cereals grown for grain harvest provide some of the benefits of the cover crop in terms of reducing nitrate leaching and protecting the soil from erosion in the fall, winter, and early spring, but are not additional to cash crop production so are not considered cover crops.

There are many species options for cover crops including grasses (winter cereals such as wheat and rye, spring cereals such as oat or barley, forage grasses such a ryegrasses), legumes (e.g., alfalfa, vetch, clover, lentil, pea) and non-legume broadleaves (e.g., radish, buckwheat, purple tansy). An increasing practice is to have a mix of species and types to both better capture the various benefits provided by each and to have at least some species that will be suitable for whatever weather is received during cover crop growth. According to the 2019 Prairie Cover Crop Survey (Morrison, 2021), 76% of cover crops used mixtures have 2 or more species with 43% of those having 6 or more species. Research in the northern Great Plains of the US shows that individual crop species in the mixture are more important to stand productivity than having a mixture itself and that, regardless of the mixture, the timing of precipitation during the cover crop growth period was the most important factor to stand productivity (Hendrickson et al., 2021; Chim et al., 2022).

Effect of Cover Crops on GHG emissions

Soil Carbon

Global meta-analyses show that cover crops mitigate GHG emissions through increased C sequestration (Abdalla et al., 2019; Bai et al., 2019). This mitigation is attributed to increased C input to the soil from the cover crops.

There are no measurements of C sequestration for shoulder-season cover crops in the scientific literature for the prairies. The measured C sequestration rates for the southern Ontario and Quebec are 0.24 Mg C ha^{-1} yr^{-1}(Agomoh et al., 2020), and 0.67 Mg C ha^{-1} yr^{-1} (Yang and Kay, 2001). However, in that same region, Jarecki *et al.* (2018) and N'Dayegamiye and Tran (2001) found no effect on SOC.

In the Prairies, measured mean rates of C sequestration were 0.2 Mg C ha^{-1} (Campbell *et al.*, 2007) to 0.32 Mg ha^{-1} (Biederbeck *et al.*, 1998) for a cover crop grown instead of fallow in the Brown soil zone. In North Dakota, green manure cover crop did not increase SOC measurably over one year compared to no cover crop (Liebig *et al.*, 2010)

The strong relationship between C inputs and SOC stocks is well established (Liang *et al.*, 1998; Maillard *et al.*, 2018; Smith *et al.*, 2018) and has been used to estimate effect of cover crops on SOC (Koga *et al.*, 2011; Poeplau and Don, 2015). The global mean C input of 1.87 Mg C ha^{-1} yr^{-1} from the cover crop (Poeplau and Don, 2015) has been associated with an SOC increase of 0.32 Mg C ha-1 yr-1. Comparable values of C input have been observed for mixed wood plains ecozone of Ontario and Quebec for the favourable situation of cover crops seeded with wheat. In this latter region, assuming a shoot:root ratio of 0.28 for annual grains and 0.44 for legume cover crops (Hu *et al.*, 2018), the calculated C inputs are: average of 1.4 Mg C ha^{-1} yr^{-1} (Garand *et al.*, 2001), 1.8 (Wagner-Riddle *et al.*, 1994), and 2.8 Mg C ha^{-1} yr^{-1} (N'Dayegamiye *et al.*, 2015). Globally, there is a non-significant trend for legume cover crops to produce lower SOC increases than non-legume cover crops (Poeplau and Don, 2015; Abdalla *et al.*, 2019).

The earlier in the season the previous cash crop is harvested, the greater the expected growth of the cover crop. In Quebec, an estimated input of 1.27 Mg C ha^{-1} yr^{-1} for clover interseeded into spring wheat versus 0.31 Mg C ha^{-1} yr^{-1} was observed when interseeded into later maturing corn (N'Dayegamiye *et al.*, 2015).

Cover crop yields in the Prairies have been lower than those observed in warmer and moister climates. For the Black soil zone, cover crop biomass yields about 0.5 to 0.6 Mg C ha^{-1} yr^{-1} , (Thiessen Martens *et al.*, 2001; Thiessen-Martens *et al.*, 2015; Cicek et al., 2014a) with a range of 0.1 to 2.6 Mg C ha^{-1} yr^{-1}. In the semiarid prairie in North Dakota, Hendrickson et al. (2021) found cover crop C yields for a range of cover crop types and mixtures seeded in late August ranged from near 0 in a dry year to 0.6 Mg C ha^{-1} yr^{-1}. Also in the semiarid prairie, but in southern Alberta, C input from cover crops ranged from near 0 in dry years to 0.3 Mg C ha^{-1} yr^{-1} in moister years (Blackshaw *et al.*, 2010). At Saskatoon, Saskatchewan, Farzadfar et al. (2021) had rye cover crop yields seeded in late August to mid-September that produced 0.2 to 1.3 Mg C ha^{-1} yr^{-1}.

To estimate the rate of C sequestration for different zones on the Prairies, Drever *et al.* (2021) scaled the C sequestration to expected mean C input from cover crops. The C sequestration rate varied by previous crop type and soil zone and ranged from 0.03 Mg C ha^{-1} yr^{-1} for cover crop following a late maturing crop not well suited to interseeding in the Brown soil zone to 0.16 Mg C ha^{-1} yr^{-1} following a winter cereal in the Black soil zone. For the crop grown instead of fallow, the rate varied from 0.26 for the Brown soil zone to 0.48 Mg C ha^{-1} yr^{-1} for the Black soil zone.

In a subsequent analysis for Farmers for Climate Solutions, McConkey (2022) used the estimated additional C input for cover crops by soil zone and previous cash crop to estimate SOC change. The SOC change was estimated by using the IPCC Tier 2 steady state model (IPCC, 2019), applied at the ecodistrict level, a method that was first implemented in the 2022 Canadian NIR (ECCC, 2022). The model is based on the well-accepted Century model and estimates SOC change based on average C input to the soil and annual weather. The simulation started in 1971, after SOC initialization to match the SOC in Canada's National Soil Database (Agriculture and Agri-Food Canada, 2021), based on actual crop yield and weather data to 2019. This method should represent the current soil conditions of Prairies soils and how increased C input from cover crop will affect their SOC stocks. Another advantage of this method is there are measurements of both full season and shoulder season cover crop biomass in the Prairies to base estimates of cover-crop C input. To use a C sequestration factor as in Drever et al. (2021) requires measurements of SOC change for shoulder season cover crops in the Prairies that are currently lacking but essential to reliably estimate SOC change.

N_2O Emissions

Cover crops affect direct N_2O emissions by affecting the C and N cycles. Cover crops also affect indirect N_2O emission by reducing N lost in leaching and runoff.

Drever *et al.* (2021) estimated the impact on direct N_2O emissions based on the proportion of legumes biomass in the cover crop mix. Based on global meta-analyses (Basche *et al.*, 2014; Poeplau and Don, 2015; Han *et al.*, 2017; Abdalla *et al.*, 2019; Muhammad *et al.*, 2019), they estimated that non-legume crop reduced annual direct N_2O emissions in cold climates (Muhammad *et al.*, 2019) while emissions would be increased by legume cover crops; the latter emission increase with legume cover crops is consistent with the only published observation in Canada (Quesnel *et al.*, 2019). Drever et al. (2021) summarized the effect on direct emissions as 10% higher emission if the cover crop was 100% legume but 10% lower emissions if there were no legumes in the cover crop; the effect was further assumed to vary linearly with legume biomass proportion of cover crop between these limits. The adjustment was applied to the N_2O emission of the preceding cash crop.

Globally, compared to no cover crops, cover crops significantly reduce nitrate leaching (Thapa *et al.*, 2018). This reduction has also been shown in the northern Great Plains of the US (Bawa *et al.*, 2021). Abdalla *et al.* (2019) found leaching was reduced with non-legume cover crops by about 50% and about 30% with legume cover crops. Drever et al. (2021) estimated the reduction linearly between these two estimates, based on the biomass proportion of legumes in the cover crop. Further, these reductions based on global meta-analysis were assumed to be relevant for a global average cover crop production having C input of 1.87 Mg C ha^{-1} yr^{-1} (Poeplau and Don, 2015). Bawa et al. (2021) found that the reduction of nitrate leaching increased with increasing cover crop biomass. Therefore, Drever et al. (2021) assumed that the N loss reduction decreased to the same proportion as the estimated C input on the Prairies to that global mean C input.

Liebig et al. (2010) provide the only measurements of N_2O emissions in the scientific literature for a cover crop grown instead of bare fallow and that was only for one year. They found no difference with and without the non-legume cover crop in their study.

N balance

In southern Manitoba, Thiessen-Martens et al., (2005) investigated the N benefit for cover crop with a winter cereal cash crop at two sites over a single production cycle. At one site there was no discernable N benefit, which was attributed to dry conditions and possibly high pre-existing N level. At the other site, they determined that an interseeded legume cover crop provided an N benefit of 24-62 kg N ha^{-1} to the following oat crop, while a cover crop seeded after winter cereal harvest produced a N benefit of 23-49 kg N ha^{-1}. In northwestern Alberta, Soon and Clayton (2003) found the average N provided by a full season legume cover crop grown was 31 kg N ha^{-1} yr^{-1}over 8 years. In southwestern Saskatchewan, a full season legume cover crop provided 49 Kg N ha^{-1} yr^{-1} Biederbeck et al., 1998). In southern Alberta, Blackshaw et al. (2010) found a N benefit of extra 18 to 20 kg ha^{-1} for interseeded legumes. Drever et al. (2021) used the N benefit from legume cover crops from (OMAFRA, 2017), that is 70 kg N ha^{-1} for corn and 45 kg N ha^{-1} for all other following crops. They scaled that benefit linearly by the proportion of legume biomass in the cover crop and also scaled linearly by the 1.87 Mg C ha^{-1} biomass yield. These estimates for crops are generally consistent with the above measured values for the Prairies (all were not corn). Janzen et al. (1990) noted that the greatest advantage of cover crop N in the Prairies is the rebuilding of stable soil organic N reserves.

Drever et al. (2021) also assumed that the estimated N that the cover crop prevented from loss by leaching and runoff was added to the N benefit of the next crop. This assumption is consistent with findings of Farzadafar et al. (2021) that a non-legume cover crop reduced mineral soil N and improved N use efficiency of a subsequent crop in Saskatchewan.

The additional N provided by a cover crop to the next crop would reduce embodied GHG emissions from manufacture of N fertilizer. The fertilizer N reduction from N supplied by a cover crop was assumed to have embodied C footprint of 4.05 kg CO_2e/kg N (Dyer et al. 2017) for the Drever et al. (2021) and Burton et al. (2021) studies and, to reflect more efficient current fertilizer manufacture, 3.180 kg CO_2e/kg N (Cheminfo Services Inc., 2016) for the McConkey (2022) study.

Other emissions

There are additional emissions for cover crops. Drever et al. (2021) estimated 14 kg CO_2e ha^{-1} as the fossil fuel emissions from shallow soil disturbance for the seeding (Dyer and Desjardins, 2003). They also used 91 CO_2e ha^{-1} for the embodied emissions to produce and transport cover crop seed to the sowing location (Dewayne, 2013). Cover crops may require a separate operation for termination and possibly mechanical treatment such as crimping. Drever et al. (2021) assumed that the energy for these operations were equivalent to conventional seedbed preparation, an assumption that would need refinement where the situation requires separate termination operation.

Uncertainty about GHG effect for some cover crop situations

Full-season cover crops grown instead of a cash crop

The GHG effects of growing a full-season cover crop where a cash crop would otherwise normally be grown has not been well researched. A first estimate of its GHG effect relative to the displaced cash crop could be made from differences in C and N input to the soil using GHG estimation methods in the NIR. However, treating an unharvested cover crop the same as a conventional harvested cash-crop is questionable since an unharvested cover crop will have more labile organic matter, particularly if the green manure crop growth is terminated before complete cover crop senescence. This more labile C input will affect both C and N dynamics (Mitchell et al., 2013; Chahal and Van Eerd, 2020; De Notaris et al., 2020). In the semiarid Prairies, although there are differences in C input, measurements show no SOC benefit to growing a cover crop on bare fallow compared with growing a cash crop instead of bare fallow (Campbell et al. 2007). Research involving measurements of N_2O emissions and SOC for cropping systems with full-season cover crops compared to systems with only harvested cash-crops is necessary to accurately estimate the GHG impact of a full season cover crop.

Reducing cash crop production to grow full season cover crops raises the possibility that some stakeholders will want to attribute some emissions for indirect land-use change induced by that reduction in cash-crop production (Parra Paitan and Verburg, 2019). This will reduce the GHG benefit of growing cover crops.

Grazing cover crops

Grazing cover crops can provide important economic benefit (Thiessen-Martens and Entz, 2011); 58% of respondents to the 2019 cover crop survey listed grazing as a reason they grew cover crops. There is little information on how grazing affects the GHG balance. Burton et al. (2021) provided preliminary estimates of the effect of grazing on shoulder-season cover crop GHG impact so it could be included, although they did not include grazed cover crops in their estimates of potential adoption. They assumed that grazing decreases total growth by 20%, grazing removes 70% of aboveground growth with 80% digestibility, and the root:shoot ratio for cover crop of 0.2 in upper 30 cm of soil (Hu et al., 2018); under these circumstances, the grazing would reduce by about 50% the C returned to the upper 30 cm of soil.

The effect of grazing on N leaching of cover crop is more complicated. Burton et al. assumed that 50% of N that would have been prevented from leaching by the cover crop did not occur due to reduced growth from grazing and return of readably leachable N in urine of grazing livestock. Consistent with this assumption, Cicek et al. (2014b) found greater soil nitrate in the 120 cm soil profile in Manitoba with a grazed cover crop than when ungrazed. Cover crops seeded after grazing could reduce the residual N (Cicek et al. 2015). Based on two recent studies in the northern Great Plains of the US (Abagandura et al., 2019; Singh et al., 2020), the grazing of shoulder-season cover crops had no effect on direct N_2O emissions from the soil. The indirect GHG effects of the new feed from cover crop is complex. As a preliminary estimate, Burton et al. (2021) suggested using the assumptions that feed provided by cover crops does not increase GHG emissions from the livestock or from elsewhere.

Cover crop after perennial forage or pasture

In the 2019 Prairie cover crop survey, 16% of respondents had cover crops after termination of perennial forage or pasture. Clearly, this situation is different from the perspective of C and N cycling than that after harvested cash crop and will depend on the species mix in the perennials. Further, the timing and nature of perennial termination will impact the effect of the cover crop – termination by herbicide versus by intensive tillage will undoubtedly affect the impact of a subsequent cover crop and a perennial terminated in the spring would be expected to have different impact than if termination was during the summer or early fall. Owing to the complexity, it is not possible to estimate the GHG effects of cover crop without more research and not possible to generalize without information on the perennial type and termination timing and method.

Potential Impact of Cover Crops on Prairie GHG Emissions

Current Adoption

Based on the 2017 farm management survey and assumptions about suitability for cover crop, Drever et al. (2021) estimated there was 11 000 ha of full-season cover crops and 104 000 ha of shoulder-season cover crops in the prairies. The cover crop definitions in this survey were ambiguous. The 2019 Prairie cover crop survey went to great effort to reach cover crop growers and had responses from 211 growers who grew 34 000 ha of cover crops, of which 20 000 ha were shoulder-season cover crops and 14 000 were full season cover crops. The respondents came from across the Prairie provinces but 46% of the respondents and 54% of the area in the 2019 Prairie cover crop survey were in Manitoba.

Potential Adoption and total GHG benefits

Drever et al. (2021) presumed that cover crops are attractive for adoption with cash crops that are typically harvested early, such as winter cereals and pea. They also presumed that cash crops that are suited to interseeding cover crops due to their competitiveness and no major interference with cash-crop harvest, such as cereals or canola, would favour adoption. Finally, they assumed that areas of the prairies with a combination of water and warmth in the fall, particularly, the Black soil zone, would be more attractive than other soil zones that are more likely to have either dry or cold weather after cash crop harvest. Finally, they assumed that as suitability for cover crops decreased, the maximum feasible adoption rate also decreased. The reason was that as the suitability of the situations for cover crops decreased, cover crop adoption would be restricted to the most favourable portion of those situations such as for the earliest harvested portion of cash crop, years where the weather for cover crop growth was expected to be good, and/or where next year's planned cash crop is expected to work well with the preceding cover crop. For example, maximum feasible adoption of a cover crop was assumed to be up to 90% following a winter cereal in the Black soil zone but only 5% after a late maturing cash crop not suited to interseeding in the typically dry Brown soil zone or in the typically cold Gray soil zone.

Based on the above factors, Drever et al. (2021) analyzed how the value of net GHG emission reductions, in \$ t^{-1} of CO_2e, affected potential cover crop adoption by 2030. The only additional private benefit was for N added by the cover crops. At \$10 t^{-1} CO_2e, the prairies were estimated to increase full season cover crop by 219 000 ha on existing fallow but decrease shoulder season cover crop area by 76 000 ha (Table 4). At \$50 t^{-1} CO_2e, all available fallow land was in cover crops (860 000 ha) with additional 1 949 000 ha of shoulder-season cover crops compared to the 2017 area estimate. A carbon value increase from \$50 to \$100 was predicted to increase the area of shoulder-season cover crop to increase by 8 635 000 ha in the Prairies. The estimated total GHG mitigation potential at \$100 t^{-1} CO_2e was 6 Mt CO_2e. The assumed maximum feasible adoption on the Prairies was estimated to be 18.1M ha with a total mitigation potential of 7.4 Mt CO_2e. Of note, 97% of maximum potential adoption occurred in Manitoba at \$100 t^{-1} CO_2e while that carbon price only incentivized 60% and 54% of maximum cover crop area in Alberta and Saskatchewan, respectively. On the Prairies, Manitoba is much more favoured for cover crops because of more favourable (wetter) growing conditions along with greater N benefits from cover crops than the other Prairies provinces. Alberta and Saskatchewan have significant portion of their cropland that is assumed marginal for cover crops from the perspective of GHG benefits. The 6.6M ha of more marginal land from a GHG perspective between maximum adoption and that estimated to be adopted at \$100 t^{-1} CO_2e only added GHG mitigation by 1.5 Mt CO_2e.

The Farmers for Climate Solutions (Burton et al. 2021), used the same production assumptions of Drever et al. (2021) but the economic incentive for cover crop adoption was a per hectare external payment or subsidy (Table 5). Private grower benefits from cover crops that were additional to this subsidy were the N benefit plus other non-N benefits assigned a value in \$ t^{-1} of above-ground biomass (AGB) of cover crop, presuming that the non-N average private benefits are proportional to average cover crop production. In this analysis, the size of the per hectare payment dominated the adoption of cover crops. To get 5% cropland area adoption on the Prairies of cover crop production would require a \$77.90 ha^{-1} subsidy if cover crop was only valued at \$5 t^{-1} of AGB. The estimated mitigation provided was 0.5 Mt CO_2e.

None of the above analyses include any cover crop situations that are more complex to estimate because of either uncertain GHG benefits or uncertain effect of adoption incentives: full season cover crop grown instead of a feasible cash crop (i.e., full-season cover crop in analyses was limited to area of fallow), grazed cover crops, or after terminated perennial pasture or forage.

Barriers to adoption

The cost for cover crop seed and its sowing is a major barrier to adoption. In the 2019 Prairie Cover Crop Survey, 54% of respondents reported cover crop seed costs from \$25 to \$75 ha^{-1}, while 26% reported costs more than \$75 ha^{-1} and 6% reported more than \$124 ha^{-1}. Black medic is a persistent self-seeding cover crop and provides agronomic and soil health benefits with a one-time cost for seed (Stainsby et al., 2020; May et al., 2022)

Another barrier is labour and equipment constraints for seeding cover crops. If seeded after cash-crop harvest, the cover crop seeding can conflict with general cash-crop harvest – a time of usual labour shortage.

The uncertainty regarding the timing and value of private benefits to the farmer from cover crops is a barrier (Thompson et al., 2020); this is probably particularly important on the Prairies where there is relatively little hard data on the benefits of cover crops. Elsewhere, farmers generally report that benefits for improved soil resilience to various stresses, reduced loss of soil nutrients with soil erosion and leaching, reduced extra tillage to repair channeling from soil erosion, better soil biological health that improves soil structure and nutrient cycling, and/or less expensive management of weeds, diseases, and/or pests are at least sufficient to pay for seeding costs (Bergtold et al., 2017; Roesch-McNally et al., 2018). Based on a survey for farmers across the US, many of the soil benefits increase over time and some benefits only occur periodically depending on conditions, so it may take 3 years to just cover seeding costs and 5 years of continual use to have total benefits that exceed annual costs for seeding cover crops (Myers et al. 2019). Thus, the benefits are difficult to quantify exactly. Nevertheless, the benefits can be high; O'Reilly et al. (2011) reported on private value of cover crops as high as \$600 ha^{-1} yr^{-1} for seed corn in southern Ontario.

Full-season cover crops do not qualify as fallow for the next cash crop for crop insurance purposes and thereby can reduce the available coverage for that cash crop compared if no cover crop was grown.

Table 4: Effect of scenario in 2030 on added area of cover crops on existing fallow (A/f), added area of cover crops with cash crops (A/c), and emission change (Mit.) from the baseline scenario (Drever et al., 2021).

	---------- Scenario ----------											
	\$10 (Mg CO_2e)$^{-1}$			\$50 (Mg CO_2e)$^{-1}$			\$100 (Mg CO_2e)$^{-1}$			Maximum		
	A/f	A/c	Mit.	A/f	A/c	Mit.	A/f	A/c	Mit.	A/f	A/c	Mit.
Province	(‘000 ha)		(Gg CO_2e yr^{-1})	(‘000 ha		(Gg CO_2e yr^{-1})	(‘000 ha)		(Gg CO_2e yr^{-1})	(‘000 ha)		(Gg CO_2e yr^{-1})
AB	45	-25*	-73	256	767	-767	256	3128	-1781	256	5353	-2322

MB	38	-9	-65	40	378	-300	40	2788	-1376	40	2861	-1394
SK	136	-42	-251	564	804	-1218	564	4648	-2848	564	9014	-3825

*negative value indicates area loss from assumed baseline

Table 5. Effect of scenario in 2030 on added area of cover crops on existing fallow (A/f), added area of cover crops with cash crops (A/c), and emission change (Mit.) from the estimated baseline scenario for values of cover crop biomass and area payment (Burton et al., 2021).

Province	Cover crop benefits per Mg ha^{-1} of above-ground biomass + area payment											
	\$10 per Mg ha^{-1} +\$10 ha^{-1}			\$10 per Mg ha^{-1} +\$70 ha^{-1}			\$20 per Mg ha^{-1} +\$50 ha^{-1}			\$30 per Mg ha^{-1} +\$70 ha^{-1}		
	A/f	A/c	Mit.	A/f	A/c	Mit.	A/f	A/c	Mit.	A/f	A/c	Mit.
	('000 ha)		(Gg CO_2e yr^{-1})	('000 ha		(Gg CO_2e yr^{-1})	('000 ha		(Gg CO_2e yr^{-1})	('000 ha)		(Gg CO_2e yr^{-1})
AB	38	-32*	-52	256	361	-480	256	2360	-1361	256	4910	-2052
MB	28	-13	-43	40	373	-270	40	2451	-1142	40	2853	-1274
SK	107	-59	-174	564	270	-805	564	2523	-1801	564	7668	-3282

*negative value indicates area loss from assumed baseline

Table 6: Area and emission reduction from baseline for a scenario of increased adoption to 5% of the Prairie cropland area (McConkey, 2022).

			-------------------- per ha of additional cover crop -------------------				
Province	2030 baseline cover crop adoption (ha)	2030 scenario cover crop adoption (ha)	SOC increase (t C/ha/yr)	Decrease in direct and indirect N_2O-N emissions (kg/ha/yr)	Added N credit (kg N/ha/yr)	Additional emission for cover crop seed and machinery operations (t CO_2e/ha/yr)	Total Emission Reductions from the Baseline (t CO_2e/yr)
AB	33 200	437 100	0.11	0.28	8	0.105	178,000

MB	15 500	203 800	0.08	0.33	8	0.105	67,000
SK	58 800	773 900	0.09	0.23	8	0.105	267,000

Co-benefits

Soil health

With the possible exception of increased N_2O emissions, in a meta-analysis, Daryanto et al. (2018) found that, overall, the ecosystem services from cover crops are positive and cover crops should be a recommended practice for all cropland. Cover crops increase biodiversity of soil organisms (Elhakeem et al., 2019), as well as their activity and abundance (Kim et al. 2020) and increase soil health (Obour et al., 2021). In the 2019 Prairie cover crop survey, the three most important reasons for adoption cover crops relate to soil health and soil biology: 90% of respondents listed building soil health, 78% listed keep roots in soil, and 74% listed feed soil biology as reason for adopting cover crops.

Added Nitrogen

In the 2019 Prairie cover crop survey, 68% of respondents listed this as an important benefit. This benefit was covered above in the section related to GHG benefits.

Erosion control

Cover crops help reduce wind and water erosion (Baumhardt et al., 2015, Kaye and Quemada, 2017); 53% of respondents to Prairie cover crop survey listed erosion control as one of the reasons they have adopted cover crops.

Weeds and Pests

Cover crops suppress weeds in the prairies (Flood and Entz, 2018) and US Great Plains (De Haan et al., 1997; Kumar et al., 2020); weed suppression and pest control were listed by 62% and 30%, respectively, of respondents in the Prairie cover crop survey.

Nitrogen contamination of water

Cover crops reduce nitrate leaching (Thapa et al. 2018) and can reduce nutrient loss in runoff (Dabney et al., 2001). No farmer in the Prairie cover crop survey mentioned this as a benefit.

Cash-crop yields

There are limited data on the effect of cover crops on subsequent cash crop yield. In humid climates where corn is a major crop, preceding cover crops increased corn yield by an average of up to 33% (Marcillo and Miguez, 2017). However, for a global meta-analysis of dryland agriculture where precipitation is less than potential evapotranspiration, Garba et al., (2022) found that the relative yield of cash-crop after cover crop compared to no cover crop decreased as annual precipitation decreased. This yield decrease was related to lower water conservation after cover crops. Nevertheless, considering only regions with continental dryland climate that is relevant to the Canadian Prairies, the average relative yield with cover crops was 104% of that without cover crops. In the Brown soil zone of the Canadian Prairies, average spring wheat yield after a legume cover crop was not statistically different than that grown on fallow without a cover crop over a 25-yr period (Kröbel et al., 2014).

Trade-offs

Potential P losses

There is concern about cover crop increasing P losses in winter and spring runoff (Daryanto et al., 2018), an important potential P-loss pathway for Canada (Liu et al., 2019). However, the limited field studies with cover crops in Canada have not shown an increase in P loss (Lozier et al., 2017; Schneider et al., 2019). Further investigation is needed to determine if cover crop adoption may need some restrictions because of P losses to surface water (Liu et al., 2019).

Water conservation–trade-off or benefit?

Water is the greatest limitation to crop production in the prairies. The soil zone is a commonly used proxy for the general moisture regime. The overall water limitation is greatest in the semiarid Brown soil zone and then decreases moving into Dark Brown, decreases further into the subhumid Black and is lowest in the Gray soil zone. Cover crops use water, thereby reducing its conservation (Unger and Vigil, 1998) and so need to be managed with care in semiarid climates to prevent their water use from decreasing water available to the subsequent crop (Robinson and Nielsen, 2015). For example, in the Brown soil zone, Zentner et al. (2004) determined that full-season cover crops grown instead of fallow need to have growth terminated by early July or water use by the cover crop reduces the yield of the following cash crop, as compared to no cover crop. However, obtaining satisfactory water conservation forces a compromise as the required early termination does not allow maximum attainable growth of the cover crop and thereby does not allow maximum benefit of the cover crop to the soil.

Winter cover crops reduce leaching of nitrates into groundwater but that the portion of the reduction in nitration leaching that due to reduced downward water percolation could also reduce groundwater recharge in dry climates (Tribouillois et al., 2018).

Conversely, the Prairies can also have problems with excess water, particularly in the spring. Under that situation, the reduced water conservation after cover crops is beneficial. Excess water is most likely in the Black to Gray soil zones although it could happen anywhere depending on seasonal weather. In southern Manitoba, cover crops dried the soil, thereby allowing more infiltration and deep percolation of snowmelt (Kahimba et al., 2008); otherwise, that uninflitrated snowmelt could have created excess surface wetness.

Knowledge Gaps

Generally, there is limited knowledge for the Prairies about how to best fit cover crops into production systems and the benefits and trade-offs of cover crops.

Need for agronomic research

Research and development on optimal cover crop species, mixes, and seeding methods that provide the maximum agronomic and soil benefits under conditions of low potential biomass production for the Canadian Prairies is important (Morrison, 2021). Interseeding typically improves the performance of cover crops when there is a short season after cash crop harvest. Therefore, developing practical, low-cost but effective techniques to interseed cover crops into a wide range of suitable cash crops are needed. The optimal method and timing for termination of cover crops, including allowing cover crop to overwinter, for the varying conditions across the Prairies needs to be determined.

The short-term and long-term agronomic and soil benefits of cover crops for cash crop production need to be better quantified to inform land managers about merits of cover crop adoption (Morrison, 2021). The risks of cover crops, both the risk of suboptimal cover crop establishment and growth and the risk that cover crops could reduce cash crop production, need to be better quantified for the conditions across the prairies. As examples of the latter, experience on the prairies is that cover crops can attract and bridge some pests and diseases that are deleterious to cash-crops (Morrison, 2021). Cover crops can also complicate weed control.

The effect of cover crop on N balance and on the response to N of important subsequent cash crops requires more research across Prairie conditions.

The knowledge needs should be to be tailored to region within the prairies, be specific to type of farm, and be specific to the type of cover crop: full-season, shoulder-season, and with and without grazing (Morrison, 2021).

Need for Research on GHG impact

There is a need for significant investigations to develop data on past, current, and future cover crop use and to develop rigorous estimates of the effects of cover crops on GHG emission, particularly N_2O, and removals (SOC stock change) for the Prairies. To include cover crops in Canada's National Inventory Report, their adoption must be monitored. Cover crops can vary greatly in species composition and amount of growth -- practical methods that account for their variability are needed to estimate their impacts on SOC and N_2O emissions. The impact of growing full-season cover crops where a cash-crop would otherwise be grown needs more investigation including how grazing of that cover crop affects the net GHG emissions.

Other Environmental Considerations

The effect of cover crops on P loss to surface water requires more research to determine if there needs to be restrictions on cover crop adoption in some watersheds or localized portions of the landscape, or whether the risk to water quality can be mitigated through cover crop management practices.

The effect of cover crop on N loss to leaching and runoff needs research so that benefit can be considered for public policy.

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3. Intercropping

Description

Intercropping consists of growing two or more grain crops together. Typically, in short growing seasons, the crops are harvested together, and then the grains are separated. Intercropping is more common in organic production systems but is gaining interest for application in conventional production systems.

The main advantage of intercropping is overyielding. Overyielding is producing more grain than would have been produced if each grain was produced as a single crop. A land-equivalent ratio (LER) greater than 1 indicates overyielding. LER is defined as:

$$LER = \frac{YC1_{intercrop}}{YC1_{single\ crop}} + \frac{YC2_{intercrop}}{YC2_{single\ crop}}$$

Where $YC1_{intercrop}$ is the yield (kg/ha) of crop 1 in the intercrop, $YC2_{intercrop}$ is the yield of crop 2 in the intercrop, $YC1_{single\ crop}$ is the yield of crop 1 as a single crop, and $YC2_{single\ crop}$ is the yield of crop 2 as a single crop.

A LER > 1 means that the amount of grain produced from a unit area of intercrop is greater than that from the intercrop being grown as single crops on their fraction of the intercrop unit area. To illustrate, 1 ha of intercrop of crop A and B would produce more total grain than that from 0.5 ha of crop A alone plus 0.5 ha of crop B alone. If 30% more grain was produced, then the LER would be 1.3.

Intercrops of legumes and non-legumes have been shown to be complementary and produce better use efficiencies of water, nutrients, and light compared with single crops (Duchene et al., 2017). This complementarity provides an N advantage to the non-legume from N transferred from the legume (Chapagain and Riseman, 2014; Duchene et al., 2017). Intercropping with a non-legume also increases biological N fixation of the legume in the intercrop (Chapagain and Riseman, 2014a; Cong et al., 2015). Other agronomic advantages of intercropping are reduced disease pressure and better weed suppression (Rob et al., 2015; Gu et al., 2021). For example, *mycosphaerella* blight has become the most widespread and economically damaging disease in pea in Canada but its incidence can be greatly reduced when the pea is intercropped (Dowling et al., 2021). This benefit may not always be the case; in a study with intercropped pea with canola or mustard in Manitoba, it was inconclusive whether the intercrop affected root rot in pea caused by a complex of *Fusarium* spp. and *Aphanomyces* (Manitoba Pulse and Soybean Growers, 2021)

The above agronomic advantages explain why intercrops can have LER >1. Dowling et al. (2021) reports on 17 research trials with various legume-oilseed intercrops conducted on the Prairies. In 11 of those studies, the intercrop had higher yields than the single crops, one study had no difference, and four studies had lower yields. One of those studies with lower yields were in the Brown soil zone in Saskatchewan. These variable results suggest intercropping may not be as agronomically advantageous in drier regions of the prairies. The yield effect in one study was dependent on fertilizer addition – the LER was greater than one if both N and P were applied in fertilizer, but the LER was less than 1 if no fertilizer was applied or either N or P were applied alone. For the 49 site years of intercropping vs single crop comparisons reported by Dowling *et al.* (2021), the average LER was 1.17. The LER for legume-cereal intercrops on the Prairies are similar to those for legume-oilseed. The average LER for a pea-canola intercrop in one research trial was 1.15 while at the same site the LER for a pea-barley and pea-oat intercrops were 1.15 and 1.37, respectively (Manitoba Pulse and Soybean Growers, 2021).

The extra operation to separate the intercropped grains that are harvested together is an important additional cost to intercropping compared to single crops. However, if the grains are harvested together but fed as a mixture as livestock feed then separation is not necessary. Some intercrops are grown for silage which also negates the need for separation. Additionally, relay cropping – where one of the intercrops is harvested separately (before the other intercrop) – means grain separation is not necessary. In this case, the limitation then becomes the added time and cost for two harvest operations.

To separate two grains that are harvested together requires additional equipment for separating grains based on physical characteristics. This requirement limits the types of possible intercrops, since the two grains must be sufficiently physically different for high-throughput separation at relatively low cost. It also requires additional grain storage because storage is needed for the mixed grains before separation. Labour availability to separate the busy harvest season is a limitation. This labour constraint is made worse by the fact that conventional seed cleaning equipment does not have a high throughput since the equipment is typically designed to separate a relatively small proportion of material from a mixture that is primarily one grain. For mixed grains from intercropping, a large proportion must be separated from the mixture. Commercial equipment purpose-designed for high throughput separation of the grain mixtures from intercropping is not yet available.

There are many intercrop possibilities but those involving a legume and non-legume are particularly attractive to increase efficiency of using applied N fertilizer. Some potential intercrops for the prairies involving four non-legume-legume intercrops for the prairies are: 1) pea and canola, 2) chickpea and flax, 3) barley and lentil, 4) lentil and canola, and 5) winter wheat and soybean. Pea and canola (often called "peaola") and chickpea and flax are relatively well studied on the prairies (Dowling *et al.*, 2021). The barley and lentil and canola and lentil has been shown to work successively elsewhere (Martin-Guay *et al.*, 2018, Dowling *et al.*, 2021). Canola and lentil are much different seed sizes so can be separated easily and barley and lentil can also be mechanically separated relatively easily (Milligan, 2009). The two grains in each of the above intercrops have similar maturities. Winter wheat and soybean is grown as a relay intercrop, with the soybean seeded into the growing winter wheat in the spring and harvested in a separate operation after the wheat. There is considerable interest in this cropping practice in Ontario (McIntosh, 2021) as well as on the prairies (Manitoba Agriculture Diversification Centres, 2020).

Some intercrops are hayed or ensiled to feed cattle. Mixing a legume with cereal is common in such intercrops to produce a feed that is both high energy and high protein while taking advantage of the N fixation capability of the legume to reduce N fertilizer needed. Example intercrops for forage include pea and barley and soybean and corn.

Typically, the more competitive non-legume in the intercrop will have the higher yield (Echarte *et al.*, 2011; Chapagain and Riseman, 2014a).

Intercrops can be grown in different seeding configurations. In the Prairies, Dowling *et al.* (2021) cites three separate research studies that compared crops mixed in the seed row with each crop grown in alternating rows. In two studies the mixed rows had higher LER than the alternating row and in one study there was no LER difference between configurations. In the lower mainland of BC, alternate rows of legume and cereal were found to have higher LER than when the two crops are mixed in the same row (Chapagain and Riseman, 2014a,2014b).

Effect of intercrop on GHG emissions

Soil Carbon

The more efficient use of water, nutrients, and sunlight in intercropping systems produce more biomass than single crops. This efficiency increases the C input to the soil compared to single crops. The LER of biomass is typically greater than the LER for grain alone (Chapagain and Riseman, 2014a; Cong *et al.*, 2015). This situation arises because the competition between intercrops produces relatively more leaf, stem, and root biomass than in single crops. The LER is usually lower in drier areas. In fact, the LER for grain can be less than 1 for intercrops under dry conditions, such as southwest Saskatchewan (Dowling *et al.*, 2021). However, to understand the effect on SOC, we only need the LER for C input (residue biomass) which is expected to more consistent and higher than the LER for grain. Based on literature, McConkey (2022) estimated that the C input LER could average 1.1 for semiarid Prairies (Brown and Dark Brown soil zones) and 1.2 elsewhere in the prairies.

SOC should increase with increased C input from intercropping. Indeed, intercropping has been shown to increase SOC as compared to single crops (Cong *et al.*, 2015). McConkey (2022) estimated the SOC change using the IPCC Tier 2 steady state model (IPCC, 2019), applied at the ecodistrict level, a method implemented in the Canadian NIR starting in 2022. The model is based on the well-accepted Century model and estimates SOC change based on average C input to the soil and annual weather. The simulation started in 1971, after SOC initialization to match the SOC in Canada's National Soil Database (Agriculture and Agri-Food Canada, 2021), using actual crop yield and weather data to 2019. This modelling approach should represent average Canadian soil conditions as affected by previous crop production practices. The 2020-2030 period was set to the average weather factors and crop area for 2015-2019. The yields for the 2020-2030 period were increased based on continuation of 2005-2018 trend on an ecodistrict basis. The SOC change in 2030 was then estimated as the difference between baseline no-intercrop and the intercrop scenario for SOC change between 2029 to 2030. The average rate of SOC change for each Prairie province is given in Table 7.

There are few studies of N_2O emissions regarding intercopping. In China, with same amount of fertilizer N applied to single-crop maize as maize-soybean intercrop, N_2O emissions were less in an intercrop than the single crop (Huang *et al.*, 2019). In Argentina, where the rate of N fertilizer was adjusted to the area of maize in a maize-soybean intercrop, the N_2O emissions for the intercrop were lower than either soybean single crop or maize single crop (Dyer *et al.*, 2012). While research is needed on N_2O emissions and intercropping in the Prairies, given extant evidence it's likely intercropping could reduce N_2O emissions. This reduction would be consistent with a greater total crop growth with intercrops that scavenge mineral N effectively and thereby reduce the nitrification and denitrification reactions that produce N_2O.

N fertilizer

Dowling et al. (2021) reports on several legume-oilseed rotations conducted in the Prairies that include N fertilizer rate. For four sites, one-half the highest rate of N has little effect on LER. Assuming that the highest N rate was the normal N rate for the non-legume and that the intercrop area would be twice the total of the two single crop areas of the intercropped crops, this would represent the same total N use for the intercrop area as for single crop area since the legume receives no fertilizer N, (i.e., intercrop with 50% N rate x 100% of the total area = single crops 100% N rate x 50% of the total area). Under those assumptions, there is no reduction in the N usage although there is higher total grain production since LER >1.

The proportion of grain from the non-legume has been shown to increase as the N rate increases (Manitoba Pulse and Soybean Growers, 2016). Therefore, N management is also a way to manage the relative grain production between crops in the intercrop. Note with individual crops alternating rows, it is possible to place the N fertilizer so as to give the row with the non-legume preferential access to that N fertilizer; in rows with mixed crops, this is not feasible.

Dowling also reports on four Prairie studies where 0 N fertilizer was compared to conventional fertilization. In these studies, the LER averaged 1.4 (range 0.93-1.9). Consequently, there appear to be opportunities to reduce N use in legume-non-legume intercrops.

The Canadian methodology for estimating N_2O emission from crop production is based on an emission factor of N_2O emissions from applied N and varies with climate, soil, tillage, topography, and irrigation. If N use was reduced by just 10 kg/ha in intercrops from one-half the typical non-legume N application rate, the emission reduction across the ecodistricts of Alberta varies from 4.7 to 95.6 kg CO_2e/ha, across Saskatchewan from 36.1 to 5.6 kg CO_2e/ha, and across Manitoba from 36.1 to 5.6 kg CO_2e/ha. The avoided embodied emission from that 10 kg/ha reduction in N application is 31.8 kg CO_2e/ha (Cheminfo Services Inc. 2016).

Therefore, there is potential for significant reductions in emissions of N_2O from legume-non-legume intercrops compared with their single crop production. The impact of the intercropping on the emission factor could add to these reductions.

Other emissions

For relay cropping, there is an additional sowing and harvest operation. Dyer and Desjardins (2003) estimate emissions of 0.038 kg CO_2e/ha for sowing and 0.030 kg CO_2e/ha for harvest in central Canada. We could not find an estimate for grain separation required for other intercrops in the literature. Therefore, we used the 0.030 kg CO_2e/ha emission for harvest as an estimate for grain separation since the combine-harvester involves the actions needed for separation of intercropped grains: separation of grain from much other material, plus lateral conveyance and elevation of the grain. If electricity with a low carbon footprint is used for some or all of the power used for on-farm separation of intercropped grain, the emissions would be lower for that operation than those based on a diesel fuel-powered combine-harvester.

For intercrops, the seeding rate of the less competitive crop needs to be more than one-half the seeding rate of that crop as a single crop. Conversely, the seeding rate of the more competitive crop may require less than one-half the seeding rate of that crop as a single crop. The pesticide input can usually be reduced in an intercrop. Based on carbon footprints for Canadian canola ((S&T)^2 Consultants Inc. 2017a) and pea ((S&T)^2 Consultants Inc. 2017b), a preliminary analysis for pea-canola suggests that the embodied emission for extra seed (5.9 kg CO_2e/ha) and for reduced pesticides (8.3 kg CO_2e/ha) roughly cancel out. More accurate analyses for intercrops are needed to generate information on both optimal seeding rates and pesticide use in intercrops, as compared with single crop counterparts.

Table 7: Preliminary estimates of the effect of intercropping on embodied CO_2e emission for seeds and pesticides.

Input	Cropping system	Pea	Canola	Average CO_2e/ha
Seed	Single crop	180	5.6	--
(kg/ha)	Intercrop	150	2.8	--

Seed	Single crop	18.1	2.5	10.3
(kg/CO_2e/ha)	Intercrop	15.1	1.2	16.2
Pesticide	Single crop	31.6	17.2	24.4
(kg/CO_2e/ha)	Intercrop*	10.4	5.7	16.1

*Assumes that the pesticides needed by the crops in the intercrop is 67% of that needed as single crops

Potential Impact of Intercrops on Prairie GHG Emissions

Current adoption

There are not sufficient data on current adoption of intercropping. Surveyed experts suggest 1 to 5% of cropped area is currently intercropped.

Potential adoption and total GHG benefits

McConkey (2022) analyzed 2019 data to identify the potential crop area by province for five intercrops: 1) pea and canola, 2) chickpea and flax, 3) barley and lentil, 4) lentil and canola, and 5) winter wheat and soybean. Ecodistricts were identified that produced the two crops in the intercrop. This approach is only an indicator of *potential* as it shows that the two crops are commercially grown in the same ecodistrict (each ecodistrict is defined as having a single climate and related soil landscapes) but does not show they are grown on the same farm. The potential intercrop area was estimated as twice the smallest of the areas of the two crops in the intercrop. To illustrate, if 1100 ha of chickpea and 1350 ha of flax were grown in an ecodistrict, the potential chickpea and flax intercrop area was 2*1100 ha = 2200 ha with 250 ha of flax remaining as a single crop. This assumption means there would be no change from intercropping required of the area of other crops. Canola and lentil could appear in two different intercrops. Only the canola area in excess to that assigned for intercrop with existing pea area and the lentil in excess to that assigned for intercrop with existing barley area was available for the lentil-canola intercrop.

In this analysis, the winter wheat and soybean relay cropping were limited to southern Manitoba. The rationale was that this province typically has more total and late summer precipitation than is typical for Saskatchewan or Alberta, and the late summer precipitation would be needed by the soybean. This boundary is artificial and eastern Saskatchewan has precipitation patterns similar to those in Manitoba.

Table 8 shows the potential areas based on recent coincidence of two crops in intercrops. Of the intercrops investigated, pea-canola had the largest area where the crops are grown together in all three prairie provinces. The other intercrops ranking varied with province. For example, lentil and canola were frequently grown in same ecodistrict in Saskatchewan but, after considering potential barley and pea areas for intercrops with the former crops, there was little coincidence for lentil-canola in Alberta and none in Manitoba. From this simplistic analysis with only 5 intercrops based on current single-crop areas, the potential area of these intercrops was 21%, 28%, and 7% of the total cropland area of Alberta, Saskatchewan, and Manitoba, respectively.

Table 8: Indicator of potential and estimated emission changes (negative is net reduction) from SOC change, additional mechanical operations, and seed and pesticides.

Province	---------potential area of intercropping (ha) ----------------					SOC change	Additional Operations	Seed and Pesticides*
	Chickpea-flax	Pea-canola	Lentil-barley	W wheat-soybean	Lentil-canola	(t CO_2e per ha per yr)	(t CO_2e per ha per yr)	(t CO_2e per ha per yr)
AB	15125	1543285	285757	0	1934	-0.55	0.030	-0.002
MB	0	159554	12633	108752	0	-0.62	0.045	-0.002
SK	108830	1713345	1099960	0	1409346	-0.40	0.030	-0.002

*based on pea-canola (*Table 7*)

Table 9 presents a preliminary estimate of the potential GHG reductions from intercropping. This analysis uses reduction per ha from Table 8 with the assumption that 20% of the area of annual crops legume-non legume intercrops and a reduction of 60 kg CO_2e/ha is possible from avoided manufacture and reduced GHG emissions from 10 kg N/ha lower use of fertilizer. The emission reduction is significant.

Table 9: Potential greenhouse gas emission reductions from 10% of annual crop area as intercrops

Province	Area (ha)	SOC change (t CO_2e/ha)	Additional mechanical operations (t CO_2e/ha)	Seed and pesticides (t CO2e/ha)	N_2O emission* (t CO_2e/ha)	Total (Mt CO_2e/yr)
AB	1 748, 000	-0.55	0.030	-0.002	-0.060	1.02
MB	816, 000	-0.62	0.045	-0.002	-0.060	0.54
SK	3 096 000	-0.40	0.030	-0.002	-0.060	1.34

*assumed average reduction for avoided fertilizer manufacture plus reduced N_2O emissions

Many other intercrops are possible including mustard in place of canola, soybean in place of chickpea, and oat or spring wheat in place of barley. Fava bean-spring wheat and dry bean-spring wheat intercrops have been found successful in lower mainland of BC (Chapagain and Riseman, 2014b). A global literature synthesis has shown that fava-bean-cereal intercrops are particularly effective for suppressing cereal diseases (Zhang *et al.*, 2019). Intercrops with non-legumes crops are possible although less researched. Of course, intercrops with non-legumes would not have the N benefit of the legume crop to the non-legume crop.

Assuming that one-half the potential area in *Table 9* is intercrops with the listed emission reductions plus an N_2O emission reduction of 0.050 t CO_2e/ha, the total potential emission reductions are 0.53, 1.23, and 0.08 Mt CO_2e per year for Alberta, Saskatchewan, and Manitoba, respectively. Consequently, there is significant potential emission reductions from the adoption of intercrops. Because the overall production would be increased, the carbon footprint of grain would be decreased more than just indicated by those emission reductions. The lower footprint could give crops produced with intercropping a competitive advantage in markets that value that attribute such as a feedstock for biofuel.

Barriers to adoption

Seeding: Seeding mixed grains at the same time in the same row is not difficult with modern air seeders. However, in having different grains in different rows, that can provide better performance than mixed crops in the same row (Chapagain and Riseman, 2014a), would require modifications to seeding equipment. Seeding is a busy time so making modifications to seeding equipment between different single crop and different intercrops would be a barrier. The seeding rates for each crop affects the productivity of the mixture but the optimal seeding rates by crop and cultivar are not established. Seed costs will be higher for the less competitive crop in the mixture compared to having the same production of that crop done over a smaller total area as a single crop.

In-season crop management: Intercrops will have limited options for herbicides so some weeds may be difficult to control. Agronomic information on thresholds for control of various pest and diseases are not established for intercrops. There is insufficient research on fertilizing intercrops and they are not based on strong evidence.

Crop rotation and disease break: Intercrops can shorten the interval of the same crop types being grown. Recommendations for interval vary with crop. For example, pea and/or lentil should not be grown any more frequently than once every 6 years if root-rot causing *Aphanomyces* is present in the soil (Manitoba Pulse and Soybean Growers, 2021). This could limit the frequency of intercrops with those species. *Sclerotinia* stem rot can be carried by all prairie broadleaf crops so, while cereal single crop in rotation would provide a disease break from that disease, a cereal-broadleaf intercrops would not.

Harvest: Managing harvest so that all grain is in good condition while minimizing harvest losses is difficult. There is risk of shattering and/or weathering losses if harvest of one crop is delayed because of the other crop. If harvest of one crop is earlier than optimal to meet the harvest needs of the other crop, then there may be quality and/or quantity loss.

Post-harvest: Separating grains is a major barrier. It is an additional operation required during harvest that is typically the most labour demanding season in prairie crop production. Equipment for conventional seed cleaning is designed to separate a relatively small amount of material from one grain. When applied to an intercrop grain mixture, it has low throughput because it is not designed to separate a large amount of material from one grain. So, the equipment available for separating adds to the barrier.

Grain value downgrade: Suboptimal harvest timing that lower grain grade and/or excessive contamination of the grains with the other grain will lower the quality and, therefore, the market value of the grain. Where the grain production in contracted, there can be significant penalties if the grain does not meet the minimum quality required by the contract.

Social pressure: The surveyed experts noted that intercropping does not conform to normal cropping practices that is a barrier for adoption for some farmers.

Riskiness: Evidence shows that there is potential of lower overall yields with intercropping in some situations. Depending on growing conditions and seeding rates, the less-competitive crop in the intercrop can be overwhelmed by the more competitive crop so that the intercrop becomes equivalent to a weedy single crop of the more competitive crop.

The surveyed experts stated that some sort of external incentive would be beneficial to increased adoption to de-risk the transition to intercropping.

Co-benefits

Pod height

When grown as a single crop, the pods of the pulses can be very close to the ground. This makes harvest more difficult and increases the chance that harvested pulses comes in contact with soil during harvest that downgrades the grain quality. Special harvest equipment such as floating cutter bar can be needed and/or the land has to be rolled after seeding to level the ground surface to make harvest easier. However, when grown in an intercrop, the legumes grow taller because of the competition with the taller non-legume. This raises the pod height. In one study in Manitoba, as a single crop all pea pods were within 15 cm of the ground surface but, when growing in an intercrop with canola, all pods were at least 15 cm above the ground (Manitoba Pulse and Soybean Growers, 2015). Reducing the cutting height of the combine header for the pulses will result in more non-legume straw to thresh through the combine than it would as a non-legume monocrop, and this may increase energy consumption and overall diesel fuel use in the combine.

Increases suitable area for pulse crops

For legumes that require drying conditions to maximize grain yields, such as chickpea and lentil, then the water extraction by the non-legume intercrop can increase the legume grain yield (Dowling *et al.*, 2021) and expand the area suitable for those crops beyond the drier regions of the Prairies.

Reduced pesticide use

Intercrops have been found to require less pesticide use. This reduces the amount for off-field loss to broader environment. The lower use also reduces harm to insects and other animals that contact the intercrop.

Biodiversity

Some wildlife may prefer the more complex canopy and vegetation of an intercrop to the relatively uniform canopy of a single crop. The reduction in pesticide use can also provide an environment that foster a more varied and vigorous population of small fauna.

Inherently, the intercrop has more crop species at one time. Less use of herbicides can allow small populations of various weeds to exist that would not exist in single crop production that uses more herbicides.

Weather resilience

When the weather is adverse, one crop will likely do better than the other crop in the intercrop and thereby help stabilize grain yield.

Trade-offs

Economic performance – co-benefit or trade-off?

The gross returns depend on the price for each crop and its production in the intercrop. The overall grain production increases for well-managed intercrops but the amount of each crop produced depends on management of the intercrop and weather. Therefore, it can be difficult to maximize production of the crop that is expected to be the most profitable in the intercrop. There are extra costs for separating grains or, for relay crops like soybean after winter wheat, an extra harvest operation. There are also extra machinery costs for separation and, potentially, for seeding. Additional labour may be required to separate grains after harvest.

Potentially reduced pesticide use and, for legume-non-legume intercrops, reduced fertilizer N application will reduce input costs. When there is good agronomic information so that optimized management of intercrops becomes typical, it is expected that intercrops should have higher net returns over the long term because of higher production. However, in the short term, especially during the transition from single crop production, the risk-return balance for intercrops may not be perceived to be as favourable as single-crop production.

Increased complexity for managing pest and diseases when combined with cover crops and increased grain legumes

Complete breaks in time between production of one species serves to reduce the carryover of pests and diseases. Legume-non-legume intercrops increase the area of legumes and if combined with other efforts to increase grain legumes, managing legume diseases and insect pests across crop rotations could become difficult. Cover crops that contain legumes could complicate the pests and disease management. Cover crops with diverse mixtures of species including legumes and non-legumes could also complicate the management of diseases and pests for both the legume and non-legume grain crops.

Knowledge gaps

Agronomic

Many of the limitation for adoption can be characterized as knowledge gaps. Generally, there is limited knowledge to develop optimized management to obtain desired production outcomes for intercrops across the prairies. This produces uncertainty regarding the risk and potential returns from adopting intercrops.

A comprehensive program of research is warranted to determine the effect of specific agronomic management on intercrop performance. The performance of different intercrops or relay crops needs to be more thoroughly investigated. Optimizing the inter-related management of seeding rates, seeding configuration (mixed seed rows or number of alternating rows, etc.) and plant nutrition is important to improve performance and provide a desirable grain output between intercrops. Optimizing management of pests, diseases and weeds both during the intercrop year and across years. The data produced from the research program on costs, returns, and risks will give farmers valuable information for adoption decisions. All this research must produce data relevant to specific soils and climates across the Prairies so farmers can make better informed management decisions for their situation.

The guidance on best use and placement of different plant species over time in a crop rotation that include intercrops and potentially also include cover crops and increased legume crop generally need to be developed to manage crop insect pests, plant diseases, and unwanted plants (weeds).

There is a need for research on how crop breeding for intercrops can improve productivity and seed quality. For example, varieties could be developed for intercrops to mature at closer timeframes to avoid shattering at harvest.

Engineering

There are engineering challenges regarding cost-effective grains separation. If special row configurations are needed the general seeding equipment to produce those are not commercially available. Low-cost high throughput grain separation technology would help reduce one challenge in the harvest operation.

Environmental

The effect of intercrops on soil health needs to be investigated to understand the benefits of intercrops.

Greenhouse gas emissions

Given the potential benefit from SOC increases from intercropping, field research on the effect of productive intercropping systems on SOC is warranted.

There is a need for information on the optimal fertilizer N requirements and the N_2O emissions from various intercrops.

Pollution of water

The impact of intercropping on pesticide and nutrient contamination of surface and ground water needs to be investigated to know the effect of intercrops.

Biodiversity

The effect of intercropping on soil biota and the above-ground animals needs investigation to quantify the potential benefits.

Market services

Grain production statistics that are important for well-functioning markets will need to be fundamentally modified in terms of grain production prediction and the reporting the area of different types of intercrops and the final production of individual grains from intercrops.

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4. Increased legume crops

Description

Increasing grain legumes at the expense of non-legumes is an effective way to reduce GHG emissions. Grain legumes most appropriate for Canada include soybean (*Glycine max* (L.) merr.) and the pulse crops of pea (*Pisum sativa* L.), lentil (*Lens culinaris* Medik), chickpea (*Cicer arietinum* L.), dry bean ((*Phaseolus vulgaris* L.) of several types (pinto, kidney, navy, white), and fava bean (*Vicia fava* L.). The GHG emission reductions are primarily due to lower N_2O emission from greatly reduced use of N fertilizer in the grain legume crop year compared to that used for a non-legume crop. Because it's incorporated directly into plant tissues, the atmospheric N captured by the legume from symbiotic fixation produces no direct N_2O emissions. In contrast, there are direct and indirect N_2O emission from the N added needed to support the growth non-legume crops.

Effect of Legume Crops on GHG emissions

Soil Carbon

Generally, research on the Prairies has shown that SOC positively responds to C input for rotations with or without legumes (Shrestha *et al.*, 2013; Congreves *et al.*, 2015; Maillard *et al.*, 2018; Fan *et al.*, 2020). The amount of C input depends on the type of crop and the yield, but generally, grain legumes produce lower C input than other important non-legume crops such as wheat and canola. Therefore, rotations with frequent grain legumes have been shown to have less SOC than rotations without grain legumes (Maillard *et al.*, 2018) although this is not always true (Congreves *et al.*, 2015; Liu et al., 2020). Hence, the impact of grain legumes on SOC needs further research. Therefore, the effect of more frequent grain legumes on SOC was not included in the analysis.

N_2O and other emissions

Drever et al. (2021) estimated the mitigation potential of increased use of grain legumes. They developed a carbon budget that included direct and indirect N_2O emissions including and the emissions for fertilizer manufacture Drever et al. (2021) assumed an N credit of 10 kg N/ha that reduced the fertilizer N requirement of non-legumes following legumes.

Potential Impact of Increased Grain Legumes on GHG Emissions

Current Adoption

Crop statistics for 2018 from Canada's national GHG inventory (provided by Environment and Climate Change Canada)) showed 6.46 Mha of grain legumes on the Prairies, representing 22% of the annual crop area in Canada. Alberta has the least grain legumes as a proportion of annual crops at 15.8% while both Manitoba and Saskatchewan have 24.9% of annual crop being grain legumes. The predominant grain legume in Manitoba is soybean while it is pea in Alberta. Saskatchewan has a more even mix of all grain legume types. Owing to its small area historically, there were no statistics for fava bean.

Potential Adoption and total GHG benefits

For the 36 ecoregions on the Prairies with agriculture, Drever et al. (2021) analyzed the crop inventory (Government of Canada, 2020) for 2015 to 2018 for the area and crops in fields with and without grain legumes (soybean, dry bean, pea, lentil, chickpea). The fields with legumes were assumed to represent the mix of crops with legumes that is agronomically and economically feasible. To keep the proportion of legumes consistent with the assumption of feasibility, grain legumes were substituted for non-legume crops in field area without legumes to the average proportion of grain legumes of the field area with grain legumes in that ecoregion. For example, if 100 000 ha in an ecoregion had no legumes while 300 000 ha has an average proportion of legumes of 27%, then 27% of the 100 000 ha without legumes would have legumes substituted for non-legumes. The substitution of a specific grain legume for a specific non-legume crop reflected the proportion of crops grown in the ecoregion. For example if pea was the dominant grain legume and wheat the dominant non-legume, pea would be substituted for wheat. Three different substitution pairs were developed for each ecoregion to cover the major legumes and non-legumes in each ecoregion. The prairie grain legume area was increased by 1.71 Mha by this substitution (Table 10). The proportion of annual crop area in grain legumes after substitution would be 28%.

Because the potential adoption was based only on crops in a 4-year window, it can overestimate the proportion of legumes on fields that had legumes during that period. To illustrate, if grain legumes were grown once every 3 years in a rotation, then, over a 4-yr window, 50% of fields with that rotation would show grain legumes grown once every 2 years and 50% of fields with that rotation once every 4 years. The average proportion of legumes would be 37.5% rather the true value of 33.3%. If the frequency of legume is lower than one year in four, for fields in those longer rotations for which a legume appears during the 4-year window, the apparent proportion of legumes for fields with legumes will be 25% and so higher than the actual proportion. When a legume does not appear during the window that land is assumed to require legume addition. For these reasons, the rate of potential adoption may be a high compared to what farmers with legumes are finding currently feasible. Using a longer window would provide better statistics on the frequency of grain legumes in rotations and thereby provide better information as to the potential for increasing legumes. Also, using a finer analytical area than ecoregion, such as ecodistrict, would better identify areas that appear to be better or worse suited to grain legumes based on current production.

Table 10: Area of grain legumes and potential area for increased grain legumes and the potential total emission reductions in the Prairies.

2019 Area (Mha)	Additional grain legume area (Mha)	GHG reductions for substituting a grain legume for non-legume (t CO_2e/ha)	Total GHG reduction (Mt CO_2e)
6.46	1.71	0.427	0.728

Barriers to adoption

One barrier to increasing the area under cultivation by grain legumes relates to increasing the risk of plant disease due to shorter period between grain legumes. Increased genetic plant disease resistance would reduce this barrier.

Grain legumes have relatively high production costs. They have a high seeding cost due to their large seed size. They may require fungicides.

The price of pulses is sensitive to the grain quality. Weather related downgrading of grain quality is costly.

Lentil and chickpea production requires a plant stress signal, typically from drying conditions to maximize seed set. Hence the area for adoption of these crops is currently limited to the drier areas of the prairies, principally the Brown and Dark Brown soil zones.

Co-benefits

The major co-benefit will be reduced N fertilizer use and thereby lower losses of N to the environment from volatilization, runoff, and/or leaching, as well as lowered input costs for fertilizer N for producers.

Grain legumes production reduces the impact of fertilizer price increases.

Production of grain legumes reduces the carbon footprint of the whole crop rotation including that for crops grown after grain legumes (MacWillian et al. 2018). Hence increasing grain legume can increase the value of other crops in markets that value low carbon footprints.

Trade-offs

Owing to their grain being close the ground, harvesting grain legumes is typically slower than for cereals and canola (MacWilliam et al., 2018)

Since Canada is the largest supplier for the world's international trade of peas and lentils, large production increases may lower their price in Canada and thereby the relative profitability of their domestic production.

Knowledge gaps

Optimizing integrated plant disease management including better genetic disease resistance is important for pulse crops. Fava bean is not a major grain legume crop yet on the Prairies so there is still need to develop best management practices for the crop and to expand markets. Improving grain legume harvestability remains important.

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5. Reduced field burning of crop residues

Description

Burning of crop residues is to remove residues to ease seeding. Generally, the reasons residues will cause problems with seeding because of their particular situation (matted, piles that collected due to wind or runoff, windrows that were not baled), amount exceeds what can easily be handled with preferred tillage or seeding implements, or toughness (flax). Sometimes, expediency to remove residue is the reason when time or wetness prevents tillage.

Another reason for burning residue is to reduce disease in subsequent crops. However, in Saskatchewan study showed the burning is ineffective for disease suppression in barley or canola (Kutcher and Malhi, 2010). Some crop residue is burned each year in accidental fires. The amount of burning has steadily decreased over time (Table 11).

Table 11. Percent of crop residue burned per year (ECCC, 2022).

Province	1991	1996	2001	2006
AB	0.8	0.7	0.2	0.2
MB	12.6	10.1	8.9	2.3
SK	8.1	5.8	3.9	1.5

Effect of burning on GHG emissions

SOC

Burning residues reduces the C input to the soil and so reduces SOC (World Bank, 2012). However, the effect will the completeness of burning that is determined by the burn conditions, so general rates of SOC reduction are not available for Canada or elsewhere.

Other emissions

The CO_2 released from burning is not considered an GHG emission since it derived from CO_2 that was recently removed from the atmosphere. The major emissions are CH_4 and N_2O during burning. ECCC (2022) uses the Tier emission factors from the IPCC (IPCC, 2006).

Potential impact of residue burning on GHG mitigation

Current Adoption

The Canadian national inventory (ECCC, 2022) report provides estimates of residue burning for 2006 and that value is assumed to be valid currently. Given that residue burning had been decreasing before 2006 could indicate that estimate for currently could be an overestimate. Table 12 summarizes the estimated extent of residue burning for the Prairies.

Table 12: Estimated residue burning (% of crop area) by province (ECCC, 2022).

Province	Spring Wheat	Winter Wheat	Oats	Barley	Mixed Grains	Flaxseed	Canola
AB	0	0	0	0	0	8	0

46

MB	2	3	3	1	0	17	1
SK	0	0	0	0	0	15	1

Potential Adoption and GHG impacts

Since the majority of growers manage without burning there is potential to reduce planned crop residue burning completely. Flax is clearly the residue where the most effort is needed. Creating markets for flax straw would help or some GHG-based incentive for not burning. Table 13 gives the potential GHG emission reductions if all field burning were stopped.

Table 13: Potential emissions reduction from eliminated field burning of crop residue in 2020 (ECCC, 2022)

Province	Emission Mt CO_2e/yr
AB	.020
MB	.001
SK	.030

Co-Benefits

The particulate matter in smoke from burning crop residues is an important human health concern so stopping burning has an important and very visible co-benefit. Smoke form fires can also cause visibility problems on roads so is also a safety concern.

Trade-Offs

None.

Knowledge gaps

None.

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6. Improved N management

Description

In response to concerns relating to potential negative environmental of nutrients, particularly synthetic fertilizers, the fertilizer industry developed the 4R Nutrient Stewardship™ Program to promote improved fertilizer management. The 4R nutrient stewardship program refers to four key practices in nutrient management: 1) right source – choose plant-available nutrient forms that provide needed nutrients with release matched to crop demand, 2) right rate – ensure adequate, but not excessive, amounts of all limiting nutrients are applied to meet plant requirements in relation to yield and quality goals, 3) right time – time nutrient applications considering the interactions of crop uptake, soil supply, environmental risks, and field operation logistics, and 4) right place – place nutrients to take advantage of the root-soil dynamics, spatial variability within the field, and potential to minimize nutrient losses from the field (Reetz et al., 2015).

The right source, right time, and right place practices refer to changing the N_2O emissions per unit of nitrogen (N) applied to the soil as fertilizer, amendment, and crop residues, (i.e., a change to the emission factor). The right rate practice refers to changing the N applied that changes the N_2O emissions without changing the emission factor. If one of the 4R practices is very suboptimal, then addressing that factor alone can have large impacts on N_2O emissions. However, as the N management practice becomes optimal, it is necessary to consider all the 4R practices as a system. In fact, Maaz et al. (2021) noted that in a global meta-analysis, it was difficult to impossible to discern the impact of individual 4R practices on N_2O emission. However, when multiple N management practices are considered together including in the 4R context, the reduction in N_2O emissions becomes more obvious (Eagle et al. 2017; Young et al. 2021; Venterea et al., 2016).

Right source refers not only to the form of the nutrient but also the use of enhanced efficiency fertilizers that control the release of N to avoid periods when the crop requirement is low or chance of loss is high. Inhibitors slow N transformations of urea hydrolysis or nitrification whereas controlled release fertilizers slow the decomposition of urea through coatings that reduce dissolution. These products alone can reduce N_2O emissions by up to 50% but can also increase emissions in some circumstances (Burton, 2018).

Right rate refers to matching rate of N application to the crop needs. This matching needs to be done on a whole field basis to match N application to realistic yield goals. It can also involve precision farming techniques to consider the productivity- N relationships of subregions in the field based on characteristics such as soil organic matter, landform effects on soil moisture, soil texture, pH, salinity, and soil structure that affect those relationships. It also aims to reduce overapplying fertilizer by avoiding overlap on the field during N application.

Right time refers to timing of application including split application during the crop growing season. Spring application of mineral N fertilizer near or at seeding generally reduces N_2O emission but fall application may have similar (lower or higher) emissions providing the soil is 10°C or less when applied. On the Prairies, split application can reduce N_2O emissions when moist soil conditions from rainfall or irrigation would cause high losses in the early growing season before the split (Burton, 2018). An exception is irrigated potato production where split application at hilling reduces N_2O emissions. A general recommendation is that at least 1/3 of fertilizer applied in the growth period will generally reduce N_2O emissions.

Right place refers to soil location of application. On the Prairies, banding fertilizer in the fall reduces N_2O emissions compared to broadcast application. Although there may be better agronomic performance with banding in the spring, the effect of spring banding has mixed effects on N_2O emissions (Burton, 2018). In addition, dribble banding of urea ammonium nitrate (UAN) should be avoided (Burton, 2018). Ammonia volatilization that occurs at the time of fertilizer application causes indirect N_2O emissions downwind, can be reduced with proper placement in the soil. Therefore, broadcasting urea on the surface without incorporation should also be avoided to consider volatilization.

Table 14 summarizes the key features of different 4R practices. Importantly, although this discussion is confined to N_2O emissions from improved N management, a fundamental requirement for all levels of 4R practice is that all nutrients are included within 4R practice. Therefore, the assumption is that all other nutrients such as phosphorus (P), potassium (K) and sulfur (S) are managed well so they do not impede the agronomic or environmental performance of 4R practices for N.

Table 14: General definition of 4R implementation level.

	4R Implementation Level		
4R Practice	**Basic**	**Intermediate**	**Advanced**
Right Rate	N rate based on target crop removal and N status of soil, manure N estimated, based on individual field	Basic + sub-field zones based on land characteristics.	Intermediate + sub-field application based on in-depth field analysis, in-season crop monitoring, regular re-evaluation based on data.
Right Source	Ammonium-based fertilizer	Basic + enhanced efficiency fertilizers for at least 1/3 of the N used.	Intermediate + enhanced efficiency fertilizer for at least 1/2 the N used.
Right Time	Fertilizer applied in spring (fall when soil cool in prairies), split N for potato and corn, no application on snow or frozen soil	Basic + multiple fertigation (irrigated)	Same as Intermediate
Right Place	Placed in soil, no more than 1/3 on surface, sideband at seeding	No surface application unless incorporated with 1 day or with enhanced efficiency fertilizers	Same as Intermediate

Impact on improved N management on GHG

SOC

SOC will be affected by the balance between the amount of C input from the crops and SOC decomposition. The assumption is that improved N management does not substantially affect either part of the balance so SOC will not be affected by reduced N_2O emissions.

N_2O

For each level of 4R practice, there are direct reductions in N_2O emissions from fertilizer and manure by a reduction modifier based on the expert opinion of 12 Canadian scientists convened by Fertilizer Canada in 2018 (Fertilizer Canada, 2018) (Table 15).

Table 15: Definition of 4R practices constituting basic, intermediate and advanced implementation of 4R: EEF = enhanced efficiency fertilizer, RM = N_2O emission factor reduction modifier.

	Basic	**Intermediate**	**Advanced**
Manure	Follow provincial rate recommendations RM = 0.85	Periodic nutrient analysis as spread RM = 0.85	Regular nutrient analysis as spread RM = 0.85
Non-legume	Follow provincial rate recommendations RM = 0.85	Use of EEF on 1/3 of field RM = 0.75	Use of EEF on half of field, use of variable rate applicators RM = 0.65
Potato (irrigated)	Follow provincial rate recommendations RM = 0.85	Use of EEF on 1/3 of field RM = 0.80	Use of EEF on half of field, use of variable rate applicators RM = 0.75

Other emissions

Drever et al. (2021) and Burton et al. (2021) assumed that N rate application could be reduced by 10% for intermediate 4R practice and 20% for advanced 4R practice. The estimated N use efficiency for the intermediate and advanced N rates were 76% and 85%, respectively (Drever et al., 2021). The rate reduction also decreased indirect N_2O emissions since it reduced the amount of volatilization and leaching/runoff losses of N. There are significant embodied GHG emissions in N fertilizer from its manufacture so reductions in N use will also reduce these embodied emissions. reductions will also occur from less fertilizer use from re also observed from embodied N when rate was reduced due to reduction in fertilizer manufacturing.

Potential impact of practice on Prairie GHG emission

Current adoption

Drever et al. (2021) estimated the 2017 adoption rates as 50% of basic 4R practice, 8% of the intermediate practice, and 2% of the advanced practice of nutrient management across Canada. Burton et al. (2021) considered only 4R adoption for corn, spring wheat (and durum), winter wheat, canola, and potato. Their adoption rates are given in Table 16.

Table 16: Estimated baseline (2017) adoption rates for basic, intermediate, and advanced 4R management used in modelling (Burton et al. 2021).

Crop	***Basic***	***Intermediate***	***Advanced***
Corn	27	22	11
Winter Wheat	30	20	10
Canola	45	12	6
Spring Wheat	30	20	10
Potato (irrigated)	30	20	10

Potential Adoption

Burton et al. (2021) estimated maximum adoption by 2030 based on incentive programming was 70% of fertilizer under 4R practices due to higher costs associated with advanced implementation, requiring more equipment and technology (70% total, delineated by 10% Basic; 10% Intermediate; 50% Advanced; 30% of nitrogen use remains not under 4R management). The impact on GHG emissions of potential adoption compared to current (2017) adoption is given in Table 17.

Table 17: Potential emission reductions from 2017 for the Prairies by 2030 with strong programming (Burton et al., 2021).

Crop	***GHG reduction (Mt CO_2e/yr)***
Corn	0.020
Winter wheat	0.008
Spring wheat	0.295
Canola	0.386
Potato	0.002
Total	**0.711**

Drever et al. (2021) estimated potential adoption in 2030 for all non-legume crops based on a $100/ t$CO_2$e value for CO_2e reductions. The impact on GHG emissions of potential adoption compared to current (2017) adoption is given in Table 18.

Table 18: Total area of adoption in 2030 and potential emission reductions for the Prairies from 2017 by 2030 with $100/t CO_2e (Drever et al. 2021).

Province	***Area of no 4R***	***Area of basic 4R***	***Area of intermediate 4R***	***Area of advanced 4R***	***Emission Reduction (Mt CO_2e/yr)***

	('000 ha)	('000 ha)	('000 ha)	('000 ha)	
Alberta	348	1,513	1,374	3,727	1.42
Manitoba	150	451	25	2,378	1.14
Saskatchewan	574	2,323	2,322	6,249	2.22
Total	**1,072**	**4,287**	**3,721**	**12,354**	**4.78**

Drever et al. (2021) reports much larger emission reductions than Burton et al. (2021) because the former analysis assumes more total crop area for adoption and much higher adoption of advanced 4R. Burton et al. (2021) adjusts the incentives to reach 4R practice area level targets and incentives based directly on the GHG reductions in the Drever et al. (2021) analysis. Therefore, because it has higher N_2O emissions due to its generally wetter climate, Manitoba is particularly induced by GHG-driven incentive ($100/ tCO_2e) to potentially adopt advanced 4R practice. Whereas there would still be significant basic and intermediate 4R practice area in generally drier Alberta and Saskatchewan with the same GHG-based incentive of $100/ tCO_2e.

Barriers to adoption

Costs for soil testing is a barrier of adoption of all 4R practices. The cost of enhanced efficiency fertilizers is a barrier for adoption of intermediate and advanced 4R. The cost for set-up and management of precision farming is a barrier to advanced 4R management. In addition, a high level of management, particularly at the advanced level of 4R practice, is required.

Many growers may apply a high level of N fertilizer to ensure yield potential is reached if the weather is particularly favourable or conditions cause appreciable N losses. Thus, they may not see any yield increase from 4R application and so not see 4R as profitable when N fertilizer prices are low relative to crop prices.

Although growers are motivated to be good environmental stewards (Amiro et al., 2017), the producers' impact on N_2O emission is invisible to them. N_2O losses are a very small part of the N budget so they do not directly benefit from undertaking management to reduce N2O emissions. Burton et al. (2021) proposed that subsidized testing of residual mineral N after harvest is a good indicator of successful 4R practice and so becomes a tangible and meaningful measure to producers of their accomplishment.

Co-benefits

The 4R practices were initially designed primarily to reduce N losses from volatilization, and runoff and leaching to surface and ground waters. Therefore, the reduced environmental harm from off-field N losses is a major co-benefit. Advanced 4R practices with precision farming may identify areas within the farm where there can be large saving in nutrient application without impacting crop yield or quality and also those areas where crop yield can be increased from additional nutrient application compared with historical non-4R nutrient management. Therefore, they can be important yield increases that increase revenue from crop production.

Trade-offs

Practicing 4R will increase the cost of nutrient management. The increased efficiency of crop N use should offset the moderate cost increases at the basic level of 4R adoption. The increased costs are most pronounced at the intermediate and, especially, advanced level, partly due to the greater proportion of N applied with enhanced efficiency fertilizers. Therefore, increases crop N use efficiency may not offset the increased cost of nutrient management for intermediate and advanced 4R practice when management when weather and other factors like pests and diseases impact the effectiveness of 4R practice. Therefore, particularly when crop prices are relatively low compared with total production costs, 4R adoption at intermediate and, especially, at the advanced level could reduce net income from crop production in some years.

Knowledge Gaps

Research is needed to reduce the uncertainties regarding the expected long-term emission reduction that can be expected for different levels of 4R practice across the soils, crops, and weather conditions of the Prairies.

Given the higher price for enhanced efficiency N fertilizers, better knowledge of how to optimize their use for various soil, crop, and climate combinations to reduce N2O emissions while not lowering crop production is needed.

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7. Biochar Addition to Soil

Description

Biochar is solid material obtained from the thermochemical conversion (i.e., pyrolysis) of biomass in an oxygen-limited environment. It is stable and long-lived in the soil. It is used as a soil amendment and to increase sequestered carbon in the soil.

The physical and chemical properties or biochar differ with the type of feedstock and the conditions during pyrolysis. Further, the rates of application and whether they are applied with other amendments such as manure or mineral fertilizers also impact its effects. Consequently, it is difficult to generalize effects on soils and vegetation grown in biochar amended soils (Lévesque *et al.*, 2022).

A key physical feature of most types of biochar is their highly porous structure and large surface area. This affects many physical, chemical, and biological processes in the soil.

General crop yield increases from application of biochar depend on the quality and fertility of the soil. In the Prairies, crop yield response to a biochar-manure mix was greater in the lower fertility soils in the Brown soil zone compared to higher fertilizer soils in the Black soils zone (Hangs *et al.*, 2021). The opposite effect was observed for these soils for production under the same controlled conditions. However, with a different biochar and study, the positive effect in the Brown soil zone was not observed (Alotaibi and Schoenau, 2016). Compared to tropical soils, the rates of biochar required to affect crop yields in temperate soils are too high to be economically feasible (Lévesque *et al.*, 2022). Abedin and Unc (2021) found that 10 Mg/ha of biochar only had a short-term benefit to improving soil properties for agricultural production in low-fertility podzols recently cleared of boreal forest cover.

Biochar is carbon-based and, depending on its properties, can increase C mineralization in Prairie soils (Weber *et al.*, 2022). Overall, Gross *et al.* (2022) found biochar was very effective for increasing SOC and concluded that the main rationale for biochar addition in the prairies should be for carbon sequestration purposes for GHG emission mitigation.

Effect of Biochar on GHG emissions

SOC

Drever et al. (2021) estimated the long-term sequestration of carbon in soils from biochar is 0.18 Mg C/Mg of biomass, or 0.66 Mg CO_2e/Mg of residue. They also accounted for the emissions associated with additional N fertilizer and residue harvest required to replace nutrients and collect the residue, as well as loss of SOC from reduced residue inputs, as described under the bioethanol pathway. The net benefit of biochar application to the soil was thus estimated as 0.27 Mg CO_2e/Mg of residue.

N_2O and other emissions

Worldwide, reduction in N_2O emissions from biochar from by about ½ has been found in meta-analyses (Cayuela *et al.*, 2014; Lyu *et al.*, 2022). The N_2O emission reductions have been related to greater absorption of ammonium and nitrate and so reduced nitrification and/or denitrification (Lyu *et al.*, 2022). Hangs *et al.* (2016) reported similar reduction in N_2O emission with biochar addition in N-fertilized situation for soils in both Black and Brown soil zones. However, other studies have observed no effect on N_2O emissions in the Prairies (Romero *et al.*, 2021a; Romero *et al.*, 2021b; Gross *et al.*, 2022)

Hangs *et al.* (2016) also observed higher CH_4 uptake with biochar. Worldwide, the effect of biochar on CH_4 emissions and uptake in upland soils has been variable.

Potential impact of biochar on Prairie GHG emissions

Current adoption

Current adoption is considered negligible.

Potential Adoption and GHG impact

Drever et al. (2021) used the same residue availability as they described for the bioethanol BMP. They considered small road-transportable biochar production facilities that would be moved around the residue supply area. The biochar produced would then be spread locally on the fields from which the crop residue was removed. The grower was paid for residue harvest and N fertilizer to replace that removed in grain. However, they assumed the value of the agronomic benefits of biochar were sufficient to cover the biochar production and biochar application. Table 19 provides their estimates of potential GHG reduction with biochar.

Table 19: Technical potential GHG reduction from biochar in the Prairies from Drever et al. (2021).

Province	GHG reduction (Mt CO_2e/yr)
Alberta	1.52
Saskatchewan	2.16
Manitoba	0.85

Barriers to adoption

To be feasible, growers need to be confident that the there is a value proposition for removing residue and applying the biochar produced from that biochar on the land. Investment in mobile pyrolysis reactors will also require expectation of positive economic returns.

Co-benefits

There are new business and employment opportunities in rural Prairies for baling crop residues, transporting residues to local collection point for pyrolysis, operating pyrolysis businesses, and spreading biochar.

Trade-offs

The major trade-off is the soil health implications of residue harvest. On the prairies the effect on SOC has been negative (Lemke *et al.*, 2010; Smith *et al.*, 2013) and has lowered the more active fractions of soil organic matter (Malhi *et al.*, 2011c, a, b). Straw retention has been shown to have little effect of crop yield over short-term 4 years (Malhi and Lemke, 2007) and over some decadal-scale studies (Lemke *et al.*, 2010) but reduced grain yields by up to 10% over the decadal periods in another study (Malhi *et al.*, 2011d). When determining the sustainable rate of residue removal, if there is one, it is important to recommend using crop residue as a biomass feedstock for bioenergy as a BMP for GHG reductions.

It is uncertain what the benefits are of biochar application on Prairie soils.

Research Gaps

Better understanding of effect of feedstock type and pyrolysis conditions on the agronomic and soil impacts for Prairie soils is necessary to understand the full value of their application.

When applied for GHG reduction, better understanding of the type of biochar that reduces soil N_2O emission is essential to determining the potential benefit.

Research is needed into soil health and crop yields impacts of crop residue removal for the Prairies so farmers can make informed decisions about how much, if any, residue removal in which they will participate.

Other biomass sources including biomass herbaceous crops and short-rotation trees provide additional opportunities for biomass supply for biochar production from farmland in the Prairies (Liu *et al.*, 2014). These represent additional opportunities to augment crop residue and provide other new income opportunities for farmers where their farmland has areas better suited for such biomass production than for conventional agricultural uses.

Biochar can be a co-product of production of bioenergy (methane and/or liquid fuels) through thermochemical processing. That biochar effectively reduces the carbon footprint of biofuels (Fan *et al.*, 2021). Research is needed if biochar useful for improving Prairies soils including reducing N_2O emission can be produced as part of a thermochemical bioenergy pathway. If so, such bioenergy may prove to be an effective GHG reduction method.

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BMPs Pertaining to Livestock and Pasture Systems

This section covers BMPs that relate to the production of livestock and pasture systems, including the application of organic amendments, rotational grazing, and integrating perennial forages into crop rotations. These BMPs can increase rates of soil organic carbon, reduce the required rates (and associated emissions) of inorganic fertilizers, and increase plant and animal diversity on agricultural lands. These BMPs also contribute to improvements in soil health and can make for more resilient farming systems.

8. Increasing Organic Amendments Applied to Agricultural Lands

Description

Soil amendments can be broadly defined as any substance added to agricultural lands which changes the properties of the soil and include both organic and inorganic fertilizers, compost, manure, crop residues, liquid biological solutions, and many others. Organic amendments are considered those which come from living or biological processes, the most common types being livestock manure and compost. A non-exhaustive list of organic amendment types includes livestock manure, compost, microbial fertilizers, biochar, digestates, blood meal, biosolids, compost tea, and kelp meal.

Different amendments serve different purposes. The application of manure, compost, and biological fertilizers is done to replenish soil nutrients that are removed from the system in grains. Other amendments, such as compost tea, are intended to stimulate microorganisms which have a symbiotic relationship with plants (St.Martin & Brathwaite, 2012). However, all amendments are applied with the intention of improving the soil or production system. Composted materials rich with organic matter can also help to increase the soil organic matter levels, which come with many additional benefits (Frick et al., 2001).

The methods of application vary by amendment type and can pose considerable challenges to producers who manage large quantities of the material or apply across a broad area. For example, beef cattle manure has around 5kg of total nitrogen per ton of product, therefore potentially requiring rates of 57 tonnes per hectare to meet a crop's nitrogen demand for that year (PennState Extension, 2022). The logistical challenges and energy usage of applications at that scale are considerable. Often it is more economical to apply the amendment in conjunction with inorganic fertilizers, rather than as a total substitution. Amendments are frequently incorporated into the soil as well, requiring additional equipment or tillage passes in the field.

Current Adoption

There is little data available on the adoption rate of adding organic amendments to agricultural soils as a BMP since it is not tracked by the Census of Agriculture and there is no literature estimating its level of adoption. Livestock populations are tracked through Statistics Canada and can be used to estimate manure production on a regional basis. Due to estimates of livestock populations and the lack of data on non-manure organic amendments, we assume that manure is the most used organic amendment in the Canadian Prairies. Manure contains key nutrients needed in common Prairie crops like nitrogen, phosphorous, potassium, sulphur, and others. Most livestock manure is ultimately applied to agricultural soils already, with volumes being especially high in regions with abundant livestock. Some manure types, like those from beef and dairy cattle, are relatively high in phosphorous when applied at rates to meet a crop's nitrogen demand. This causes excess phosphorous levels within the soil, which build up but can also be lost from the agriculture system by erosion and run-off causing problems with nutrient loading in water bodies (Hooda et al., 2000). For this reason, the Prairie provinces have set regulations and guidelines on application timing and rates to mitigate this risk. While some regions would benefit greatly from an increase in available manure fertilizers, other regions have a manure surplus from a concentration of feedlots or other livestock operations.

Canada produces a significant amount of animal manure, estimated at around half a million tonnes per day or 180 million tonnes for the year of 2006 (Hoffman, 2008). The greatest quantity of manure comes from beef cows (38%). Milk cows, calves, and heifers each make up around 12% of manure production, with steers (10%), pigs (9%), and poultry (3%) covering the majority of the rest (Hoffman, 2008). Regions with the highest concentration of livestock are Alberta's southern and central regions, south-western Ontario, and Quebec's south-east according to a 2006 survey (AAFC, 2006). However southern Manitoba and Saskatchewan have high-density livestock clusters as well. Livestock populations have increased since 2006 by 3% across the Prairies, however, the composition of animals has changed (Statistics Canada, 2021). In 2021, the composition of livestock manure has changed due to 17% increase in hens and chicken populations, whereas all other major livestock (beef and dairy cattle, pigs, sheep and turkeys) have decreased by 11% on average (Statistics Canada, 2021).

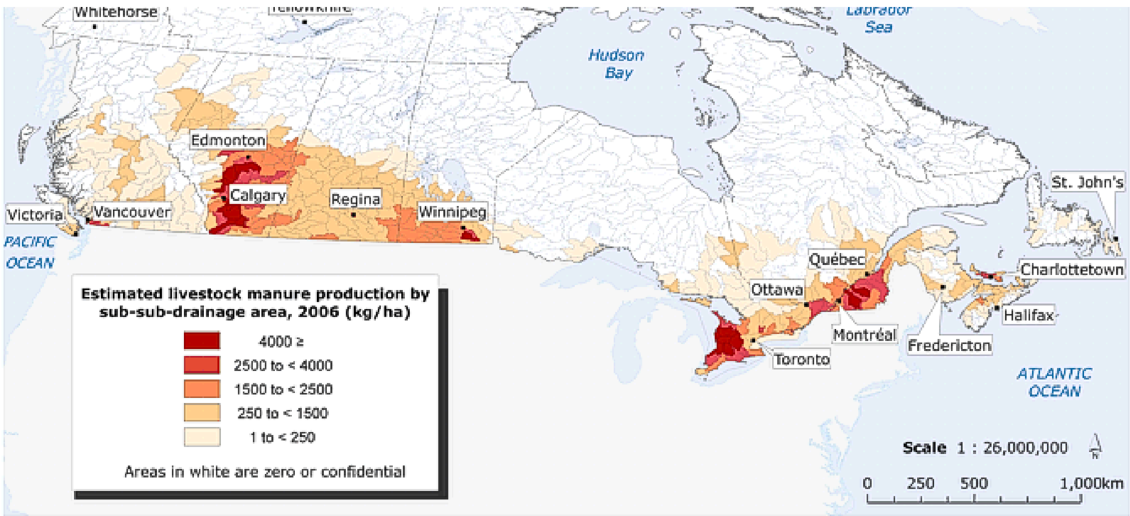

Figure 1. Distribution of livestock manure production in Canada. Source: Agriculture and Agri-Food Canada and Statistics Canada, Customized tabulations, Census of Agriculture, Census Geographic Component Base 2006.

The use of other organic amendments is much lower than manure. While there are limited data on adoption levels of organic fertilizers and other soil amendments on the Prairies, there is an increasing level of interest in organic amendments as an alternative to inorganic fertilizers in both the scientific and business communities. A company out of Brandon Manitoba, for example, is blending elemental sulphur with grocery food waste into a macro-nutrient fertilizer product (Bio Sul, 2022). This process diverts food waste from landfills and provides valuable nutrients and organic matter to agricultural soils in western Canada. In addition, novel biological products are being developed as fertilizer alternatives but the concept is new and not widely proven as a potential inorganic nitrogen replacement (Source, 2023).

Potential impact of practice on GHG emissions

There are several ways adding organic amendments to agricultural lands can affect a farm's GHG emissions. They can act as a source of N_2O, CH_4, and CO_2 when subjected to a soil's biological and chemical processes in decomposition (Cayuela et al., 2010). Total methane emissions from the Prairie provinces' animal waste sector equated to 1.42 Mt of CO_2e in 2004 with most of those emissions originating from liquid manure storage (Desjardins et al., 2006). In 2020, manure management was responsible for 2.2 Mt CO_2e in Manitoba, 1.8 Mt CO_2e in Saskatchewan, and 0.430 Mt CO_2e in Alberta (ECCC, 2022). The provincial differences suggest that regions with high concentrations of dairy and swine production (e.g., Manitoba) have high emissions because those livestock operations have liquid (anaerobic) manure storage in lagoons, whereas Alberta's predominantly beef manure is in solid (aerobic) storage.

Conversely, organic amendments used as fertilizers can produce comparatively lower emissions than synthetic fertilizer when considering all the soil dynamics and upstream emissions in fertilizer manufacturing. Still, there is significant variability in the emission factors (EF) from the different fertilizer types and organic amendments which makes GHG impacts challenging to estimate at a macro scale. For example, Walling and Vaneeckhaute (2020) reviewed the emission factors of various organic amendment types and found a wide range of uncertainty. Table 2020 outlines the EFs associated with compost production from different waste types per tonne of product.

Table 20: Compost emission factors from differing waste sources. Table recreated from Walling & Vaneeckhaute, (2020).

Waste Type	EF including CO_2 (kg CO_2e / tonnes of waste)	Source of EF
Hen carcasses and manure	45-82	Zhu et al (2014)
Dairy manure	145-173	Ahn et al (2011)
General	323	White et al (2012)
Grass and green waste	380	Hellebrand (1998)
Cattle manure	400	Hao et al (2004)
Garden and biowaste	46-942	Boldrin et al (2009)
Municipal waste	286-363	ADEME (2012)

Evaluating emissions per tonne of waste product is one way of assessing its GHG impact. However, organic amendments can be used as a substitute or in conjunction with inorganic fertilizers to provide required crop nutrients. Therefore, assessing the emissions per kg of nitrogen is in some cases more useful than comparisons by volume or weight. Nevertheless, direct comparisons between synthetic and organic amendments must account for factors such as the production energy source, storage type, application method, and climatic or soil influences on application. Even when controlling for those variables the emission factors for both organic and inorganic fertilizers can vary drastically, as shown in Table 2121.

Table 21: Emission factors from various studies comparing organic fertilizers with inorganic N fertilizers. Table recreated from Walling and Vaneeckhaute, (2020).

Emission factor for Synthetic N (kg CO_2 per kg N)	Emission factor for Manure or Slurry (kg CO_2 per kg N)	Emissions factor for Compost (kg CO_2 per kg N)	Emission factor for Digestate (kg CO_2 per kg N)	Source of EF
1.85	0.60 – 3.70	0.60	---	Akiyama et al, 2004
5.36	1.37-3.78	1.5	----	Lopez-Fern ́andez et al, 2007
9.66	6.88	0.33	----	Alluvione et al, 2010
2.44	1.07	1.82	0.72	Meijide et al, 2009
6.17	2.98	4.62	1.70	Vallejo et al, 2006
0.09-0.36	0.27-0.33	-----	0.15-0.30	Collins et al, 2011
0.51	0.48-1.07	-----	0.30	Baral et al, 2007
0.51/2.06*	0.89-3.67	-----	0.42/1.19*	Chantigny et al, 2007
1.8	1.19-1.79	-----	7.2-8.9	Saunders et al, 2012
2.1	11.0-13.4	-----	3.3-6.0	Lemke et al, 2012
2.7	7.7-14.3	-----	5.4	Bertora et al, 2008
4.62/26.61*	10.51/16.27*	-----	8.3-17.6	Chantigny et al, 2010
0.18-0.57	----	0.06-0.75	1.9-15.2	Verdi et al, 2018

**Slashes represent multiple emission factors produced from a study.*

Relative to emissions from other stages of organic amendment life cycles, the transportation emissions had a minimal effect on total emissions (Walling & Vaneeckhaute, 2020). A Manitoba study also found that the energy associated with transporting and spreading liquid pig manure was much lower than the energy cost of producing anhydrous ammonia or urea where the land on which it was applied was near the manure source (Wiens et al., 2008). The energy consumption per kg of available N from the pig manure was less than that of anhydrous ammonia up to 8.4 km away from its source, and 12.3 km for urea N (Wiens et al., 2008). However, the logistical costs of transporting large quantities of manure over large distances is a major limitation for distribution.

GHG emissions from storage is an important variable for some but not all amendments. Inorganic fertilizers emit negligible amounts of GHGs in storage because they are stored in a highly stable form, while organic amendments like compost or manure can emit large amounts of GHGs in storage, depending on the method, storage time, and composition (Walling & Vaneeckhaute, 2020). Estimates on CO_2, CH_4, and N_2O released during the storage of organic fertilizers vary widely in the literature, further complicating the accounting. Compost is the organic material least likely to emit GHGs in storage with digestate also being low (<0.01 kg CO_2e kg N^{-1} day^{-1}) by many estimates (Walling & Vaneeckhaute, 2020). Storage of manure can be a high-emitting process but varies depending on if the manure is treated (0.14 kg CO_2e kg N^{-1} day^{-1}) or untreated (0.33 kg CO_2e kg N^{-1} day^{-1}) (Amon et al., 2006).

Potential adoption of the various organic amendments is hard to predict, and the trends of usage are not monitored in contrast to the fertilizer industry. Given that, and the wide range of uncertainty and variability that exists between amendment types, GHG mitigation potential for the broad "organic amendment" category cannot be estimated. Individual amendment types such as using biological nitrogen fertilizers, compost, and manure optimization should be evaluated on their own for climate change mitigation.

Impact on Soil Carbon

Applying organic amendments to agricultural lands can be an effective way to increase soil carbon levels (Li et al., 2021). The levels and permanence of accumulated soil carbon resulting from the amendments will be determined by the amendment type and the many factors that influence the soil carbon dynamics. Li et al., (2021) found that farmyard manure applications, green manure crops and straw residues returned to the soil all contribute to an increase in soil organic carbon sequestration at different levels, but all were positive. This increase can be especially true for degraded soils (Janzen et al., 1998). The two-year carbon sequestration efficiency (ratio of C inputs to SOC sequestration) was estimated to be highest for farmyard manure at 55.9% on average (Li et al., 2021). In Alberta, two research sites with eroded soils tested SOC levels five years after a manure application of 75 Mg ha^{-1}. The researchers found an average increase of 12 Mg ha^{-1} of soil organic carbon in the sites that received the manure application compared to the unamended treatments (Izarurralde et al., 2018). However, much of the increases to soil organic carbon from manure is fundamentally a transfer of carbon from one agricultural system to another in a different location. Truly "additional" carbon removals from this BMP are those resulting from an increase in net-primary productivity caused by improvements to a soil's biophysical characteristics or carbon inputs that originated from a non-agricultural process (like composted food waste destined for a landfill).

Sub-BMP: Bale Grazing

Bale grazing is the action of feeding livestock bales on agricultural fields which allows animals to feed and recycle nutrients over a larger and less concentrated area. While the organic amendments (manure from livestock) would eventually be applied to lands whether in dry lot pen feeding or bale grazing on fields, this practice fits into this BMP because it can be used as an effective method for dispersing nutrients on a broader area of land. In North Dakota, bale grazing has been shown to be an effective method for winter-feeding which can improve the average daily gain (ADG) of cattle and reduce system costs compared to dry lot feeding (Undi and Sedivec, 2022). A Manitoba study looked at the effects of winter bale grazing on forage productivity and found that forage dry matter production was negatively impacted in the first year (68% decline) after grazing but showed no decrease in the second year after grazing (Donohoe et al., 2021). It also found that certain quality parameters in the forages improved in both years of bale grazing, including crude protein concentrations and nutrients like nitrate N, phosphorous and potassium (Donohoe et al., 2021). Donohoe et al. (2021) suggest forage growth reduction from bale grazing was negatively impacted due to smothering of the grass from excess hay and species, particularly in and around where the bale was situated. This effect was not noticed in an earlier Saskatchewan study which found forage yields to increase where bale grazing had occurred (Jungnitsch, 2008).

A paper from North Dakota State University reviewed the literature on the effects of winter bale grazing in the Northern Great Plains (Bachler, 2019). Many soil health benefits were found in the literature Bachler reviewed including improved nutrient concentrations and cycling, increased levels of soil organic matter, and better forage quality and production. A decreased risk of nutrient pollution was also noted (Bachler, 2019).

Omokanye et al. (2018) studied pasture rejuvenation in Northern Alberta. Bale grazing was among the options for rejuvenation and outperformed the alternative management scenarios (manure and fertilizer applications, breaking and reseeding, high-stock density grazing, and pasture resting) when it came to dry matter yielding. In fact, bale grazing generated a 200% increase in total dry matter compared to the control site (Omokanye et al, 2018). Bale grazing was also the method with the highest economic performance when excluding the cost of the bales (Omokanye et al., 2018). However, there may be tradeoffs when it comes to increased labour required for bale grazing and other supplies like fencing. An increase in diesel consumption is also expected since the bales need to be transported a further distance, though there will be energy/fuel savings from reducing the need to spread manure.

While such studies show improvements to the soil and production system generally, there has been little research done to quantify the soil organic carbon stock changes or GHG emissions over time from bale grazing. It is expected that an increase in overall forage biomass as well as straw residue from the bales, in conjunction with the added nutrients from livestock feces, will increase soil organic carbon stocks over time. However, more research is needed to confirm the potential of this practice as a GHG mitigation tool on the Prairies.

Barriers to Adoption

The main barrier to adoption is the availability of organic amendments and having a nearby source that is cost-effective to apply at meaningful quantities. There are logistical challenges to their use on top of sourcing, such as application method. Organic amendments tend to have a significantly lower concentration of nutrients than inorganic fertilizers and therefore require application rates beyond what traditional farm equipment can deliver at the time of seeding. So an additional field pass will be required and new farm equipment may be necessary, such as manure spreaders. There is also a lack of information on non-manure amendments in farm communities.

Co-Benefits

The co-benefits of applying organic amendments should be an important consideration for their use. Many types of organic amendments rich in nutrients and soil organic matter have the potential to improve crop productivity, water holding capacity, soil structure, and many other soil properties that are key for improving crop productivity and sustainability (Malhi et al., 2013, Chen et al., 2018). By helping to increase a cropping system’s net primary productivity, soil amendments can boost carbon inputs into the soil and increase levels of soil organic matter (Bolinder et al., 2007a). Organic amendments provide a myriad of soil-related features, including structure and tilth, aeration, moisture infiltration rates, additional and improved nutrient cycling, soil life, and reduced GHG emissions (OMAFRA, 2023) .

Many organic amendments used on agricultural soils come from products that are deemed as waste from other industries, such as rotten food from households and grocery stores or manure from livestock operations. Applying them to cropping systems is a solution to the problem of waste accumulation in landfills or manure storage facilities.

Trade-offs

Each amendment type comes with its own specific trade-offs. Manure, for instance, can spread weed seeds onto cropped fields when applied and contribute to higher weed density (Pleasant & Schlather, 1994). Solid-form amendments like compost and digestate can also be challenging to spread precisely without additional equipment that many grain farmers in the Prairies don't currently own. A concern with the reuse of biosolids like sewage waste is its potential to carry and spread pathogenic bacteria onto agricultural soils (Al-Gheethi et al., 2018). Nutrient loading of waterways and other environmental problems is an important trade-off where fertilizers and manure are over-applied or accumulate near waterbodies. Judicious use of the amendments across a broad region as the BMP adoption increases could alleviate this problem. The biggest trade-off facing virtually all organic amendments is the cost and energy required for scaling and distributing the amendments widely across the Prairies. Currently, application rates of organic amendments required to meaningfully substitute inorganic fertilizers or effectively improve soils have a prohibitively high cost of production and distribution.

Knowledge Gaps

There are several crucial knowledge gaps to fully understand the potential of organic amendments as a BMP within the Canadian Prairies. Firstly, there is a significant lack of data on the level of use of organic amendments on agricultural lands. Information on the production of manure is available because Statistics Canada monitors livestock populations, but adoption levels of amendments like compost, digestate, biosolids, compost teas, and others is severely lacking. There is also a lack of clarity on the emission factors associated with the amendments, as they vary widely on the type, composition and treatment. Studies have not aligned closely on levels of GHG emissions released during the production, storage, and use of organic amendments.

More research is also needed on the leakage potential of manure applications as a BMP. Most livestock manures are already utilized on agricultural lands, so the mitigation potential relies on the optimization of the amendment and applying them more efficiently. The potential for improved manure optimization across a wide region like the Prairies is not well understood.

Appendix

Table 22: Nutrient content of various types of manure

Manure Type	Total N (kg per 1000 L)			Total P (kg per 1000 L)		
	Min	**Avg**	**Max**	**Min**	**Avg**	**Max**
Liquid Pig	0.6	3.6	6.8	0.1	0.2	1.3
Liquid Dairy	0.7	3.1	7.6	0.01	0.3	1.7
Liquid Chicken	3.0	7.9	11.5	6.0	2.0	5.6

	Total N (kg per tonne)			Total P (kg per tonne)		
	Min	**Avg**	**Max**	**Min**	**Avg**	**Max**
Solid Beef	1.2	4.5	16.5	0.3	1.0	5.4
Solid Dairy	2.7	4.5	6.6	0.4	1.2	7.0
Solid Chicken	2.5	18.5	37.0	0.5	8.2	23.0

Source: Organic Field Crop handbook. Canadian Organic Growers, 3rd edition, 2017

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9. Rotational Grazing

Description

In Canada, grazing land in 2019 covered 14.3 million hectares. Grazing land consists of grazing land and tame pasture. The former is normally permanent with low level of external inputs and consists of native species, a mix of native and tame species (possibly seeded tame or invasive tame), or primarily tame species (the latter sometimes called naturalized grassland (Sheppard et al., 2015)). In contrast, tame pastures are typically terminated and reseeded periodically when productivity declines, when desired species are not present, and/or when there is a presence of excessive undesired plant species. In 2006, 32% of tame pasture managers rejuvenated tame pasture every 5 years or less, and 40% every 6-10 years, with 11% never being rejuvenated (Sheppard et al., 2015). In 2011, 13% of tame pastures received fertilizer (Sheppard et al., 2015) and, on average, 22% of the vegetation sward was legume (Sheppard et al., 2015) – two practices that increase productivity and forage quality.

Rotational grazing is the practice of moving grazing cattle through a set of paddocks. It contrasts with continuous grazing where cattle are in a single paddock through the grazing season. The main advantage of rotational grazing is increased vegetation growth (Alemu et al., 2019; Sanderman et al., 2015) and better graze quality (Wang et al., 2015), although Popp et al. (1997) found no significant effect on either herbage or quality from rotational grazing in Manitoba. There is a wide range of grazing practices within rotational grazing. Basic rotational grazing provides the opportunity for grazed plants to recover. Intensive rotational grazing has much shorter grazing periods, moving animals more often, to reduce stress on the plant from grazing (sometimes referred to as avoiding the “second bite” of any plant during a grazing period) and allowing for sufficient time for plant recovery after grazing. Unfortunately, there are not widely accepted definitions of this range of practices.

For this analysis we divided rotational grazing into 4 categories:

1. Continuous: no rotational grazing, continuous season-long grazing

2. Basic: grazing in which animals are rotated through multiple paddocks at least once.

3. Intermediate: multiple paddocks, in which animals are rotated through each paddock two or more times in a season and/or grazing is intentionally deferred in each paddock during critical vegetation growth periods over time. Therefore, grazing periods are shortened and there are longer and more strategic rest periods between grazing than basic rotational grazing.

4. Intensive: short grazing duration (< 8 days) per paddock with rest period between grazing on each paddock based on sufficient time to reach vegetation state consistent with long-term vegetation health. In some of these systems, the livestock may be moved to a new paddock daily.

Adaptive multi-paddock grazing (AMP) is rotation grazing with flexibility to adjust grazing based on conditions of the pastures and needs of the herd. Both intermediate and intensive rotational grazing could fit under the definition of AMP.

Effect of Rotational Grazing on GHG emissions

SOC Changes

Background SOC change

The grasslands of Canada gained an average of 130 kg C ha-1 yr-1 during the early 2000s based on atmospheric inversion models (USGCRP, 2018), although this value refers primarily to arctic tundra grasslands in addition to grazing land. In the Great Plains, grasslands in the same period were a sink of 240 kg C ha-1 yr-1 and are expected to remain a sink at a similar rate to 2050 (USGCRP, 2018). Nevertheless, the sequestration rate varies widely by year, including being a source in drought years, in response to weather. Grazing generally increases SOC compared to no grazing (McSherry and Ritchie, 2013) with rates of 72 to 190 kg C ha-1 yr-1 in the northern Great Plains (Wang et al., 2016; Wang et al., 2014). Therefore, much of observed increases may be due to recent improved grassland management that is restoring SOC that was lost from past poor management. In agreement with this, Wang et al. (2014) associates the increase in SOC from simply grazing rangelands to likely restoration of SOC after mismanagement in the Northern Great Plains. Over stocking in particular was a mis-managed practice in the first half of the 20th century. Similarly, initial findings showed that European grasslands appeared to be a continual sink of C as high as 1.29 Mg C ha-1 yr-1, sufficient to more than offset the emissions of CH_4 and N_2O from the grazing livestock (Soussana et al., 2010). However, Chang et al. (2016) showed that this SOC increase is the result of significant lowering of stocking on European grasslands due to policy changes during the 1980s and 1990s. Smith (2014) cautions that grasslands cannot be expected to be a perpetual sink as they eventually come to an equilibrium C, after which there will not be sustained increases in C stocks. So, the general grassland C sink will decrease over time as it approaches a new SOC equilibrium. Some of the studies do not have a control grazing system with which to compare C stock change. If there is SOC recovery from a degraded

SOC change from adoption of rotational grazing

New adoption of rotational grazing represents an opportunity to increase SOC on pastures. The available data (Appendix) does not allow robust analysis of additional C sequestration from adoption of rotational grazing in Canada since there are few studies for Canada and results are variable elsewhere. The general results globally are that rotational grazing increases SOC (Byrnes et al., 2018). This finding is supported by the subset of studies that have relevance to Canada; all the studies based on soil measurements either showed an increase or no effect on SOC. Only one study measured SOC loss, and that study simulated grazing with mechanical harvest rather than livestock. If rotational grazing had no benefit and the distribution of results is symmetrical about the mean, we would expect more studies showing loss of SOC from rotational grazing.

Byrnes et al. (2018) found that rotational grazing had greatest positive effects in humid climates. Compared with continuous grazing, grazing exclusion tends to increase SOC in wetter climates and decrease SOC in drier climates, with the effect being linear with annual precipitation in the range of 200 to 1000 mm (Derner and Schuman, 2007; Hu et al., 2016; McSherry and Ritchie, 2013). Therefore, rotational grazing will likely be more effective for increasing SOC as precipitation increases because the increased vegetation recovery time mimics some aspects of no grazing.

Having legumes in pasture has been shown to improve C sequestration (Conant et al., 2017; Fornara and Tilman, 2008; Henderson et al., 2015) and improve herbage quality (Bélanger et al., 2017; Peprah et al., 2018). The recovery periods and reduced sustained grazing stress with rotational grazing improves longevity and maintenance of seeded legumes (Forsythe, 2018). Burton et al. (2021) assumed all natural and tame pasture under intensive grazing will also be managed so that they will have sufficient legumes to provide the nitrogen requirements of the sward, whereas the continuous and basic scenarios may or may not have adequate legume content.

Wang et al. (2021) point out the difficulties of detecting the soil impacts of grazing systems including interactions between climate, soil, vegetation, animal species, and management. Other difficulties include variation in actual pasture management practice so that the management over time is hard to place into a specific category.

Based the available data (Appendix) and our expert opinion, we estimated conservative average C sequestration rates that would be applicable over 30 years (Table 23). Note that rates of 100 kg C ha^{-1} yr^{-1} or less, even over a 30-year period, would be difficult to detect through measurement and so may be reported in scientific literature as no change (Maillard et al., 2017). The values are highly uncertain due to limited amount of evidence specific to Canada. Therefore, we suggest that uncertainties would be in the order of ±100%, i.e., ranging from no change to double the derived gains.

Table 23: Estimated mean rates of C sequestration from changing from continuous grazing for the Prairies (Farmers for Climate Solutions, 2022).

		Zone	
Pasture Type	grazing system	Semiarid Prairies*	Subhumid Prairie*
Natural		--- C sequestration from continuous (kg C ha^{-1} yr^{-1}) ----	
	Basic	0	0
	Intermediate	20	40
	Intensive	90	180
Tame		--- C sequestration from continuous (kg C ha^{-1} yr^{-1}) ----	
	Basic	0	0
	Intermediate	30	60
	Intensive	120	240

* the semiarid Prairie is the Brown and Dark Brown soil zones of Alberta and Saskatchewan, subhumid Prairie is the rest of the Prairies.

Other emissions

Improvements in forage quality will be expected to reduce enteric methane emissions. Some studies show increased forage quality with rotational grazing (Billman et al. 2020), some show no effect (Popp et al. 1997), while others show a reduction (Alemu et al. 2019). Given the uncertainty, it is difficult to estimate how rotational grazing will affect enteric methane emissions.

Potential impact of rotational grazing on Prairie GHG emission

Current adoption

Currently, about 50% of beef producers use rotational grazing according to 2016 Census of agriculture (Beef Cattle Research Council, 2019) with adoption similar across provinces. In 2011, about 25% of beef producers reported using continuous grazing on tame pasture and 35% using continuous grazing on native pastures (Sheppard et al., 2015). Fully 66% of beef producers had 2-4 paddocks for tame pasture and 58% had 2-4 paddocks for native pastures in 2011 (Sheppard et al., 2015). These would be classed as basic rotational grazing by our definition. The lack of standard definitions makes it difficult to interpret and reconcile surveys. Because of characteristics of different pasture areas and the different nutritional requirements of different groups of livestock, a producer could have some pasture area with basic rotational grazing and some with more intensive rotational grazing; this adds to confusion when survey asks for only one type of grazing system.

Kristine et al. (2021) surveyed 97 pastures on 28 ranches distributed across southern and central Alberta to assess the effect of grazing and other factors on range health. The ranchers where volunteers so may have been more interested in range health and thereby possibly more likely to use rotational grazing than non-volunteers. Nevertheless, only two pastures, both tame, had a grazing period of 1 day. Eighteen pastures (included 4 native pastures) had grazing period of 2-8 days so would be classed as intensive in our typology. Thirteen pastures (7 native and 6 tame) had a grazing period over 60 days indicating continuous rotation grazing. Twenty-three pastures had a grazing period of 9-21 days which were assumed to represent intermediate rotational grazing in our typology. The remaining 42 pastures with grazing period between 22-60 days would be basic rotational grazing. Of note, range health scores, for both native and tame pastures, tended to decrease linearly as grazing period lengthened. This trend is consistent with the concept that transition to more intensive rotational grazing improves the quality of pasture which in turn improves soil quality.

The trend in Canada is towards increased rotational grazing and a shift towards intensive rotational where the pasture area is suitable in terms of availability of livestock watering sources and soil-vegetation landscapes amenable to numerous small paddocks. Rotational grazing is promoted by the Canadian beef industry and governments. Table 24 lists the estimated current adoption rates.

Potential Adoption and Impact on GHG

By 2030, Burton et al. (2020) assumed substantial increases in advanced basic and intensive rotational grazing, particularly in the subhumid prairie. Table 24 lists the estimated current and potential 2020 adoption rates. To realize this potential, there needs to be sufficient capacity for grazing practices, either from advisors or from farmer/rancher training, and building confidence that rotational grazing will have economic benefits that are larger than the increased costs. Cost-share for the costs, especially for up-front costs for infrastructure improvements, help build that confidence of positive net economic benefit from rotational grazing. With greater experience and more evidence of positive results gleaned from nearby adopters over time, more farmers should increase confidence of the merits of adoption without necessarily requiring any cost-share. Water availability was assumed to limit the extent of adoption of intensive grazing in dry climates, particularly for natural pastures.

The scenario of technical potential adoption and the associated GHG reduction from a baseline of current adoption are shown in Table 24.

Table 24: Current and technical potential adoption rates by 2030.

	Semiarid Prairie		Subhumid Prairie	
Grazing System	Current	2030	Current	2030
	---- Natural Pasture Adoption Rates (% of area) ----			
Basic	50	50	50	25
Intermediate	10	25	10	25
Intensive	5	15	10	45
	-- Tame Pasture Adoption Rates (% of area) --			
Basic	50	35	40	20
Intermediate	10	35	20	20
Intensive	5	25	10	55

Table 25 shows the additional C sequestration assuming the technical potential is reached in 2030 compared to continuation of current adoption rates to 2030.

Table 25: Potential C sequestration mitigation by province and pasture type.

Jurisdiction	2030 Mitigation (Mt CO_2e/yr)		
	Tame	Natural	Total
AB	0.384	0.123	0.507
MB	0.089	0.031	0.120
SK	0.243	0.076	0.320
Prairies	0.716	0.230	0.946

Barriers to Adoption

The barriers to adopting intermediate rotational grazing from continuous or basic practices are more investment in fencing and water capacity and more labour for pasture assessment, grazing infrastructure maintenance, and cattle movement. The adoption of intensive rotational grazing requires even more infrastructure and labour and can involve a lifestyle change because of the need for frequent cattle movement. Producers are more likely to move incrementally than to make large changes in management, i.e., preferring to transition from continuous to basic, from basic to intermediate, and from intermediate to advanced. Consequently, to increase adoption of intensive grazing requires increasing the transition from continuous to basic and basic to intermediate. Farmers might not transition all their herd to an improved grazing management so could have a mix of grazing practices during transition.

Co-benefits

Rotational grazing has important co-benefits of maintaining and increasing biodiversity. Rotational grazing improves soil health (Byrnes et al., 2018), increases above and below ground biodiversity (Reshmi et al., 2020; Teague and Kreuter, 2020), and maintains legumes that reduce need for N fertilizer (Forsythe, 2018). Natural grazing lands are important reservoirs of plant, animal and soil biota biodiversity and support biodiversity of many animals that use grasslands but may also migrate beyond that grazing land base.

Rotational grazing, including adaptive rotational grazing, is a more resilient grazing system to drought because of the longer land rest periods between grazing activities.

Trade-offs

Moving to advanced and intensive rotational grazing will probably reduce the area of grazing land required as the same amount of cattle can be fed on smaller land area. All things equal, this pattern could lead to a drop in grazing area and provide an incentive to convert grazing land to cropland. This conversion results in loss of biodiversity, loss of soil matter and soil carbon, movement of nutrients and pesticides to the surrounding environment, as well as increased greenhouse gas emissions from nitrogen.

Research Gaps

The effect of adoption of rotation grazing on soil health including SOC needs further investigation. Associated with this gap is information about how to determine how rotational grazing affects livestock enteric emissions.

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Appendix: Rates of Carbon Sequestration in the Literature

Table 26: Values of SOC sequestration for rotational grazing

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
-- Multi Continents --						
Global	1-98	Meta-analysis of published results	Rotational vs continuous	Can't be calculated from data provided	(Byrnes et al., 2018)	Rotation 32% higher (ln RR = 0.28)

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
Temperate (location not defined but model only validated for Montana ranches)	80 (equilibrium (Derner and Schuman, 2007))	Modelling	rotational vs continuous	16-pasture vs 4 pasture, 0-60; 4 pasture vs continuous: 0-1000+	(Ritchie, 2020)	Reduced loss since continuous grazing was estimated to be losing SOC, rotational benefit increases as stocking rate increases
Global		Review of published literature	Rotational (Holistic) grazing vs continuous	0	(Hawkins, 2017)	No evidence of difference from available studies
Global	N/A	Review of published results	Additional from "improved grazing" (assumed to be rotational)	280	(Conant et al., 2017)	
-- Prairies ---						
Saskatchewan	N/A	Modelling	Change to rotational (basic) grazing on tame pasture in Black soil zone	65	(Lynch et al., 2005)	
Saskatchewan	18	measurement	Rotational (intermediate) grazing compared to continuous native species mix established on cropland	200 (0-60 cm)	Iwaasa (unpublished), experiment described in (Alemu et al., 2019)	P=0.09

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Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
Manitoba	5	Measurement	Rotational grazing (intensive) on tame pasture vs continuous	340	(Manas et al., 2000)	Results not statistically significant
Alberta	30	Modelling with validated Century model across Alberta	Change to rotational grazing for rangeland	Rotational grazing with long duration grazing: -400 to -100 (loss) Rotational grazing with short duration grazing duration: 100-200 kg C/ha/yr 10% reduction in stocking rate 200-300 across all grazing practices	(Iravani et al., 2020)	
Prairies	?	Measurement across ranches	AMP vs conventional practices	0	(Breitkreuz et al., 2019)	No evidence of difference
Alberta	5	Native rangeland	Deferred rotational vs non grazing	0 difference between treatments	(Dormaar et al., 1997)	Grazing pressure was very light
Alberta	?	Primarily tame pasture	Rotational vs continuous	Rotational had 6% less SOC concentration than continuous, not significant	(Pyle *et al.*, 2019)	No pairing between management systems to reduce confounding effects on SOC
Prairies	10+	Ranch grasslands	Adaptive multi-paddock grazing vs non-AMP between ranches	Soil under AMP has increased CH_4 uptake and no increase in CO_2 or N_2O emission	(Shrestha et al., 2020)	Lab incubation study so can not be extrapolated to actual rates in the field

Location	Duration (yr)	Study type	Comparison	C sequestration rate (kg C ha^{-1} yr^{-1})	Reference	Comments
----------United States----------						
US grazing lands	N/A	Expert opinion		Rangeland: 70 to 300 Tame pasture: 300 to 1400	(Morgan et al., 2010)	
South Dakota	30+	Measurement across ranches	Rotational grazing vs continuous	0	(Hillenbrand et al., 2019)	
Wyoming	11	Experiment on native rangeland	Continuous heavy vs deferred heavy and short duration heavy	0 for continuous heavy vs. short duration, -590 (loss) for continuous heavy vs deferred heavy grazing	(Manley et al., 1995)	all treatments with stocking for heavy grazing had less SOC than continuous light grazing
----------Outside of North America----------						
Australia	5-15	Measurement across adjacent farm paddocks		0	(Sanderman et al., 2015)	
Argentina	8	Ranch, saline soils	Rotational vs continuous	560 for rotational vs continuous	(Vecchio et al., 2018)	Difficult to estimate precisely from data provided

10. Short-term rotation of annual crops with perennial forages

Description

Crop rotations vary across the Canadian Prairies, often utilizing more than one annual crop. The additional of perennial forages for a short duration (e.g., 2-3 years) to annual cropping sequences can provide many agronomic and environmental benefits. Some benefits of an optimized crop rotation include moisture conservation, reduced pest infestation, weed suppression, improved soil quality and increased soil organic matter (Cutforth et al. 2013, Entz et al. 2002). Current annual crop rotations involve cereals, oilseeds, pulses, vegetables and specialty crops like borage or caraway. Perennial crops which typically consist of perennial grass and alfalfa, or a combination of both crop types can be introduced into short-term annual rotations for varying lengths of time. Forages in rotations are known to provide numerous benefits (Entz et al. 2002). In the Northern Great Plains region, Entz et al. (2002) reported only 5-15% of arable land is within a rotation of annual crops with perennial crops. Perennial forage crops are known to enhance sustainability of dryland cropping systems, therefore, there is considerable interest in using this practice in modern agriculture (Jefferson and Cutforth 2005).

Effect of short-term rotation of annual crops with perennial forages on GHG emissions

Soil Carbon

A short-term change in a cropping system will alter soil properties. Previous studies have reported that Canadian Prairie agricultural soils can restore up to three-quarter of repleted native soil organic matter through the practice of rotating perennial forages in cropping systems (Maas et al. 2013). The inclusion of forage perennials in an annual cropping system alters storage of soil organic carbon (SOC) through shifts in timing and type of organic inputs, a greater root:shoot ratio, and a reduction in disturbance (i.e., minimized tillage). The deeper root systems of perennial forage plants assisted by microbial activities can sequester carbon deeper in the soil profile compared to annual crops, resulting to increased carbon storage (Culman et al. 2010, DeLuca and Zabinski 2011). In southern Manitoba, Maas et al. (2013) investigated an initial establishment year of alfalfa/grass on annually cropped lands and reported a gain of 170 g C m^{-2} y^{-1} (no harvest removal) which reduced to 84 g C m^{-2} y^{-1} following a single hay cut. From a management point of view, perennial pastures provide a large litter base that enhances soil C storage.

Estimates of C sequestration rates for short term rotations of annual crops with hay/pasture are not well established in the literature for the Prairies. In a conservation reserve program in the USA, C sequestration in cropland seeded to perennial grasses averaged 1.1 Mg C ha^{-1} yr^{-1} (Gebhart et al. 1994). Using a carbon modelling approach (Century model), soil carbon change factors induced by perennials ranged between 0.46—0.56 Mg C ha^{-1} yr^{-1} and in agreement with the empirical derived value (VandenBygaart et al. 2008).

N_2O Emissions

Rotations of annual cropping systems to perennial forages may affect N_2O emissions. Increases in below-ground biomass via deeper, denser root systems, and longer growing seasons associated with perennial systems may reduce N_2O emissions relative to annual systems (Maas et al. 2013). However, conflicting research reports N_2O emissions increase from soil following the transitioning of annual rotations to perennial cropping. Perennial systems may induce increased availability of labile C and N substrates from proportionally augmented root exudation and tissue decay, increased soil bulk density and water-filled pore space, resulting in ideal conditions for N_2O production in the short-term (Daly et al. 2022). In a study conducted on the Canadian prairies, Westphal et al. (2018) revealed that perennial forage alfalfa following an annual crop in an organic production system, when ploughed down in late summer/early fall did not result in N_2O emission as was expected. These observations contradicted other studies (Flessa et al. 2002; WagnerRiddle et al. 1997; and Brozyna et al. 2013) that reported increased N_2O emissions from alfalfa incorporation in the spring. Warm and wet conditions in the spring favoured episodic bursts of N_2O emissions indicating that soil moisture is a limiting factor for N_2O emissions in the spring. Moreover, these studies either incorporated green manures (Flessa et al. 2002; WagnerRiddle et al. 1997) or applied manure (Brozyna et al. 2013) that reinforced N_2O emissions.

N Balance

Crop rotations that include perennial legumes can effectively reduce energy consumption since they do not require N fertilizers and often add N to the soil for succeeding crops. Perennial cropping is less reliant on energy consumption and chemical inputs compared to annual cropping systems, reducing the overall GHG emissions of the systems, because the frequency of seeding is reduced in perennial rotations. However, farm operations do occur in hay systems which may reduce the benefits. Perennial legumes within rotations have shown to reduce both carbon and energy footprints of various crops (Entz et al. 2002; Maas et al. 2013; Wagner-Riddle and Thurtell 1998).

Potential impact of practice on Prairie GHG emissions

Current adoption

Current adoption rates of annual rotations with the inclusion of perennials, do not exist for Canada or the Canadian Prairies. This is because crop rotations are not considered in the Census of Agriculture, and there is little known about the adoption of perennials within an annual rotation in the Prairies. However, a 1992 survey on rotational benefits of forage crops in Prairie cropping systems documented 253 forage producers across the six agroclimatic zones in Manitoba and Saskatchewan, with an average farm size across all regions of 536 ha (Entz et al. 1995), a relatively small portion of annual cropping systems.

Barriers to adoption

The inclusion of perennial forages in rotation with annual grain crops is often avoided by Prairie producers due to reduced yield of subsequent grain crops (Entz et al. 1995). Cuthforth et al. (2010) reported that first-year wheat yields and water use efficiency were significantly lower following 6 years of alfalfa or crested wheatgrass. In the second year of wheat production, the effect of the previous alfalfa on wheat yield continued, while the wheat yield after crested wheatgrass was similar to yield from continuous wheat rotation (Cuthforth et al., 2010).

Another major limitation that discourage producers in the prairies from cycling forages into rotations include the difficulties of establishing and terminating perennial forage stands (Entz et al. 2002). Traditional techniques for forage stands establishment and termination rely heavily on intensive tillage, which can lead to soil erosion both during the forage establishment and after termination. Forage termination could also be achieved through the application of non-selective herbicides, which reduces soil erosion. However, losses of nitrogen are associated with herbicide application especially when forages are not incorporated (Mohr et al. 1998). Various studies (Glasener and Palm 1995; Janzen and McGinn 1991; Larsson et al. 1998) have shown that nitrogen may be lost through volatilization when terminated legume is not incorporated into soil. Further, Biederbeck and Slinkard (1988) observed a yield reduction of about 22% for wheat when legume was terminated by herbicide and left on surface than when legume was incorporated after herbicide termination. However, more recent research on incorporation of perennial forages is needed to assess if this agronomic barrier persists.

Co-Benefits

Added N

Perennial forages and legumes provide opportunity for soil N enrichment, which is beneficial for subsequent crops in rotations (Entz et al. 2002). The amount of N introduced may vary from modest (8 kg N ha^{-1}) (Jefferson et al. 2013) to larger amounts (62 kg N ha^{-1}) (Thiessen Martens et al. 2015). In a field experiment in southern Manitoba, Kelner at al. (1997) reported an average of 84, 148 and 137 kg N ha^{-1} for three years in succession, respectively, in an alfalfa hay stand indicating a potential benefit of perennial alfalfa towards soil N status.

Weed Suppression

Weed suppression with forages particularly perennial hay crops, is well researched. Siemens (1963) reported a <1% of wild oat composition within a perennial forage stand after rotated with annual crops compared to a 15% wild oat composition in continuous grain or fallow grain systems. The Canadian Prairie farmers survey revealed that 83% of producers reported minimal weeds after rotations of forage legumes and annual crops (Entz et al. 1995). Similarly Ominski et al. (1999) documented weed suppression under perennial alfalfa or alfalfa-grass hay crops rotated with wheat than wheat in annual grain rotations.

Trade-offs

P losses

The perennial phase of an annual rotation with perennials can deplete soil nutrient status, especially under hayed systems (Welsh et al. 2009). For example, phosphorus (P) depletion was highest from a forage-grain rotation within a relatively short time frame due to high P demand by alfalfa and the higher grain yields achieved in this rotation (Welsh et al. 2009). However, livestock manure or other amendments can return nutrients that have not been replaced to the system, to reduce soil nutrient depletion (Welsh et al. 2009). Harvesting forage through grazing can cause nutrient cycling in comparison to haying, regardless of the heterogeneous nature of nutrient deposition through grazing (Sigua et al. 2007). Harvesting through grazing also reduces costs associated with hay removal and manure application (Thiessen Martens et al., 2015).

Chemical properties

Forage legumes can affect soil chemical properties. Long term rotation of forage legumes has resulted in decreased soil pH in the Brenton plots in Alberta (Entz et al. 2002). Consequently, liming is often introduced into soils in this location to maintain the status of soil pH for crop production.

Water Use

In drier areas of the prairies where water is a critical factor for crop production, rotation of annual crops with perennial forages can significantly diminish crop yields of the following crop in sequence because of forage induced drought (Entz et al. 2002).

Knowledge Gaps

There is very little information on the effect of annual rotation with perennials on GHG emissions across the Prairies, particularly N_2O, and SOC. This is a challenging to measure due to the varying nature of crop rotations, and the different perennial species selected to integrate within an annual system. Current adoption rates need to be quantified to estimate potential mitigation values in the region.

Nutrient cycling can be complex in pasture in comparison to hay systems, which has had within the literature. The impacts of nutrient cycling on perennial forages in moist areas is different than for dry areas of the Canadian Prairies. Nutrient cycling is also impacted by the length of perennial forage stands within the sequence and the type of perennial forage (specifically between legumes and non-legumes). Therefore, investigation at the system level is needed to understand all the efficacy of this BMP for GHG mitigation.

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BMPs Pertaining to Improving Natural Systems

This section focuses on management practices that specifically improve the natural ecosystems on the land, including forests, grasslands, and wetlands. The first three BMPs pertain to increasing, managing, and conserving trees within the Canadian Prairies either on the agricultural landscape (shelterbelts, windbreaks, silvopasture) or within the neighbouring natural habitat (forest conservation). Forested areas improve natural habitat and biodiversity, as well as store carbon in woody biomass and soils. The other BMPs included within this section include conservation of grasslands or wetlands, as well as wetland restoration. These are BMPs with potential to mitigate climate change while also providing co-benefits to water quality and biodiversity. The BMPs outlined in this section often occur on non-agriculturally productive lands such as edge of fields, riparian zones, wetlands, etc.

11. Increase and Manage Trees in Working Agricultural Landscapes

Description

This BMP covers the various practices that include increasing and managing trees in agricultural landscapes, such as shelterbelts or windbreaks, vegetating riparian areas or silvopasture (tree/pasture systems).. Alley cropping, an agroforestry technique where tree species are added in rows across a crop field, was excluded from this review. This exclusion is based on the typical crop management styles and large machinery size used in the Canadian Prairies that complicate alley cropping. Moreover, expansion of trees in agricultural areas where forests are not the naturally dominant vegetation is typically discouraged due to drought risk affecting tree permanence or potentially negative impacts on native biodiversity.

Riparian Zones

The riparian zone is the area between the upland zone and the shoreline of streams and lakes. It forms a corridor between land and water, allowing animals to travel between different biomes. A healthy agricultural riparian zone contains diverse plant species as well as aquatic and terrestrial wildlife. It helps to maintain water levels, stabilize temperature of the water bodies, and prevent erosion and runoff of nutrients and other contaminants into the water body. When vegetation is removed from the riparian zone, it can negatively impact the health of the water body by decreasing water quality and reducing biodiversity.

Riparian areas do not all look the same. The vegetation of healthy riparian areas surrounding prairie pothole sloughs or southern prairie streams often consists mostly of sedges, grasses and shrubs, such as willows or dogwood. On the other hand, the riparian zones of boreal, foothill, or parkland streams usually include larger trees such as alder, aspen or spruce, in addition to grasses, sedges and shrubs. Establishment of a healthy vegetation community including grass, shrubs and trees in riparian zones surrounding water bodies has been identified as a natural climate solution for mitigating GHG emissions (Whalen, 2003; Drever et al., 2021).

Incorporating trees into agricultural landscapes offers many benefits in addition to carbon capture, including biodiversity conservation and improvements in microclimate, air and water quality (Schoeneberger et al., 2012). Riparian tree planting involves establishing tree buffers along water bodies to stabilize and cool stream channels, as well as to reduce the export of nutrients and sediment from agricultural fields into water systems (Vijayakumar et al., 2019).

Shelterbelts

Shelterbelts and hedgerows are rows or clusters of trees often planted or left standing on the edges of croplands. Shelterbelts or windbreaks are often linear planting of trees used to form a barrier and protect fields from wind damage (Ramachandran, et al., 2010). Shelterbelts and windbreaks are a common practice in the Canadian Prairies to control wind erosion, protect wind-sensitive crops, enhance crop yields, reduce animal stress and to protect from dust, odor and pesticide drift from other fields (Schoeneberger et al., 2008).

Silvopasture

Silvopasture integrates trees into animal production systems, often including trees in pasture systems where animals graze or exercise. Silvopasture systems are often widely spaced, with trees scattered across a field (Ramachandran, et al., 2010). Trees in pasture systems provide shelter and shade for pastured animals, which can improve both the water consumption and feed efficiency of pastured livestock (Kamal et al., 2018).

Effect of Increasing and Managing Trees in Agricultural Landscapes on GHG emissions

Soil Carbon

Tree planting in agricultural lands increases the above-ground and below-ground stores of carbon in tree biomass, dead organic matter in leaf litter and soil organic carbon (Schoeneberger, 2008). Tree growth rates vary based on tree planting date, species and age of tree when planted. Tree growth rates and carbon accumulation in above-ground woody biomass can be estimated for many tree species. Species selection is an important consideration for both biomass and soil carbon storage, as it impacts the growth rate, CO_2 respiration, and is heavily influenced by climate (Schoeneberger, 2008). A global meta-analysis suggests that Canadian specific above-ground growth rates are 0.96 (0.48-2.26) Mg C ha^{-1} yr^{-1}, and below-ground rates are 0.44 Mg C ha^{-1} yr^{-1} (range not reported) in the first 30 years after establishment (Cook-Patton et al., 2020). Combined, the average GHG sink from the growth of tree species in Canada is equivalent to 5.13 tCO_2e ha^{-1} yr^{-1}.

For soil organic carbon accumulation, average sink rates of 3.2 ± 3.77 t CO_2e $ha^{-1}yr^{-1}$ (± SD) in 10 years after planing CO_2e $ha^{-1}yr^{-1}$ 30 years after planting, following afforestation of tree cover, for North America are reported in the literature (Cook-Patton et al., 2020, Lal, 2005, Nave et al., 2018). Soil carbon sequestration rates representative to the Canadian Prairies are reported in Table 27.

Table 27: Carbon sequestration rates from different agroforestry techniques, where negative values indicate soil carbon loss.

Description	Carbon Sequestration (tCO_2e ha^{-1} yr^{-1})	Region	Reference
Riparian Zones – Afforestation/Reforestation	3.94	Canada	Drever et al., 2021
Riparian Zones – Smart Landscape (20% trees, 80% grass)	3.49	MB, SK, AB	Drever et al., 2021
Shelterbelt/Hedgerows (top and subsoils)	2.2 (0.33-4.07)	Temperate Climate, Global	Mayer et al., 2022
Shelterbelts	2.57	SK	Dhillon and Van Rees, 2017
Silvopasture	3.04	Canada	Drever et al., 2021
Silvopasture (top and subsoils)	-0.73 (-3.52-0.03)	Temperate Climate, Global	Mayer et al., 2022

Other GHG Emissions

In addition to the carbon storage that trees provide via biomass and soil, agroforestry systems can also impact GHG emissions from agricultural systems, depending on the system to which they are added. For example, trees in pasture can influence both CH_4 and N_2O emissions from the soil via microbial respiration and decomposition. In croplands, trees can reduce the N_2O emissions associated with fertilizer or organic amendments. Since forested areas can influence the GHG emissions within agroforestry systems, the global warming potential or net GHG emissions should be considered when implementing agroforestry systems.

Agroforestry systems reduce net GHG emissions from both cropland and pasturelands within the Canadian Prairies (Baah Acheamfour et al., 2016). Under hedgerow, shelterbelt and silvopasture systems, the global warming potential was reduced in comparison to a non-forested control, either cropland or pastureland. The trees influenced the GHG emissions of CO_2, CH_4 and N_2O, but overall contributed to a lower net GHG emissions, with silvopasture systems having the lowest net GHG emission (76 kg CO_2e per ha).

Potential impact of Increasing Trees in Agricultural Landscapes on Prairie GHG emissions

Current and Potential adoption

Based on recent Cenus of Agriculture (Statistics Canada, 2022), approximately 60% of reporting farms in the Canadian Prairies have windbreaks or shelterbelts on their farm. However, the density of trees, species and status are unknown for many of these farms. In Saskatchewan, work has established a baseline for shelterbelt species composition, row widths, stand condition and type (Piwowar et al., 2016). This work identified 262,000 shelterbelts covering 51,000 km in the province of Saskatchewan, where 95% of shelterbelts were in good condition (Piwowar et al., 2016).

Previous work illustrated the potential area for silvopasture and riparian tree planting in the Prairie provinces. Drever et al., (2021) quantified potential adoption of silvopasture of 921,894 ha in British Columbia (BC), Alberta (AB), Saskatchewan (SK) and Manitoba (MB), based on 20 hectares of silvopasture per farm and a tree density of 111 trees per hectare. For riparian tree planting, potential adoption was estimated based on 30-m buffer zones on each bank on areas currently under crop or pasture. However, based on practical considerations, we suggest a feasible area for riparian tree planting be on a 6-m buffer zone (3m on each bank). Given this assumption, the estimated potential adoption for riparian tree planting is 14,208 ha across BC, AB, SK, and MB (based on downscaling the estimates from Drever et al., 2021).

Barriers to adoption

The primary barriers to increasing trees in agricultural landscapes are from the costs and labour associated with tree planting and management. Most costs are associated with establishment and maintenance of trees, removal of dead trees and in some cases snow removal (Rempel et al., 2016). Drever et. al (2021) estimated the cost for implementing riparian tree planting in Canada is between \$100 -\$197/t CO_2e for the first 10 years but it will decrease with time. This cost includes site preparation, planting stock, tree planting, post-planting mortality replacement, and herbicide application. The cost will likely be higher than this estimation since the total cost for restoration of riparian buffers should also account for area of high-value crops being displaced. For better adoption of tree planning in riparian buffer zones, narrower zones were identified (6m instead of 30m), so costs associated with tree planting should be less than those reported in Drever et al., (2021).

The benefits of shelterbelts vary based on the crop types grown, with some specialty crops (i.e., lentils, fruits and vegetables) having a highly positive yield response to shelterbelts, while others have a low yield response (i.e., drought-hardy cereals), contributing to the removal of shelterbelts within the Canadian Prairies (Rempel et al., 2016). Increasingly, other crop management strategies have been found to replace or eliminate the need for shelterbelts to increase moisture availability and mitigate erosion (Casement and Timmermans, 2007). This has led to shelterbelts being removed, as they are unnecessarily imposing costs to producers while giving the same benefits achieved through minimal tillage practices (Casement and Timmermans, 2007).

Co-Benefits

Riparian buffers with a healthy, diverse vegetation community play a vital role in agricultural ecosystem. They can filter out pollutants from agricultural land runoff, help to stabilize eroding stream banks, and provide many other benefits to aquatic ecosystems including making a direct impact on reducing excess nutrients, sediment and other contaminants in the rivers, creeks and streams as well as increasing sequestration of CO_2 through both above- and below-ground biomass (Schoeneberger, 2008).

Shelterbelts play an important role in agricultural production due to their essential to the Canadian Prairies for erosion mitigation. Well-designed shelterbelts also protect infrastructure and people from drinking snow, dust and pesticide residues from roads and fields (Rempel et al., 2016). For livestock operations, shelterbelts and silvopasture systems provide shelter, improve feed efficiency, reduce mortality rates, improve forage and pasture crops and increase or improve onsite water quantity (Sharrow et al., 2009, Rempel et al., 2016, Broster et al., 2010).

In general, trees in agricultural landscapes provide habitat, improving biodiversity in non-diverse intensified systems (Rempel et al., 2016). Biodiversity improvements can be observed above- and below- ground, and create recreational opportunities including hunting, bird watching and hiking (Banerjee et al., 2016, Kroeger and Casey, 2007). Agroforestry systems mitigate erosion and sediment transport, which has many societal benefits including improved water quality from a reduction of sediment and nutrient loading in waterways (Brandle et al., 2004).

Trade-offs

The primary trade-off for increasing trees in agricultural landscapes is taking agricultural land out of production. Shelterbelts on livestock operations can cause repair costs for fencing, habitat for livestock predators and cause encroachment of non-pasture species (Laporte et al., 2010, Brandle et al., 2009).

Knowledge Gaps

Boufroy et al., (2019) made several recommendations regarding measuring sequestered carbon in the riparian restoration context to better estimate ground level estimates and to support simulations. Together with uncertainties raised above, it can be concluded that more research is needed to broaden the regional databases of riparian-based and small-scale forestry carbon sequestration. Long term monitoring is needed to track f how different tree species are growing in light of varying landscaping and maintenance configurations. This includes enhancing soil sampling to adjust root biomass measurements and improve soil carbon precision. It is difficult to know in advance what the outcome of the woody biomass grown in riparian strips will be; while some long-term sequestration is possible (i.e., harvesting wood to use as a construction material), short term uses (i.e., wood fuel or biochar production) may offset the cost and counter-act these benefits.

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12. Reduce Deforestation to Agriculture

Description

Forests contain enormous quantities of carbon in the aboveground vegetation and below ground roots, and further contribute to soil organic carbon stocks. CO_2 fluxes from forests can at times be a net-source of CO_2, while at other times being a net sink of CO_2. Much of this flux depends on variables like climate, fires, pests, or other changes. Anthropogenic conversion of forests to agriculture lands in the Canadian Prairies has important implications for GHG emissions and biodiversity. While agriculture is not the only cause of forest conversion, expanding crop production or livestock grazing into forest zones is occurring on the Prairies and across Canada, with pasture expansion being the more common cause (NRC, 2022). This BMP considers the GHG mitigation potential of limiting agriculturally driven forest conversion.

The practice of converting treed land to agriculture production in Canada uses high-tech methods which utilize mulchers, grinders, big crushers, and other machines used to prepare seedbeds for cultivating crops and grasslands (Western Producer, 2017). The wood is sometimes removed or sold off the land, other times it is piled, buried, or burned. Seeding over the previously forested areas allows farmers to expand their cultivation or grazing to areas that were less economically productive. Deforestation occurs gradually, similar to the conversion of grasslands and wetlands. Individual farm decisions collectively add up to significant impacts on ecosystems and the environment.

Effect of deforestation from agriculture on GHG emissions

Soil Carbon

In Canada, forested lands contain substantial stocks of carbon. For instance, the managed portion of Canada's Boreal Forest holds 28 Pg of carbon when accounting for the above and below ground biomass, dead organic matter, and soil carbon pools (Kurz et al., 2013). The Boreal Forest soils contain more total ecosystem carbon than there is in the above-ground forest vegetation (Martin et al., 2005, DeLuca & Boisvenue, 2012). Therefore, Boreal forested areas are substantial contributors to soil carbon and the potential soil carbon losses from deforestation can be a major contributor to Canada's GHG emissions. However, this phenomenon is not consistent with temperate forests in other areas of Canada (coastal BC, Southern ON and QC).

Table 28: *Soil Carbon Stocks in Canada's Boreal Forest. Adapted from Kurt et al., 2013.*

Soil Type	Area ($x10^3$ km^2)	C Stock (Pg C)	C Density ($Mg.ha^{-1}$)
Mineral Soils			
Frozen	614	33	537
Unfrozen	2752	38	138
Total	3366	71	210

Soil carbon can be released from forested lands when cleared for other purposes including crop and livestock production. Soil carbon stocks can decline when soil disturbance is increased, and net primary productivity is reduced as this reduced the contributions of plant residue inputs to the soil. Soil characteristics can also be impacted, such as bulk density, soil nutrients, moisture, and soil aggregation can also be affected, for better or worse, when forests are converted to pastures and cropping systems (Baah-Acheamfour et al., 2016). The National Inventory Report estimates little to no change in soil carbon stocks when forested areas are converted to hayland or pasture in Western Canada. This lack of change in soil carbon stocks is due to the carbon input and mineralization of soil organic carbon of both systems being approximately the same (ECCC, 2019). However, when converted to annual cropland, a soil loss of 0.5 t C ha^{-1} yr^{-1} is reported (ECCCb, 2022).

Other emission impacts

There are several emissions pathways resulting from the conversion of forested land to agriculture. The most immediately important are losses of tree biomass in the year of conversion and subsequent burning (Zhou et al., 2013, ECCC, 2019). The decaying of dead biomass pools, including tree roots and other plant debris, happens over time (Amiro et al., 2010, ECCC, 2019). The fossil fuels used in the machinery required to clear the vegetation is another emission source, as is the net change in SOC (ECCC., 2019). Land use changes also affect the surface albedo of an area and the conversion to agriculture decreases the albedo (reduction in radiative forcing) with the amount being larger for coniferous than deciduous trees (Lee et al, 2011, Longobardi et al., 2016).

Potential impact of deforestation from agriculture on Prairie GHG emissions

Current adoption

Agriculturally driven deforestation in Canada occurs most commonly in the three prairie provinces where forested lands are integrated in or edge historical agricultural regions (Kurz et al., 2013). The National Inventory report reports deforestation based on different ecological regions. The regions relevant to the Canadian Prairies include the Boreal Plains (including BC Peace), sub-humid prairies and semiarid prairies, but exclude the Rocky Mountain Foothills and the edge of the eastern Manitoba shield. The rate of deforestation between 2001 and 2020 was 14,648 ha per year for the three reporting zones relevant for the Canadian Prairies (ECCC, 2022a). Generally, rates of deforestation have declined over time, specifically in the last 10 years. However, when using the annual value of biomass C lost for 2020 (401,000 tonnes of carbon) it results in a calculated loss for such an area calculates as 27.4 t C per ha of live biomass in the year in which it was deforested.

Drever et al. (2021) estimates that between 2010 and 2017, 31% of the 39,000 ha per year of forest conversions in Canada were due to agriculture, or approximately 12,090 ha per year.

Table 29: Area and net change in carbon pools from deforestation (forest land conversion to cropland) reported for 2020. Values include deforestation from the previous 20 years (2001 to 2020). Reporting zones included in the table are relevant zones for the Prairie agricultural region. Adapted from ECCC (2022a). Negative values indicate a net loss or net emission, while positive values indicate a net gain or net removal.

Reporting Zone	Total Area Forest Land converted to cropland (ha)	Living Biomass Net Change (tC ha^{-1})	Dead Organic Matter Net Change (tC ha^{-1})	Mineral Soils Net Change (tC ha^{-1})	Total Change (tCO_2e ha^{-1})	Total Emissions or Removals ($MtCO_2e$)
Boreal Plains	192,240	-1.15	-2.23	2.18	-4.41	-8.47
Subhumid Prairies	50,816	-1.27	-2.40	2.72	-3.50	-1.78

Semiarid Prairies	829	-1.83	-2.45	3.25	-3.80	-0.03
Total	**243,885**	**-4.26**	**-7.08**	**8.15**	**-11.71**	**-10.28**

Potential adoption and GHG emissions benefits

In 2021, Drever et al. estimated the mitigation potential of reducing deforestation across Canada. This estimate was based on Canada's National Inventory Report and anticipates a forest conversion rate of 30,689 ±2,085 ha yr^{-1} between 2021-2030 if current trends continue. An estimated 20,143 ± 637 ha yr^{-1} of that projected rate could remain in forested land if conversion to agricultural land was prevented and conversion from all other drivers was reduced by 50% (Drever et al., 2021). Such preservation would result in potential GHG savings of 3.8 Tg CO_2e yr^{-1} and a cumulative savings of 26.3 Tg CO_2e by 2030. Between 2030-2050, the cumulative emissions savings are another 37 Tg CO_2e. 58% of those emissions savings come from the elimination of deforestation for agriculture (Drever et al., 2021).

Based on the rates of forest conversion to cropland within the NIR over the past 20 years, the rate of GHG emissions for the Prairies were 11.7 tCO_2e per hectare. GHG emissions include living biomass, dead organic matter, and mineral soils for the three relevant reporting zones for the Canadian Prairies, outlined in Table 29.

There is a lack of information about the decision-making process farmers use to decide whether to clear treed areas for agricultural production, but it is presumed to be primarily an economic decision. Farmers and landowners adopting the BMP of eliminating or avoiding conversion of forests is not something explicitly tracked, nor is it properly valued in the commodities produced on farms. Therefore, the decision to adopt the BMP depends on individual farmer decisions around economics, farm profitability, and personal preferences for land management. Increasing commitments by growers and other landowners to avoid converting forested lands to other uses will be challenging and likely require policies that counter the economic incentives of conversion.

Barriers to adoption

The biggest barrier to adopting a reduced deforestation BMP in agriculture settings is the foregone profits of converting the lands to more economically productive systems such as pastures and grain production. Farm Credit Canada reports the following 2021 land value prices on different prairie agriculture regions which are driven by economic and agricultural potential (Table 30):

Table 30: Average Agriculture Land Values for Prairie regions bordering forested lands

Province	**Region**	**Value (CAD/ha)**	**Value Range (CAD/ha)**
Alberta	Peace	5,900	3,000 – 8,600
Alberta	Northern	8,900	3,900 – 17,300

Saskatchewan	North Western	5,700	2,500 – 8,600
Saskatchewan	North Eastern	5,900	2,700 – 8,600
Manitoba	Parkland	6,400	3,200 – 10,100
Manitoba	Interlake	7,900	2,500 – 10,400
Manitoba	Westman	7,600	3,700 – 10,400

Source: 2021 Farmland Values Report, Farm Credit Canada

Drever et al provide an economic analysis which estimates the opportunity costs of not converting the forested lands to agriculture, based on 2018 land values for various regions (both forested and not). Marginal GHG abatement costs range from 0 to nearly CAD 60 /Mg CO_2e from foregone land rental opportunities (Drever et al., 2021).

Other barriers to adopting the BMP of reducing or eliminating deforested areas includes cultural and technological trends. Farm decisions are made based on many different potential factors (Karali et al., 2014), including the behaviours of neighboring farmers. Farmer peers may band together to share the costs associated with tree clearing. Average clearing costs in western Canada are estimated around CAD 2000/ha (Drever et al., 2021), so sharing in the rental costs of the required heavy machinery or transportation fees of equipment brings such costs down and may act as cultural/economic driver of deforestation from an individual landowner's perspective.

Land obstacles such as trees and rocks affect a farm's operational efficiency and the convenience at which field operations can be executed, further driving preferences for homogenous fields. This preference may be contributing to deforestation activities on top of the other economic and socio-cultural incentives, adding to the barriers of no-conversion beneficial management practices. However there was not sufficient peer reviewed evidence to support this claim conclusively.

Co-Benefits

Many co-benefits are associated with reducing conversion of forested areas to agriculture. Drever et al outline these benefits on the basis of improvements to air, biodiversity, soil, water, and social aspects (Drever et al., 2021).

Air quality is positively affected from forests through the removal of air pollutants through leaves and capturing particulate matter on the physical surfaces of the trees. In cities, where air pollution is the most highly concentrated, the value of trees for human health thanks to their removal of pollutants was estimated between CAD 52.5 – 402.6 million (Nowak et al., 2018). Forests are also effective at reducing ozone concentrations by removing O_3 and NO_2 gases (Kroeger et al., 2014).

Biodiversity benefits of forested areas are enormous (Buotte et al., 2020). The Western Boreal Conservation Initiative monitors wildlife species and populations and has shed light on the importance of maintaining forest cohesion for their biodiversity of insects, birds, plants, mammals, and more. Maintaining forests is a key part of the Canadian Government's strategy to conserve 30% of lands and oceans by 2030, in part for their biodiversity benefits.

Forest soils provide a multitude of functions and help support terrestrial ecosystems. Conserving forested lands will help maintain soil biota and the incredibly rich species of microbes living within it (Motiejūnaitė et al., 2019), as well as maintain the nutrient, water, and carbon cycling which supports all life in forested areas (Jurgensen et al., 1997).

Reducing deforestation provides considerable social value (Lim et al., 2015). Public surveys have shown that the aesthetic value of forested areas is of high importance for Canadians, as is their value for recreational use, religious and spiritual importance, ecotourism, education, and much more (Lim et al., 2015).

Trade-offs

The primary trade-off from implementing this BMP is the opportunity costs of conserving the forests instead of transforming it into a more profitable agricultural landscape, as was discussed in the barriers to adoption section. Furthermore, preserving forested areas at the expense of agricultural production results in either less agricultural commodities or may lead to 'leakage', where crop and livestock production is more intensified on existing agricultural lands.

Another possible trade-off includes a potentially lower albedo effect from conserving forests when compared to grasslands and cropping systems, but this trade-off depends on the type of transition of forest to field, being most pronounced when the transition is from a dark conifer canopy to light green crop cover.

Knowledge Gaps

Strategies which encourage the preservation of forested areas are still not properly understood and designing policies to address the economic incentives of individual land-owners is a challenge. Assessing the ecological and economic value of forested areas against agricultural lands is not simple. Both systems have a fundamental importance to individuals, companies, provincial governments, etc., which makes overcoming deforestation particularly difficult. The economic/profit gap between crop and livestock production and forested lands needs to close if deforestation is to be voluntarily halted. The types of economic incentives that motivate forest conservation by farmers, such as tax breaks, payments for ecosystem services or diversified farm revenue from non-tree forest products, require more study.

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13. Reduce loss of Woody Biomass in Agriculture (Avoided Conversion of Shelterbelts)

Description

Similar to the two forest-related BMP's described previously on increasing and managing trees in agricultural landscapes and reducing forest land conversion to agriculture, conserving the current woody biomass in agricultural landscapes (i.e., shelterbelts) is important for improving the GHG balance of Prairie agriculture.

Throughout the 20th century and until 2013, many federal programs provided trees for farm plantings of shelterbelts to reduce soil erosion and improve water conservation. Since these programs, many of these trees have not been managed properly, causing them to deteriorate and become unhealthy, prompting their removal. Many Prairie farmers also find trees inconvenient for operating large farm machinery, and view that shelterbelts are less relevant due to the adoption of no-till.

Effect of Shelterbelts on GHG Emissions

In addition to the loss of biomass carbon from healthy trees, there is a loss of SOC from removing trees from the landscape. The loss of C stocks was estimated to be 85.6 t C km^{-1} but that loss was partially offset by 7.3 t C km^{-1} equivalent due to change in albedo (Drever, 2021) Table 31 provides estimates of range of C sequestration rates for new shelterbelts.

Table 31: *Carbon sequestration rates for new shelterbelts.*

Tree Age	Carbon sequestration rate per hectare of treed land (Mg C ha-1 yr-1)	Reference
5-100	Hybrid poplar: 3.2-5.2 Caragana: 1.3-2.9 Green ash: 2.0-3.9 Manitoba maple: 2.8-5.3	(Amichev et al., 2016)
40	SOC: 0.15 to 0.9	(Amadi et al., 2016)

Potential impact of Avoided Conversion of Shelterbelts on Prairie GHG emissions

Current Adoption

Amichev et al. (2020) estimated there are over 51,000 km of shelterbelts within Saskatchewan, of which 2,491.2 km were lost between 2008 and 2016 (~4.9%). Extending these rates to rest of the prairies, Drever et al. (2021) estimated a potential to reduce loss of existing shelterbelts by 586 km yr^{-1} (227, 303, and 56, km yr^{-1} in Alberta, Saskatchewan, and Manitoba, respectively). They assumed the avoided loss would be accomplished from rejuvenating existing shelterbelts with replacement of unhealthy trees with new trees.

Potential adoption and GHG emission benefits

Drever et al. (2021) estimated total potential C conserved from avoided shelterbelt conversion in Prairies as 0.31 $MtCO_2e\ yr^{-1}$. Given the barriers to adoption, they did not consider there was much potential for new shelterbelt plantings. In their analysis, Drever et al. (2021) assumed that farmers would have to be compensated for full costs of shelterbelt renovation and maintenance. They estimated only about 0.2 $MtCO_2e$ per year based on income to the farmer based on GHG-based incentives of \$50 and \$100/t CO_2e.

Barriers to adoption

Drever et al. (2021) report reasons why farmers are not interested in maintaining or planting shelterbelts. The main reasons for not maintaining or planting shelterbelts were the cost and labour for planting and maintenance. Many farmers did not want to lose the crop production potential of land taken up by shelterbelts. Competition between shelterbelts and crops was another reason. Finally, the inconvenience of using large farm machinery provided another reason not to have shelterbelts.

Co-benefits and trade-offs

Kulshreshtha and Rempel (2014) identified a range of co-benefits including habitat for biodiversity, landscape aesthetics, erosion control and storm protection. They concluded that the public benefits of shelterbelts for non-GHG reasons exceeded their private benefit to farmers. The private benefits come from the more favourable microclimate provided by shelterbelts (Kort, 1988), and dependent on the shelterbelt species, berries and fruits.

The trade-offs are primarily the loss of productive farmland and competition with adjacent crops.

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14. Avoided Conversion of Grassland, Pasture and Hayland

Description

Grasslands store vast amounts of carbon as soil organic matter generated through the input of both the above- and the below-ground plant biomass. Thus, grassland soils are significant reservoirs of organic carbon that will, if left undisturbed, continue to store this carbon belowground. However, when grasslands are disturbed for crop cultivation, a large portion of the stored carbon will oxidize over time, releasing CO_2 into the atmosphere (Spawn et al. , 2019).

Early land management practices for cultivation such as including frequent plowing and fallow were devised to accelerate soil carbon decomposition to "mine" plant nutrients that are bound by the soil organic carbon. Soil carbon decomposition and associated CO_2 emissions can be avoided by preventing this accelerated decomposition (Ahlering et al. , 2016). Hence, avoiding conversion of grassland, pastures and haylands to cropland, thereby preserving soil carbon stocks, has been identified as an important opportunity as a nature-based solution to mitigate GHG emissions in Canada (Drever et al., 2021). However, the net benefit depends on how conservation of lands in perennials used as feed sources for ruminant livestock affects the total livestock population and their total GHG emissions (Liang et al. 2020)

Conversely, if cultivated lands are converted back into grassland ecosystems with good management, the soil carbon store may be able to recover to some extent. The extent of the recovery largely depends on restoration activities, grassland management, grassland species, natural disturbance, underlying soils, and the extent to which the soil carbon store was depleted. Drever et al. (2021) examined the potential for riparian grassland restoration of 30m around pothole wetlands and other water bodies in the Prairies, but not beyond the riparian areas. See 'Note on Grassland Restoration' below for more information.

Effect of Grassland Conservation on GHG emissions

The stabilization or loss of soil carbon from modified plant communities is affected by soil texture, temperature, moisture and other climate factors, as well as the quantity and chemical characteristics of plant residues produced. Reduced soil organic carbon under cropping is partly caused by root mass declined from certain crops, contributing to less soil organic carbon (Dormaar, Adams, & Willms, 1994).

The relative carbon sequestration potential of cultivated land, tame grasslands and native grasslands affect both avoided conversion and restoration activities. Native grass has been characterized by its high root biomass, especially in the deeper rooting zone, and may result in higher organic carbon storage than tame grassland. However, a search of literature did not find sufficient evidence to confirm, or to quantity this assertion. A study conducted at 3 sites in southern Alberta compared six agricultural cropping practices including monoculture and mixture of grass species (Jefferson, 2010; Whalen, 2003). The study showed that when native grassland was converted to tame grass the stores of soil carbon were more stable and had lower losses than when converted cropland. The native and the introduced grass were similar in productivity and root biomass. Potentials for soil organic carbon sequestration under these plant communities are similar. In some cases, the tame perennial grasses are easier to establish than the native grasses (Jefferson, 2010). Therefore, this review presents the same values for avoided conversion of both native and tame perennial grasslands.

In terms of measurements on the ground, research plot studies in Alberta measured a rate of carbon loss of 6.2 t CO_2e /ha/year for the first four years, followed by a slower rate of 1.17 t CO_2e /ha/year for the next 9 years associated with a switch from native grassland to continuous wheat cropping (Wang e. a., 2010). However, extrapolation of on-the-ground measurements is complex and involves significant uncertainty.

Long-term studies of SOC stock change in the Canadian Prairies (Wang, VandenBygaart, & McConkey, 2014) determined that managed temperate grasslands sequestered carbon at rate as high as 2. 1 tCO_2e/ha/yr for the first 20 years, but can also be a net GHG source (see below). The average rate is 0.7 tCO_2e/ha/yr for the 50 years following restoration. The net carbon sequestration rate will decrease with time and eventually it will reach an equilibrium state.

N_2O Emissions

Grasslands, pasture and forages supply nitrogen to the soil through legumes when present, and converting perennial grasslands to annual crop production affects the flux of N_2O emissions from the soil. Mielenz et al observed very high N_2O emissions following pasture to crop conversion in a first-year wheat system in Australia. Even without additional nitrogen fertilizers, N_2O emissions were as high as 48 kg N_2O-N per hectare in the first year following perennial pasture conversion (Mielenz et al., 2017). A Manitoba study assessing N_2O emissions resulting in the conversion of forage grass to annual crops showed a similar N_2O spike after the land-use change (Adelekun et al., 2019). The increase in N_2O emissions will decrease over time following the land use change as the organic materials decompose and the soil processes (nitrification and denitrification) reach a balance.

In 2009, three grassland fields in the USDA's Conservation Reserve Program were changed to row crops using no-till methods in some areas and conventional tillage in others, while a fourth field was kept in grassland as a control plot. All converted fields saw increased N_2O and CO_2 fluxes after herbicides were applied, and the conventional till treatment saw a much more substantial increase in both greenhouse gases following the tillage event. The researchers also observed N_2O fluxes for 201 days following the conversion and recorded daily average N_2O fluxes (g N_2O-N ha^{-1} d^{-1}). The conventional tilled plots were by far the highest emitting (47.5), with the no-till converted plots following (16.7), and the grassland control plot emitting the least (2.51). When considering both the N_2O and CO_2 emissions of the three different systems, the global warming impact followed the same order: Conventional tillage conversion = 11.5 Mg CO_2e ha^{1-}; no till conversion = 2.87 Mg CO_2e ha^{-1}; grassland control plot = (-)3.5 Mg CO_2e ha^{-1} with continued mitigation (Ruan & Philip Robertson, 2013). These results demonstrate similar patterns found in other studies, like Grandy and Robertson (2006) who showed substantial N_2O emissions following perennial grassland conversion. N_2O emission increased 3.1-7.7-fold over a three-year period in that study (Grandy & Robertson, 2006).

Other emission impacts

Methane emissions were monitored in the Ruan and Philip Robertson study (2013), however the differences in CH_4 emissions between treatments were unimportant in comparison to the CO_2 and N_2O effects. Liang et al. (2020) analyzed the relationship between cattle population and hay and pasture area in Canada. They analyzed the emissions per head of cattle with the SOC loss from conversion of pasture and hayland to annual crops. They concluded that the SOC losses were 62% of the emissions of the cattle associated that area. Therefore, if avoided pasture and hayland conversion maintains a cattle herd size historically associated with that area, Canada net GHG emissions will increase. Avoided conversion is only beneficial for GHG emissions if that does not induce a cattle herd larger than it would have been without the avoided conversion.

Potential impact of Grassland Conservation on Prairie GHG emissions

Current adoption

Avoided conversion and restoration activities on grasslands are pertinent across most of Canada, from the Prairies (AB, SK, MB) to Ontario, Quebec and the Maritime (PEI, NL, NS) provinces (Drever et al., 2021; WWF, 2021). Saskatchewan represents the highest potential area and mitigation outcome for avoided conversion over the next decade with 3.41 million tCO_2e/yr (Drever et al., 2021). The large available land base of existing and converted grasslands is an important contributing factor to the potential of grasslands as a significant climate mitigation tool.

Potential adoption and GHG emissions benefits

Based on the most recent Census of Agriculture (2011 and 2016) (Statistics Canada, 2016) conversion of more than 2.5 Mha of native grasslands and managed pastures and haylands to cropland can be avoided between 2021 and 2030, primarily in the Prairie Regions. This includes all of the "natural land for pasture", "tame or seeded pasture", and "all other tame hay and fodder crops" classifications by the Canadian Census data. If this conversion could be avoided completely, Drever et al estimate cumulative 12.7 Tg CO_2e/yr by 2030 and 4.1 Tg CO_2e/ya by 2050 (Drever, et al., 2021).

Conserving natural grasslands are particularly important for biodiversity conservation. The Canadian Government can play a role in conservation of grasslands and has committed to conserve 25% of total lands by 2025 and 30% by 2030. Despite ECCC acquiring nearly 81,000 hectares of native prairie to manage in southwest Saskatchewan, only 6% of temperate grasslands have some form of protection (Canada. Environment and Climate Change Canada, 2022).

Barriers to adoption

Conversion of grasslands to alternative land-use types typically arise from economic pressures (Rashford et al., 2011). Conversion to crop land is a major driver with crop production being more profitable than livestock production in many regions and landscapes (Statistics Canada, 2022). The number of cattle on Canadian farms and feedlots have declined by over 2 million between 2009 and 2019, while crop production designated area continually increases (Statistics Canada, 2022). Overcoming the profit differential between cropped and perennial grazed lands is an important barrier for producers on the prairies.

A lack of understanding or appreciation for the climate mitigation benefits and ecosystem services of maintaining perennial grasslands may also be a barrier among both farmers and policymakers.

Co-Benefits

There are many co-benefits to grasslands as they provide a host of ecosystem services (Pilipavičius, 2015). Beyond the carbon sequestration, perennial grasslands are important for providing habitat and feed for wildlife (including many species considered at-risk) and managed livestock, improving soil quality, purifying water, supporting diverse plant and insect species, providing important medicinal and biotechnological genetic material, and can be a sustainable feedstock for bioenergy, and much more.

Trade-offs

The trade-offs are significant and make protection of land in perennial pasture or hayland complicated.

If the avoided conversion of grassland, pasture, and/or hay also provides low-cost feed source that prevents a reduction in the cattle herd that would have occurred otherwise, then the emissions of methane from that avoided loss of cattle can have larger radiative forcing increase than the decrease from the SOC conserved (Liang et al., 2020)

An important trade-off associated with preserving, maintaining, or protecting perennial grasslands is the opportunity cost from alternative land-use methods. When crop production is more profitable than grazing livestock, then the conversion will be more profitable than the retaining for grazing animals. This results in forgone profits for producers and landowners when high-quality land is kept in grasslands as opposed to cultivated to annual crops.

Having the perennial area set aside from production accomplishes avoided conversion without preventing any reduction in cattle herd but that also has two trade-offs. The plant community health and total biodiversity existing in natural grasslands is greatly reduced by not have grazing animals (Lwiwski et al. , 2015). Losing tame pasture, that has less biodiversity value as natural grassland, while preventing the loss of natural grasslands for grazing, can mitigate the former trade-off. The other trade-off is that keeping agricultural land out of production can lead to claim that that action is a cause of indirect land use change elsewhere in the world to make up for the lost production.

Knowledge Gaps

Grasslands can function as reservoirs for soil carbon or sources for CO_2 emission in the global carbon cycle. It depends on how grasslands are managed or impacted by natural events and human activities. They could have either a net negative or a net positive impact on the climate depending on a number of factors. Assessing these changes generally requires a modelling approach, which is not easy and the results are far from being definitive. Uncertainty of estimated soil organic carbon emissions, using methods developed in McConkey et al. (2007) for transitions from perennial to annual crop, were given different values for eastern and western Reporting Zones. The uncertainty for the mitigation rate was assessed to be 54% for the eastern region and 78% for the western region (Drever et al., 2021). More research-standard measurements and stratification across the many variables contributing to this uncertainty is needed to refine the emission factors, reduce the associated uncertainty, and expand scientific knowledge of how different variables impact site-specific carbon sequestration rates.

Regarding costs, it is clear that fair market value is the main driver of conservation agreement costs (assumed to be equivalent to opportunity cost from avoided conversion), but the conservation agreement value may not be based on assumed conversion to cultivation (as represented by the emission factors). It is also clear that many local factors affect fair market valuations which makes it impossible to regionalise costs.

Additional note on measurements: the use of measurements and models in estimating carbon sequestration rates in grassland soils is hotly debated and ongoing. The significant variation in carbon stock measurements in grassland soils over small (<10m) distances that many measurements are required to obtain an accurate estimate of soil carbon stocks. Added to this, stock change over time (i.e. rate of carbon sequestration) is very marginal and often within the confidence limits and noise of the sampling and measurement procedures; quantification of soil carbon sequestration rate requires even more samples to be taken to reduce uncertainty and noise to a point that a change over time can be detected. It is therefore often impossible to detect a change in soil carbon stocks in grasslands over timescales less that 3-5 years. Measurement data is therefore restricted to only a few longer-term research sites where site-specific variation in land management, soil types, climate and weather make it difficult to extrapolate measurements to broader areas. For these reasons, it was decided that the modelling approaches used in the development of Tier 2 emission factors for Canada's National Inventory Report could be better applied to the national context of this study.

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15. Wetland Conservation and Restoration

Description

Functional freshwater mineral soil wetlands (FMWs) absorb carbon via complex biological processes involving aquatic vegetation and anaerobic bacteria (Bansal et al., 2021). The anaerobic conditions created by wetlands reduce decomposition rates causing net sequestration of carbon, but also cause the production and emission of methane (CH_4) as a product of decomposition. The interplay between rates of GHG emissions and carbon sequestration determine whether an FMW contributes a net warming or net cooling effect on the climate (Gleason et al., 2008). In general, wetlands that are undisturbed or have been in existence for some time have a net negative impact on GHG emissions, while recently restored wetlands tend to have a net positive GHG emissions forcing impact due to elevated CH_4 emissions which diminish as ecological systems mature (Mitsch et al., 2015). However, there is significant variability due to wetland size/class, underlying soils, seasonal or annual weather, topography and land management.

Freshwater mineral wetlands are present all over Canada. In many areas of the Prairies and southern Canada, wetlands were drained for agriculture or urban development during initial settlement of Canada. This has led to significant loss of freshwater wetlands in many regions across Canada, with significant loss of soil organic carbon in the Prairie Pothole Region (PPR) specifically from wetland draining and cropping (Bansal et al., 2021). It is estimated that there is a potential area of 250,000 ha for wetland restoration in Canada (Drever et al., 2021). The majority of the degraded or converted freshwater mineral wetlands are in the southern region of Canada, as these regions have the highest rate of human-related disturbances.

Marshes are typical throughout the Prairies, which are dotted with ponds and lakes with marsh vegetation known as the PPR (Haber, 2015). The 750 000 km^2 PPR stretches across southern Alberta, Saskatchewan, and Manitoba, and is characterized by millions of depressions varying in size and depth. However, in settled areas of Canada, up to 70% of wetlands have already been drained or degraded. In southern Manitoba, the wetland extent in 2008 was 77% of pre-1968 area, with an estimated 164,623 ha that could be restored (Pattison et al., 2011). Due to the size of the PPR, there is a large opportunity for restoration within Saskatchewan and Alberta as well. Most of the potential area for wetland restoration is within the PPR, with over 200,000 ha of potential (Drever et al., 2021).

Ducks Unlimited Canada (DUC) estimates 70% of Prairie wetlands have been lost or altered relative to their historical extent, and that since the 1950's, more than 500,000 hectares of Prairie wetlands have been lost. GHG emissions associated with the loss of carbon stores within FMW soils and biomass, and lost carbon sequestration capacity provided by FMWs, are correspondingly significant.

Effect of Wetland Conservation and Restoration on GHG emissions

CH_4 Emissions

Wetlands are significant sources of CH_4, which reduces the potential for GHG mitigation. Fluxes of CH_4 from these systems vary dramatically both spatially and temporally, and have been related to various hydrologic and climatological controls such as temperature, soil moisture, and degree of inundation (Crill, Harriss, & Bartlett, 1991) (Altor & Mitsch, 2008) (Batson, et al., 2015). For example, the wet edge, basin, and wetland pond have the highest rates of CH_4 emission, with upland, lower to upper slope and riparian areas producing the lowest CH_4 emissions

(Badiou et al., 2011). Additionally, other factors such as the trophic state of a wetland, the quality of substrate, sulphate concentrations, and vegetation community all play an important role in regulating the production and release of CH_4 (Pennock, et al., 2010) (Batson, et al., 2015) (Segarra, et al., 2015). There is significant uncertainty about the scale and impact of wetland CH_4 emissions due to the various measurement techniques used, small number of studies, and variation among sites, seasons and regions in published studies.

N_2O Emissions

Similar to methane, nitrous oxide (N_2O) can also be produced from wetlands. The water-saturated and anoxic environment found in permanent freshwater mineral wetlands mean that N_2O emissions are typically a minor component of the overall GHG emissions from these systems (Blais, Lorrain, & Tremblay, 2005). However, less permanent wetlands that alternate between wet and dry cycles, such as the ephemeral and seasonal wetlands found in the prairie pothole region of North America (PPR), can emit substantial amounts of N_2O when wetland soils begin to dry (Pennock, et al., 2010) (Tangen, Fionocchiaro, & Gleason, 2015). Nitrogen-loading into wetlands, for example via fertilizer applications to surrounding cropland, can contribute to increased N_2O emissions (Tangen, Fionocchiaro, & Gleason, 2015). However, there is a lack of literature available to adequately quantify N_2O emissions on freshwater wetlands.

Soil Carbon

The sequestration rate or change in soil organic carbon (SOC) and GHG emissions produced (CH_4 and N_2O) determine whether a wetland is a net sink or source. These rates are often reported separately in the literature and vary by region and wetland characteristics (such as soil type, water table level, amount of organic matter, etc.). Sequestration rates in freshwater wetlands vary significantly in the literature, which is attributed to wetland characteristics and the uncertainty in different measurement methods. For example, the riparian zone, wet edge, and basin of a landscape provide the highest stocks of SOC in reference and long restored wetlands (Badiou et al., 2011). Sequestration rates vary by the size of the existing carbon pool, which fluctuates on many factors(Badiou et al., 2011).

Potential impact of Wetland Restoration on Prairie GHG emissions

Potential adoption and GHG emissions benefits

Table 32 presents the available literature on wetland restoration that is applicable to the Canadian Prairie context, with mitigation potentials only including studies that included SOC. Net mitigation potential values include the increase in carbon sequestration and the increase in methane emissions after restoration. All values in Table 32 exclude avoided emissions from a cropping situation in the baseline. If cropping emissions are assumed to occur in the baseline, then 0.32 $tCO_2e/ha/yr$ should be added the net mitigation potential emission factor (Agriculture and Agri-Food Canada, 2016).

The most recent study was conducted on 549 wetland sites in the US portion of the PPR and characterized the differences between landscape position for restored, natural and cropland wetlands (Tangen and Bansal, 2020). The rate of sequestration for restored sites was based on the SOC samples from 0-30 cm depth on sites ranging from 1 to 35 years after restoration, with 83% of the sites restored less than 15 years ago (Tangen and Bansal, 2020). The rate of carbon sequestration was lower than other studies conducted in the PPR likely attributed to the age of restored wetlands (Tangen and Bansal, 2020). Previous work in the southern PPRfound SOC rate of increase was only 0.08 Mg C ha^{-1} after 4 years of restoration (0.29 tCO$_2$e ha^{-1}), not including GHG emissions (Zilverberg et al., 2018). This finding was confirmed by Tangen and Bansal (2020), where restored wetlands can be a net source of GHG emissions in the few years after restoration and could take 20-64 years before restored wetlands return to natural conditions.

Based on the available literature, net mitigation potential for restored wetlands is 6.08 t CO$_2$e/ha/yr, which is based on a 33-year period after wetland restoration (Badiou et al., 2011). This value is based on measurements across the PPR of Canada and was recalculated from the presented value in Badiou et al. (2011), due to the updated methane emissions factor (IPCC, 2013). A conservative potential range for mitigation in the Canadian Prairies restored wetlands is between -2.55 and 7.36 tCO$_2$e/ha/yr (see Table 32). This range was developed based on the available literature similar to the regions of Canada that have a high distribution of FMWs, most notably the PPR and the Great Lakes region and therefore includes several types of FMW's that exist in the Canadian Prairies.

Table 32: Carbon sequestration rates and net mitigation potential - various wetland restoration studies relevant to the Canadian Prairies. Note: for consistency across studies, the Drever et al., (2021) net mitigation potential was recalculated using a global warming potential (GWP-100) of 25, instead of 45.

Region	**Period**	**Carbon Sequestration Rate (t CO$_2$e ha^{-1} yr^{-1})**	**Net Mitigation Potential* (t CO$_2$e ha^{-1} yr^{-1})**	**Reference**
Canada	40 years	8.07	4.24	Drever et al., 2021
Prairie Pothole Region (CAN)	After 1-3 years	9.17	5.34	Badiou et al., 2011
Prairie Pothole Region (CAN)	After 7-12 years	22.37	18.54	Badiou et al., 2011
Prairie Pothole Region (CAN)	33 years	9.9	6.08	Badiou et al., 2011
Ohio, USA	After 10 years	6.97	3.14	Anderson & Mitsch, 2006
Ohio, USA	After 15 years	8.87	5.05	Bernal & Mitsch, 2013
Prairie Pothole Region (USA)	After 10 years	11.18	7.36	Euliss Jr. et al., 2006
Prairie Pothole Region (USA)	After 1-35 years (60% of sites were less than 10 years)	1.28-4.03	-2.55-0.21	Tangen and Bansal, 2020

**Includes methane emissions of 3.825 t CO_2e $ha^{-1}yr^{-1}$ for temperate restored freshwater mineral wetlands (IPCC, 2013).*

Potential impact of Wetland Conservation on Prairie GHG emissions

Current and Potential adoption and GHG emissions benefits

Drever et al. (2021) estimated that over the 2020-2030 period, there is potential for nearly 235,000 ha of wetland conservation in the Prairies, with a majority occurring in the Prairie pothole region of Saskatchewan (~160,000). Southern Manitoba's current wetland extent of 422,542 ha was estimated to be a net sink of over 4M tonnes of CO_2e between 2008 and 2020 (Pattison et al., 2011). This equates to an ongoing sequestration rate of 339,250 tCO_2e per year, or 0.803 tCO_2e/ha/yr for wetland retention (Pattison et al., 2011). Alberta's historical loss of freshwater mineral wetlands was 294,000 ha in the Prairie and Parkland cropping regions, which has been estimated to account for emissions of 96M tCO_2e from the loss of soil organic carbon, equivalent to over 300 tCO_2e/ha (Creed, Aldred, Serran, & et al., 2017).

Net GHG emissions associated with wetlands are largely driven by both site-specific variables and methane emissions assumptions or measurements. For this reason, it was decided to utilize generic national-level data as described in Drever et al. (2021) (Badiou, 2021*, personal communication*). The authors of that research described a 2030 emission factor of 10.57 tCO_2e/ha/yr (after an assumed 10-yr period of implementation starting in 2021) and a 2050 emission factor of 0.0 tCO_2e/ha/yr (after 30 yr of implementation). However, the mitigation potential is different when considering a GWP-100 of 25 for methane and including the lost additional carbon sequestration throughout the baseline wetland conversion in the first 20 years after conversion. This yielded emission factors of 18.92 (13.03-24.21) tCO_2e/ha/yr in 2030 and 2.62 (-3.04-8.31) tCO_2e/ha/yr in 2050. This mitigation potential value is estimated based on the avoided loss of SOC from wetland drainage, the avoided emissions from croplands (0.32 tCO_2e/ha/yr), and an IPCC Tier 1 emission factor for avoided methane emissions (Blain et al., 2013). Emissions from ongoing cultivation operations in the project scenario are included since it is assumed that the major threat to wetlands is drainage and conversion to cultivated land in the Canadian Prairies. It is estimated that the annual mitigation potential for avoided conversion of wetlands is much higher than the potential mitigation opportunity for wetland restoration (Drever et al., 2021).

While regional emission factors are unavailable, there is some information on existing carbon stocks in existing wetlands by region (see Table 33). It could be expected that wetland soils with larger starting carbon stocks would be expected to lose greater amounts of carbon if they were converted, and vice versa for wetlands with lower initial soil carbon stocks. The following information is provided to give an indication of comparative regional differences in initial wetland carbon stocks, but should be treated with caution as local variability is likely to have a much more significant influence on these values.

Table 33: Published figures for freshwater mineral wetland carbon pool size across sites in the Prairie Pothole Region.

Type/Description	Carbon Pool (t CO_2e ha^{-1})	Region of Study	Reference
Native, Dark Brown Soil	642.1	SK, CA	Bedard-Haughn et al., 2006
Uncultivated, Dark Brown Soil	701.0	SK, CA	Neuman and Belcher, 2011
Uncultivated, Dark Brown Soil	338.8	SK, CA	Bedard-Haughn et al., 2006
Uncultivated, Brown Soil	518.5	SK, CA	Neuman and Belcher, 2011
Cultivated, Dark Brown Soil	319.8	SK, CA	Bedard-Haughn et al., 2006
Natural, Wetland Inner and Transition landscape positions	285.4	PPR*, USA	Tangen and Bansal, 2020
Cultivated, Wetland Inner and Transition landscape positions	203.5	PPR*, USA	Tangen and Bansal, 2020

**Multiple locations across the US Prairie Pothole Region including Montana, North Dakota, South Dakota, Minnesota, and Iowa.*

Barriers to adoption

The barriers to adoption of wetland conservation or restoration are generally similar, with the major barriers consisting of costs, landowner disinterest and feasibility constraints.

A significant barrier to adoption for wetland restoration is the cost of restoration and maintenance of the wetland. For the South Tabacco Creek watershed in Manitoba, the estimated cost of wetland restoration was between $100-106 per ha per year for 28.8 and 80.1 ha wetland projects under two different financial scenarios (Yang et al., 2016). The cost of wetland restoration in the Smith Creek watershed in Southeastern Saskatchewan was estimated to be $13,585 CAD per ha (estimate by Ducks Unlimited Canada) of up-front expenditure. A Canada-wide estimate found that costs were estimated to be $278 per ha per year due to the initial upfront cost of restoration, which over a 40-year wetland restoration project equates to $11,120 per ha (Drever et al., 2021). For wetland conservation, costs are generally lower is associated with taking land out of production or opportunity costs associated with agricultural production. Drever et al. (2021) estimated an average cost for avoided conversion of wetlands of $527 per ha based on transaction costs only from unpublished data from Ducks Unlimited Canada. Management costs for habitat improvement were estimated at $34/ha/yr on top of the initial transaction cost. If only considering the opportunity cost of conserving wetlands, it would cost $88.01 per hectare to retain wetlands based on the agricultural land rental prices in Saskatchewan. The cost of avoided conversion of wetlands is significantly less than restoration and provides a higher mitigation potential (Drever et al., 2021). Ecosystem conservation costs range across the country and by conservation project type. The available cost information was on a project basis, meaning that the cost per ha includes wetland conservation and conservation of the surrounding uplands. In the PPR, cost of conservation is between $1,172-2,437 per ha across certain areas of Manitoba, Saskatchewan, and Alberta according to NCC estimates (NCC, 2021).

In a living laboratories study based in Alberta's Nose Creek watershed, researchers found that the primary barriers to wetland restoration and conservation were structures and processes that made wetland restoration impractical (Clare and Creed, 2022). Firstly, they found that the burden on rural landowners to conserve or restore wetlands for benefit of nearby municipalities was seen as unfair, which led to a disinterest and challenge of obtaining landowner consent for wetland restoration. The experience also confirmed previous work found relating to on-farm producers being more willing to participate in conservation or restoration programs as compared to owners that live off-farm (Stroman et al., 2017, Wachenheim et al., 2018). Convincing landowners to participate in a project increases resources necessary as enthusiasm for wetland restoration is typically low in agricultural landscapes (Clare and Creed, 2022).

Other barriers that contributed to feasibility for restoring wetlands in the Alberta Nose Creek watershed related to the companies available and regulatorily allowed to conduct restoration activities (Clare and Creed, 2022). A structural barrier by the Alberta provincial government provided a challenge since only one organization, Ducks Unlimited Canada, was able to conduct the restoration activities, regardless of their capacity limitations. In addition, the regulatory regime and associated processes was not designed to quickly secure permits for restoration activities. This delay also led to challenges keeping landowners within the program, since regulators significantly slowed the process and did little to increase trust with landowners. Finally, the regulatory priorities were not aligned with the wetland restoration program, as mid-way through the program the provincial government changed their eligibility criteria and required that non-permitted drainage would be charged by the enforcement agency, creating few legal options for the researchers to restore wetlands in the region (Clare and Creed, 2022).

Co-Benefits

Wetlands provide a range of ecosystem services, making it one of the most ecologically valuable land uses (Mitsch et al., 2015). Despite wetlands only covering 3% of the land globally, they provide approximately 40% of global ecosystem services (Zedler and Kercher, 2005). The many ecosystem services include aquifer recharge, sediment and nutrient retention, biodiversity, and floodwater attenuation (Pindilli, 2022).

Floodwater attenuation in wetlands is one of the most widely recognized co-benefits to intact and restored wetlands, although estimates for water storage potential is unclear across the PPR. Wetlands have the capacity to intercept and store precipitation that contributes to flooding downstream, significantly reducing floodwater risks to agricultural and urban areas (ref.). Across the US portion of the PPR, a water storage rate was estimated to be 0.34 ha m ha^{-1} of water volume that could be stored at wetlands maximum capacity across the regions investigated (Gleason et al., 2008). Floodwater retention services have shown to improve when vegetation is increased through restoration practices, which also contributes the climate mitigation potential (Pindilli, 2022). Floodwater retention in Prairie Pothole wetlands provides numerous benefits including downstream floodwater reductions, nutrient and sediment retention, and erosion reductions (Gleason et al., 2008).

Wetlands provide a reduction in sedimentation and nutrient loading in water courses (lakes, streams, and rivers) downstream due to the accumulation of suspended nutrients and sediment from runoff (Pidilli, 2022). This service is especially useful in agriculturally intensive regions, as nitrogen and phosphorus loading into water bodies contributes to eutrophication (Land et al., 2016).

Conservation and restoration of wetlands also enhances the biodiversity of the region in comparison to agricultural lands in the Prairies. Wetlands provide suitable habitat for a variety of pollinators and grassland species including amphibians, and waterfowl. In Iowa, Mitchell et al., (2022) found that for each hectare of conservation easement (wetland conservation and surrounding riparian buffer habitat), 0.6 hectares of suitable grassland bird habitat and 0.4 hectares of suitable amphibian habitat was gained. Wetland restoration improves the native species richness and quality of plant species in comparison to cropped catchments (Gleason et al., 2008). Improved biodiversity in wetlands is also linked to increased recreation activities such as fishing, hunting, and birding (Gleason et al., 2008). One study in the PPR in North and South Dakota, USA, found that the value of natural wetlands for duck habitat and production exceeded the opportunity costs of cropland conversion (Gascoigne et al., 2011).

Trade-offs

In addition to the ecosystem services wetlands provide, they also provide disincentives for conservation or restoration . Most notably, wetlands provide habitat for mosquitos which contribute to human health hazards and disease transmission (Knight et al. 2017). Prairie pothole wetlands in the Canadian Prairies may also be a nuisance to land owners and agricultural producers, as their depressional characteristics make land management challenging in some cases. The potential trade-offs of restoring or conserving a wetland include the potential leakage for activities occurring elsewhere. Although leakage does not always occur there is a risk that restoring or conserving a wetland from drainage, particularly for agricultural production, may cause land use changes elsewhere to make up for the demand in agricultural land. This potentially harmful trade-off reduces the climate mitigation potential of wetlands in comparison to other natural land uses such as grasslands and forested lands within the Canadian Prairies.

Knowledge Gaps

The accurate estimation of the potential mitigation opportunities provided by freshwater mineral wetlands is very challenging due to a few factors. Firstly, relatively little research exists to elucidate the full range of GHG impacts wetland conservation and restoration within the Canadian PPR. Secondly, where literature is available, data for GHG emissions and carbon stocks are often reported separately and are not conducted long-term. Finally, the quality of sequestration and emissions data that is available for freshwater wetlands is poor due to varying nature of wetland characteristics, methodologies used and the lack of replication in study sites. These three factors contribute to considerable uncertainty around wetland mitigation potential for natural climate solutions and will be discussed further.

Globally, the interest in wetlands research is still developing as the benefit of natural climate solutions is becoming more apparent. Due to the massive degradation and loss of wetlands from human interaction, research in this area is scarce. This has led to a significant gap in the literature relating to freshwater mineral wetlands in Canada. For example, much of the research into GHG emission and carbon sequestration rates has taken place in the Unites States, with only one study from Canada represented in a recent review (Loder & Finkelstein, 2020). Few studies have estimated gas exchange over wetlands in the PPR (Badiou et al., 2011; Neuman & Belcher, 2011; Bedard-Haughn et al., 2006). The available studies often estimate the rate of carbon accumulation or sequestration separately from GHG emissions measurements (CO_2, CH_4 and N_2O).

Data quality for gas exchange metrics is poor for many reasons. Uncertainty in measurements due to methodological differences is key problem for GHG emissions measurements across many landscapes (agriculture, forests, urban). The method used in estimating GHG emissions can provide highly variable measurements due to the variation in scale of measurement types. For example, a majority of GHG emissions measurements are conducted using static chambers, which at most cover a spatial area of $1m^2$ (Rochette et al., 2008; Machado et al., 2019). In addition, chamber measurements capture emissions from a short sampling period, and do not accurately characterize diurnal variation (Machado et al., 2019). These measurements are considerably different from continuous ecosystem level measurements such as those conducted using eddy covariance methods. Many of the peer reviewed values for wetland sequestration potential are estimates, such as Neuman and Belcher, who assumed a 1% increase in soil carbon stocks over a 20-year period as the rate of sequestration (2011). There is also no consensus on the depth at which measurements should be taken for carbon pool or carbon sequestration estimation, meaning studies cannot easily be compared.

Finally, description of sites within the literature rarely includes wetland classes. This makes comparison difficult since wetlands in the PPR can range from temporary or ephemeral to permanent with vastly different GHG profiles (Gleason et al., 2009). The present study has assumed that restoration projects are semi-permanent to permanent wetlands, where avoided conversion projects could include any or all wetland classes. Uncertainty in the data presented demonstrates the difficulty in providing regional emission factors for freshwater mineral wetlands. Regionally similar values for carbon sequestration and GHG emissions are prone to high uncertainty based on wetland classes and site parameters. Therefore, ranges in GHG emissions and sequestration are more attributable to wetland characteristics (soil type, level of inundation, microbial community, etc.) than regional differences.

For broader understanding of the potential mitigation for wetland conservation and restoration, comprehensive studies measuring all relevant GHGs, and soil carbon sequestration must occur on several study sites, representing varying wetland zones and characteristics across the Canadian Prairies. Improved research and data collection regarding the GHG mitigation potential over long-term studies would reduce the uncertainty that currently exists in estimating and reporting wetland-specific incentivization. The lack of comprehensive studies has contributed to poor adoption of wetland conservation, as the full carbon impacts from preventing wetland loss are uncharacterized. In addition, further research on carbon stock data for biogeochemical models, and monitoring technologies such as remote sensing, could provide more incentive for research and conservation (Mack et al., 2021). A greater understanding of the GHG mitigation potential for avoided conversion of wetlands would allow for greater adoption through pricing mechanisms like the voluntary carbon offset market (Mack et al., 2021).

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BMPs Pertaining to Systematic Changes

Systematic changes account for management practice(s) that affect the whole structure of the farming operation and can be felt beyond the farm gate. These BMPs include organic and regenerative agriculture systems, bioenergy from crop residues and carbon capture and storage, and integrating crops with livestock production. BMPs in this section often encompass a set of practices rather than a single farm intervention. The practices within the system sometimes have complimentary effects, while at other times they are at odds with each other.

16. Crop Residue Bioenergy and Carbon Capture and Storage

Description

There is intense interest in bioenergy and bioenergy with carbon capture and storage (BECCS) as a GHG mitigation option. This is identified as the largest land-based emission reduction potential in the US (Robertson *et al.*, 2022). Using agricultural crop residues and a biomass feedstock for bioenergy avoids the food vs fuel debate and associated attributed indirect land used change emissions. Also, there is no land-use change or need to dedicate land area only for production of biomass crops such as switchgrass. Liu *et al.* (2014) showed that agricultural residues were overall the most competitive economically for a cellulosic feedstock for biofuel production in Canada.

In 2020, Canada produced 1.7 B L of ethanol for biofuel while also importing 1.2 B L, entirely from the US (USDA, 2022). The Canadian Clean Fuel Regulation, enacted in July, 2022, is expected to significantly increase the demand for ethanol to blend with gasoline to reduce the carbon intensity of resulting fuel.

The economics of large-scale production of ethanol from processing and fermentation of crop residues on Prairies could became feasible at ethanol prices of \$0.66 L^{-1} for a large 250 ML production plant (Mupondwa *et al.*, 2017). They identified several feasible locations for plants within the Prairies. Zheng *et al.* (2021) showed, for Alberta, that including the year-to-year variability in residue production affects the optimal location and the supply area required.

Although corn stover is the preferred agricultural residue for bio-ethanol due to high ethanol yield per tonne of stover, a review of life-cycle assessments has found that ethanol produced from small grain residues (wheat) was technically and environmentally sound, including net GHG emission reduction when the ethanol was used to replace gasoline (Ingrao *et al.*, 2021).

Dolan *et al.* (2020) analyzed cellulosic from dedicated biomass crops in the US northern Great Plains and determined it was critical to estimate the effects on SOC. In their study, the SOC effects from land-use change greatly reduced the size of GHG reduction from using biomass.

Carbon capture and storage (CCS) removes CO_2 produced during ethanol production and then compresses and/or liquifies the CO_2, transports it to where it can be injected into a geological formation where it will remain stored indefinitely. With CCS, biofuels that are carbon negative are practical (Field *et al.*, 2020; Kim *et al.*, 2020; Cheng *et al.*, 2021; Lask *et al.*, 2021).

Drever et al. (2021) investigated bioethanol produced from agriculture residues without carbon capture and storage. They found it was only economical at either high prices of CO_2e, >CAD280/t CO_2e, or high prices of ethanol that were nearly double current prices. The recent (October 2022) value of credit (1 t CO_2e) for low carbon intensity fuel under the BC Low Carbon Fuel Standard is CAD 447.97 and has been as high as CAD 519.19 during 2021 (BC Government, 2022). The national Clean Fuel Regulation operates a similar carbon-intensity based method as the BC Low Carbon Fuel Standard. So, based on the Drever et al. (2021) analysis, there could be sufficient value for the emission reduction under such carbon-intensity lowering fuel programs to incent cellulosic ethanol production.

Effect of the biomass-bioethanol production on GHG emissions

<u>SOC</u>

The removal of residue reduces carbon input into the soil and thereby reduces SOC. Drever et al. (2021) modelled a residue harvesting system of drop and bale after a rotary combine. The latter is the predominant type of combine on the Prairies but breaks up the residue into small pieces that reduces the efficiency of residue collection. That system is not optimal for residue removal efficiency but does not interfere with efficiency of grain harvest. They estimated rates of residue removal, based on measured bale yields after rotary combines, of 0.5, 0.4, 0.6, and 0.55 Mg of oat, barley, wheat, and corn residue per Mg of grain, respectively, at moisture contents of 15% for small grains and 25% for corn stover. At those rates and based on above-ground residue-grain relationship (Janzen *et al.*, 2003), above-ground residue removal was 37-38% for small grains but 57% for corn. The residue removal method was specifically chosen to be sub-optimal from the perspective of potential residue removal rate. The suboptimal removal rate, though, may avoid negative long-term yield effects of more thorough residue removal on the Prairies (Malhi *et al.*, 2011b); a sustainable residue removal rate, if any, is not been established for the Prairies. Therefore, considering there is anchored standing stubble after residue removal, there would be sufficient residue remaining to control soil erosion providing there was no fall tillage.

Drever et al. (2021) estimated that residue removal causes a reduction of 200 kg SOC/ha for each t C/ha of residue harvested, based on a meta-analysis of SOC loss from residue removal in Canada and the northern United States (Smith *et al.*, 2013). Requiring land for which crop residue was removed to have fall cover crops and be in NT compensates for much of that SOC loss (the additional NT was only on the portion of Prairie cropland that was not already in NT).

Other emissions

Some nutrients are removed with the residue. Drever et al. (2021) assumed these removals could be compensated for with additional use of fertilizer. Therefore, fertilizer N additions made up for N removals with residue so removal has limited effect on N_2O emission.

Considering all emissions (baling and transporting residue, embodied emission in fertilizer to replace nutrients in removed residue, and biorefinery emissions for ethanol production), Drever et al. (2021) estimated the emission intensity of bioethanol to be 79 g CO_2e/MJ. Fully 80% of this intensity is from SOC reduction from residue removal. This intensity compares to 92 g CO_2e/MJ used for Canadian gasoline derived from fossil fuel by the Clean Fuel Regulation.

Kim *et al.* (2020) provides an average LCA global warming value for CCS extracting CO_2 from ethanol production plants of -112 g CO_2e/MJ for the US Midwest and Central Great Plains. Using this value for Canadian Prairies, would thus make ethanol a carbon-negative fuel.

Impact of bioenergy on Prairie GHG emissions

Current Adoption

The Prairies produced 515 M L of ethanol from grain (mostly wheat and corn) in 2016 (Mupondwa *et al.*, 2017). Although there are many large biorefineries producing ethanol from crop residues in the US, there are none in Canada. The Prairies has one plant in Alberta (Enerkem, Edmonton) that produces ethanol from cellulosic feedstock in the form of municipal solid waste. However, rather than processing the residue and then using biological fermenting to produce ethanol, this plant gasifies the waste and then uses the resulting gases to manufacture two biofuels: methanol and ethanol.

The Prairies are home to the world's first CCS for ethanol production. Since 2012, Husky Oil has been capturing CO_2 produced from fermentation in a 130 M L/yr wheat-grain-based ethanol plant in Lloydminster on the Alberta-Saskatchewan border and injecting the captured 250 t CO_2 per day into Saskatchewan oil fields for enhanced oil recovery. The injected CO_2 becomes stored in the geological formations of those oil fields. There are several CCS projects planned for ethanol plants in the northern US that will inject and store the captured CO_2 in North Dakota.

Potential Adoption and Impact on GHG

Biomass-Bioethanol

Given favorable economic return, Drever et al. (2021) estimated potential adoption for 2030 (based on an assumption of a 10-year period of implementation starting in 2021) could be 50% of fields with available small grain and corn residue after the estimated existing needs for livestock fodder and bedding were met. They estimated a network of 70 M L/yr capacity biorefineries across the Prairies with an average transport distance of 40 km from field to biorefinery.

One approach the Drever et al. (2021) explored to mitigate the SOC loss from residue removal is to have each field with residue harvest to use cover crops and adoption of NT (latter to the extent that land was not already in NT production).

Table 34 summarizes the potential ethanol production, residue use, and GHG emission reductions. Of course, the reductions for cover crops and no-till adoption could be achieved without residue removal. It is clear that CCS has a dramatic effect on GHG reduction.

Table 34: Potential ethanol production, residue use, and GHG emission reductions.

Province	Ethanol Production (M L/yr)	Residue use ('000 t/yr)	GHG reduction from replacing gasoline energy ('000 t CO_2e/yr)	GHG reduction from carbon capture&storage ('000 t CO_2e/yr)	GHG reduction from concomitant cover crop and NT ('000 t CO_2e/yr)*	Total GHG reduction ('000 t CO_2e/yr)
Alberta	729	2844	184	1829	1366	3379
Manitoba	402	1586	104	1009	381	1494
Saskatchewan	1031	4045	244	2587	1993	4824
All	2162	7075	532	5425	3760	9697

*Drever et al. (2021) reported a smaller value that was only the portion of the GHG reductions for cover crop and NT adoption in excess of that assumed to already incented by other programing in their study.

CCS added to existing grain-bioethanol

Grain based bioethanol is a substantial industry on the Prairies. The long-term environmental acceptability of using grain as a feedstock depends on the indirect land use change, land-use change from converting forests and/or grassland to produce grain that was induced by the reduction in grain availability for use for bioenergy (Kocoloski *et al.*, 2009; Gerssen-Gondelach *et al.*, 2017). Consequently, grain production for bioethanol would not be normally considered a BMP but CCS added to existing plants could be a land-based BMP. The C that is captured and stored is from land-based practice of harvest of the grain feedstock.

Assuming that CCS is added to all grain-ethanol plants in Prairies, other than the Husky plant that already has CCS, the reducing in GHG emissions would be 0.95 Mt CO_2e.

Barriers to adoption

The uncertainty about the long-term economics of bioethanol production from crop residues will be the major barrier. Favorable economics are needed to incent the large capital investment required to set up biorefineries. In turn, crop growers need to see a value proposition to sell their crop residue. Drever et al. (2021) estimated a return to growers as CAD 35 per tonne of residue net of baling costs plus value of the fertilizer N required to replace removed N in the crop residue. Without an assured crop-residue supply from growers, the biorefineries would not be profitable.

The long-term future of bioethanol could be a barrier. With current federal government policy to have electric vehicles replace light gasoline vehicles during the 2030s, the long-term need for bioethanol is uncertain. However, biomass can be used for other bioenergy forms such as renewable natural gas or various types of liquid fuels through thermochemical processing (Field *et al.*, 2020; Wang *et al.*, 2020; Cheng *et al.*, 2021; Taboada *et al.*, 2021) as well as directly burned for heating and electricity generation (Sanscartier *et al.*, 2014; Smyth *et al.*, 2017). Therefore, bioethanol production is only one BMP of using crop residue biomass for energy to reduce GHG emissions from fossil fuels.

Co-benefits

The major co-benefit is the additional returns to growers for crop residues (CAD 35 per tonne of residue net of baling costs plus value of the fertilizer N required to replace removed N in the crop residue).

There are also new business and employment opportunities in rural Prairies for baling crop residues, trucking residues, and servicing and operating biorefineries and other infrastructure for this potential BMP.

The bioethanol produced would replace bioethanol that Canada currently imports and thereby provide better domestic energy security.

Trade-offs

The major tradeoff is the soil health implications of residue harvest. On the prairies, the effect on SOC has been negative (Lemke *et al.*, 2010; Smith *et al.*, 2013) and has lowered the more active fractions of soil organic matter (Malhi *et al.*, 2011b, a, c). Straw retention has been shown to have little effect of crop yield over short-term 4 years (Malhi and Lemke, 2007) and over some decadal-scale studies (Lemke *et al.*, 2010) but reduced grain yields by up to 10% over the decadal periods in another study (Malhi *et al.*, 2011d). Determining the sustainable rate of residue removal, if there is one, is important to recommend using crop residue as a biomass feedstock for bioenergy as a BMP for GHG reductions.

Another tradeoff is increased heavy truck traffic associated with transporting crop residue from field to temporary storage and then to biorefineries. The potential soil compaction on the field and the increase in maintenance costs for roads and traffic safety considerations are impacts of wide-scale residue harvest for bioenergy.

Knowledge Gaps

Research is needed into soil health and crop yields impacts of crop residue removal for the Prairies so farmers can make informed decisions about how much, if any, residue removal in which they will participate.

Other biomass sources including biomass herbaceous crops and short-rotation trees provide additional opportunities for biomass supply from farmland in the Prairies (Liu *et al.*, 2014). These represent additional opportunities to augment crop residue and provide other new income opportunities for farmers where their farmland has areas better suited for such biomass production than for conventional agricultural uses. Any conversion of land producing agricultural commodities to produce biomass crops would be subject to indirect land use change emissions being included in their C footprint and so may not be attractive from a GHG perspective.

More research on the potential economics of biomass-based bioenergy for the Prairies is needed to inform provincial and federal government policy in Canada.

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17. Organic and Regenerative Ag Systems

Description

The Manitoba Organic Alliance describes organic agriculture as an "ecological method of agricultural production that respects the natural environment and avoids artificial additives". Its principles focus on improvements to soil health, water quality, energy efficiency, and a diversity of plants and animals integrated into a single system (Frick et al., 2001). It is a regulated industry in Canada and all food with an "organic" label must meet the Canadian Organic Standard certification requirements. The biggest difference between organic and conventional agriculture is that the use of synthetic fertilizers, pesticides and genetically modified organisms (GMOs) are not allowed in organic production. Organic farmers are therefore more reliant on mechanical and biological tools for pest control and nutrient management to maintain productivity. Livestock management in organic systems also has rules that differ from conventional systems. These rules are increased access to outdoors, greater indoor space requirements, and restrictions on the use of growth hormones and antibiotics. As of 2016 there were nearly 1500 organic farms across Manitoba, Saskatchewan, and Alberta which is 1.1%, 2.5%, and 1.0% of total farms in the respective provinces (Canada Organic Trade Association, 2022)

Regenerative agriculture is a term used to loosely describe a wide set of practices that improve soil health, biodiversity, and system resiliency. Common practices associated with the term include cover cropping, intercropping, silvopasture, integrating livestock with crops, improved grazing strategies, and reducing soil disturbance. Many of the principles in the regenerative agriculture movement overlap with those in organic agriculture; however regenerative methods are not regulated and may be thought of more as a socio-cultural trend than a certifiable production system.

As agricultural "systems", regenerative and organic methods encompass many different beneficial management practices. Taken as a whole, the set of practices and its outcomes can be compared to conventional (non-regenerative or organic) production systems.

Effect of Regenerative and Organic Systems on GHG emissions

Soil Carbon

Increasing soil organic matter is a pillar of regenerative agriculture systems. Much of the excitement behind the regen ag trend is driven by the soil's CO_2 mitigation potential from improved land management. The soil building practices most used in regenerative and organic systems are cover crops, perennial crops and forages, improved grazing practices, manure and compost applications, and reduced soil disturbance. Studies that compare organic or regenerative systems to a conventional cropping system on soil carbon sequestration potential are rare and need to be further examined; however, some research has occurred:

Pimentel et al compared an animal-based organic system, a legume-based organic system, and a conventional cropping system. Results showed the animal-based organic rotation accumulating 981 kg of annual soil carbon per ha, ahead of the organic legume-based rotation (574 kg C per ha) and conventional systems (293 kg C per ha) (Pimental et al., 2005).

Lychuk et al, modeled soil organic carbon levels and microbial respiration in Saskatchewan under future climate scenarios with distinct cropping systems. These were either low, reduced, or high-input cropping systems in combination with low-diversity, diversified annual grains, or diversified annuals and perennials. Regenerative agricultural systems align most closely with organic or reduced crop input systems and diversified annual grains or diversified annual grains + perennials. Results showed that at an additional 2 degrees Celsius of climate change, reduced input and diversified annual grain systems were optimal for reducing soil carbon respiration (Lychuk et al., 2019).

Gattinger et al performed a meta-analysis of 74 studies comparing the soil carbon sequestration rates of conventional and organic farming systems. When they considered studies with the best quality data which measured carbon and nitrogen inputs, as well as bulk density, they found increased soil organic carbon stocks in the organically farmed systems. SOC in organic systems were estimated to have 1.98 +/- 1.5 Mg C ha^{-1} more than the conventional systems (Gattinger et al., 2012). The authors note that some of the practices inherent in organic systems (like mixed farming of livestock and crop production and forage legumes in rotation) are a likely driver of the soil carbon increases, and that conventional agriculture can improve SOC levels by adopting these practices too. What they are describing is, in essence, regenerative agriculture.

More regionally specific studies are needed to better understand the effects of these systems on soil carbon in the Canadian Prairies.

N_2O Emissions

Careful management of nutrients is another important aspect of organic and regenerative systems. Reducing or eliminating inorganic nitrogen fertilizers is an effective way to achieve large nitrous oxide emission reductions from both the manufacturing process and N_2O losses in the soil. One way regenerative and organic farming replace inorganic fertilizers is with biological N-fixation, where nitrogen is taken from the air by rhizobium bacteria and shared with legume plants. Annual leguminous crops like peas, lentils, chickpeas, and dry beans can provide a nitrogen credit to carry over to the next crop season, though it is not normally sufficient in meeting the full nitrogen requirements of the following year’s crop. If, instead of harvesting the legume crop, it is terminated prior to producing viable seeds and incorporated into the soil it can provide a much larger nitrogen credit. Table 35 shows the potential N-fixation for various green manure legumes.

Table 35: Estimated N-fixation and Biomass production from Green Manure Legumes on the Canadian Prairies.

Species	Location	Green Manure System	Dry Biomass Produced (kg/ha)	Estimated N-fixation (kg/ha)
Chickling Vetch	Swift Current, SK	Spring Seeded	1800-2800	45-70
Chickling Vetch	Southern MB	Spring seeded	3700-5000	85-111
Field Pea	Edmonton, AB	Spring-seeded	8400 - 9200	210-230
Faba bean	Saskatoon, SK	Spring-seeded	3600	90

Hairy Vetch	Melfort, SK	Spring-seeded	4800	120
Indianhead Lentil	Swift Current, SK	Spring-seeded	1500-4000	40-100
Sweet Clover	Southern MB	Underseeded to preceeding crop	2300-5800	60-145
Alfalfa	Southern MB	Spring seeded, terminated same fall	4900-6300	125-160

Source: *Organic Field Crop Handbook – 3rd Edition. Canadian Organic Growers Inc, 2017* (Frick et al., 2001)

Research at the Natural Systems Agriculture Lab at the University of Manitoba has performed field tests to assess N_2O emissions from conventional and organic wheat systems. The organic wheat system emitted substantially less N_2O emissions over that crop year (163 g N_2O-N / ha) than the conventional wheat did (602 g N_2O-N / ha). Conventional wheat was provided urea as its nitrogen source while the nitrogen for the organic wheat was supplied by biological nitrogen fixation (alfalfa) in previous years. N_2O emissions from the alfalfa plow down in the prior year were closely linked to soil moisture content (Westphal et al., 2018). More research is needed to understand the N_2O emissions associated with the many different scenarios and climates regenerative agriculture functions in.

Other emission impacts

J.W. Hoeppner et al. (2005) studied the energy use and efficiency of two production systems in Manitoba: One conventional, the other organic. When averaged across the various rotations in the study, organic systems had an energy efficiency 40% higher than its conventional counterpart, despite using more mechanical tillage. This efficiency was mostly due to the avoided energy costs associated with nitrogen fertilizer manufacturing (Hoeppner et al., 2006). When directly compared, the wheat-pea-wheat-flax crop rotation had an energy consumption of 24,233 MJ ha^{-1} for in the organic system and 68,498 MJ ha^{-1} in the conventional system (Hoeppner et al., 2006).

Potential impact of Organic and Regenerative Ag on Prairie GHG emissions

Current adoption

The Prairies make up a combined 29% of Canadian organic agriculture producers, 45% of organic hectarage and 80% of organic field crops. There was 763,000 ha under organic production in 2020, an increase of 32% since 2009. Saskatchewan has by far the most organic at 475,000 ha. Alberta is following at 246,000 ha and Manitoba has 42,000 ha in organic production as of 2020 (Canada Organic Trade Association, 2022).

Cereals are the most produced organic crop category and were planted to over 304,000 ha in the Prairies in 2020. Pulses (68,000 ha) and oilseeds (41,000 ha) were the next most common annual crop type. 324,000 ha of farmland in the Prairies were in pasture, forage, green manure, or in some other form of natural area. Fruit and vegetables made up a smaller portion of organic hectares (3000) in 2020, with wild areas covering roughly the same (Canada Organic Trade Association, 2022).

The Prairies make up a smaller portion of Canadian livestock operations at 14% in total (2020). There were 108 organic livestock operations in the Prairies in 2020, with 57 in Alberta, 24 in Saskatchewan, and 25 in Manitoba (Canada Organic Trade Association, 2022).

Regenerative agriculture is not universally defined or standardized which makes it challenging to estimate adoption rates. Specific practices covered under the *regenerative* term can be estimated, however. Full season cover crops on the Prairies are estimated at 11,000 ha and shoulder season cover crops at 104,000 ha (Drever et al, 2021). In 2016, over 19 Mha were using zero-till techniques, which is over 50% of total cropped land (Statistics Canada, 2016). Still, there is a lack of data on other practices like improved grazing and composting applications. What does seem clear is that many farmers, companies, and industry groups have taken a strong interest in regenerative agriculture and it is being promoted widely across the Prairies. New and developing carbon markets and other sustainability initiatives are an important driver for regen ag adoption.

Potential adoption and GHG emissions benefits

Organic crop production has seen an increase in hectarage on the Prairies over the past decade, however the number of organic producers seems to have flatlined in recent years. There are several macro-economic and cultural/social factors that affect adoption rates, for instance, grain prices, seed and fertilizer costs, land prices, and performance of farmer peers in the region. Adoption rates could be driven upward from policy makers. This trend is happening in the European Union, where a target in place to have 25% of land be in organic production by 2030. While a similar policy proposal seems unlikely in Canada, global market demands could encourage Prairie farmers to transition to help meet organic production demand. Canada is a net-importer of organic foods which suggests a potential gap to be filled in production and processing. A likely scenario is that the number of hectares will increase gradually, as it has over the past decade (Canada Organic Trade Association, 2022).

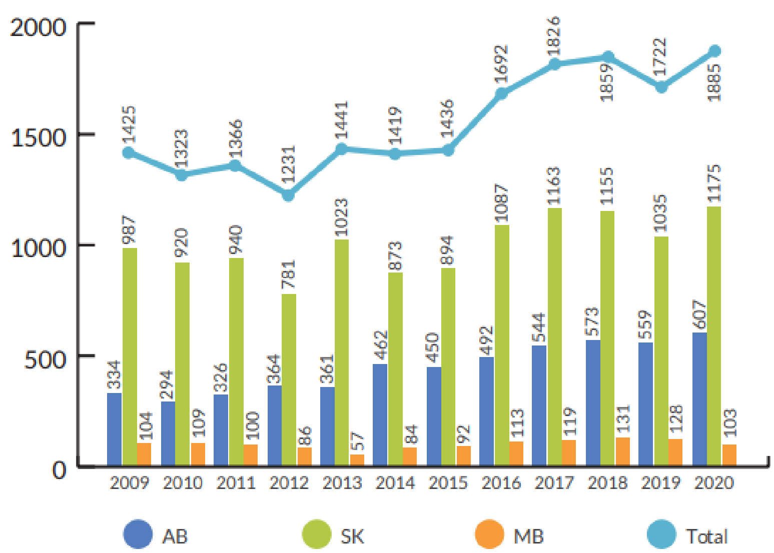

Figure 2: Organic Acreage in the Prairies from 2009 to 2020 (x 1000). Source: 2020 Data – Organic Agriculture on the Prairies Report by the Prairie Organic Development Fund

The outlook for regenerative agriculture may look different as new markets evolve and incentive sustainable farm practices. Companies like General Mills, Nestlé, and McCain Foods are making large investments into farms within their supply chains. As other companies seek to reduce their scope 3 emissions on the farms they source from, more investments and financial incentives for farmers can help to drive adoption.

Pelletier et al modelled the scenario of switching all conventional production of the four major food crops of canola, corn, soy and wheat production to organic using an LCA approach. The analysis showed a potential 33% reduction in greenhouse gas emissions mostly because of changes in fertilizer use. Energy consumption in the organic scenario was averaged out to be 39% of the conventional energy use (Pelletier et al., 2008). If, instead of 100% conversion for those crops, we assumed a 20% conversion from conventional to organic by 2030, that could result in emissions savings of 6.6% based on the Pelletier et al. analysis.

Barriers to adoption

There are multiple barriers to adoption of organic agriculture. The transition period from conventional to organic takes several years, requiring farmers to produce crops and livestock using organic methods without benefitting from the organic price premium for their commodities. Farmers transitioning to organic production is more common when conventional grain and livestock commodity prices are low and transitioning back to conventional production is more common when conventional commodity prices are high.

Many farmers prefer 'clean' fields that have as little weeds as possible, and there is a social stigma around messy fields. Many farmers may also feel obligated to always maximize production as part of the "feeding the world" narrative as well.

Managing pests is also challenging for organic producers because they cannot use conventional herbicides, insecticides, or fungicides. This requirement can necessitate new equipment and tools such as interrow cultivators, blade rollers, or robotic weeders, etc. Nutrient management must also be handled differently if synthetic fertilizers are restricted. Supplementing crops with manure or other organic sources of nutrients can be an expensive alternative and there may not be sufficient supply. To maintain nitrogen supplies, organic and regenerative producers will also often rely on leguminous green manures to fix nitrogen biologically. Maximal nitrogen inputs to the soil are achieved by plowing or incorporating those legumes into the soil prior to them producing seed, which takes the land out of cash-crop production for that growing season.

A general barrier to both organic and other regenerative agriculture systems is a lack of technical or agronomic knowledge of the methods. Concepts like cover cropping, intercropping, green manures, and increased crop diversity are not perfectly understood by farmers or researchers. Furthermore, the measurement and monitoring tools that are needed to effectively produce food with regenerative principles are not always present. Improved access to soil testing, variable rate technologies, field maps, and other advanced tools will help to increase adoption of more sustainable production systems.

Co-Benefits

Increased biodiversity is one of the biggest benefits to organic and regenerative systems. This benefit consists of more plant and animal diversity, as well as more diverse micro-organisms (Frick et al., 2001) and a higher abundancy of birds (Kirk & Lindsay, 2017). Integrating livestock into a cropping system produces synergies on the farm, like more efficient nutrient recycling from grazing cover crops. Reducing inorganic fertilizers mitigates the risk of nutrient losses to the atmosphere and waterways, and reducing pesticides on agricultural lands diminishes its harm to humans and the environment (Frick et al., 2001).

Organic and regenerative systems are evolving to utilize novel solutions to modern challenges. The restrictions imposed on organic producers force them to innovate in ways that conventional producers aren't required to, such as with weed control and nutrient management. Many progressive farm practices now being promoted across the agriculture industry were pioneered in the organic industry. Green manures, robotic weeders, and livestock integration are three examples of organic farm practices being adopted in conventional farm designs.

Trade-offs

Organic crop and livestock production systems have historically been less productive than conventional farm designs. For grain production, this lower productivity is primarily caused by increased competition with weeds and a limited nutrient supply. In North America, organic crop yields have been approximately 84% of conventional yields (de Ponti et al., 2012). This leads to a potential risk of leakage, where emissions are increased elsewhere in order to make up the difference in grain production. Reduced overall biomass in cropping systems can also result in less plant residues being cycled back into the soil, impacting soil carbon stocks.

Another trade-off in organic production is the increased dependency on tillage. Very few herbicide options exist for organic producers which often leaves them dependant on mechanical tools to manage weeds and terminate green manure crops. Mechanical tillage is energy intensive, so organic farms can require more fuel in their operations than conventional farms (Hoeppner et al., 2006). Increased tillage can inhibit soil carbon accumulation.

Knowledge Gaps

On regenerative ag, there lacks a standard definition which makes it hard to both describe and evaluate at the "system" level. Organic production is regulated and does have specific set of requirements, however there is still considerable differences between individual organic operations, as is true for any farm category. This creates challenges in estimating the GHG impacts of a system because an organic farm in one region may be using different kinds of practices (green manure, crop rotations, manure applications) than a neighboring organic producer uses.

There also remains uncertainty around the GHG emissions in organic agriculture and how those compare to conventional, in particular when it comes to soil carbon accumulation and nitrogen cycling in soils.

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18. Integrated Crop-Livestock Systems

Description

Integrated crop-livestock systems encompass several beneficial management practices that incorporate livestock and crop production. Generally, this BMP involves systematic changes to include livestock within cropping, grassland, or rangeland systems, where synergies are achieved through the integration of multiple production types. Depending on the region and livestock species involved, synergies that can be achieved through an integrated crop-livestock system include feed and nutrient self sufficiency, improved recycling of nutrients, and reduced financial and energy costs from forage harvest and manure transport. Crop-livestock systems also provide improvements in soil organic carbon storage and reductions in GHG emissions from pasture or land management improvements, fuel reductions from reduced forage harvest and manure transport.

Integrated crop and livestock systems range based on the crop and livestock type included and whether integration occurs at the farm or a regional level (Peterson et al., 2020). Common types studied relevant to the Canadian Prairies include 1) forage rotations, where livestock graze a multi-year rotation of annual and forage crops, 2) cover crop grazing, where livestock graze the off-season cover or forage crop after an annual cash crop is harvested, 3) stubble grazing, where animals graze the standing residue left over after harvest and 4) dual-purpose crop grazing, where livestock graze the vegetative stage of a crop that is left to mature and is harvested for grain (Peterson et al., 2020).

An integrated system can cause on-farm circularity of nutrients and feed products. This circularity allows farm or regional self sufficiency and decreases dependency on inorganic fertilizers. Under life cycle comparisons of integrated crop-livestock systems, the greatest GHG emissions reductions occurred when upstream emissions, such as feed production, fertilizer and pesticide manufacturing and fuel production, were reduced (Zheng et al., 2022). Under the several integrated crop-livestock systems studied, each was less GHG intense than the reference level of traditional hog systems due to the reduced dependency on upstream inputs (Zheng et al., 2022). Reductions of upstream inputs requires recycling of nutrients from animal systems, which an integrated crop-livestock system can provide.

Integrated crop-livestock systems relevant to the Canadian Prairies may also be conducted through improving land management and introduction of livestock in rangeland and grassland systems. Ecologically sustainable use of grazing can enhance grassland ecosystems through nutrient cycling and soil organic matter maintenance (Asgedom and Kebreab, 2011). Such grazing practices include multi-paddock or rotational grazing, complementary grazing (moving cattle to forage that matures at different times in the season) and reducing stock density (Lynch et al., 2005).

Scenarios testing different integrated crop and livestock systems has given variable results, based on geographic region, crop rotation, livestock type and organization. There are many considerations when adopting an integrated crop-livestock system, as it may significantly impact farm income, livestock or crop production, GHG emissions, energy and fuel consumption, and nitrogen use efficiency (Sneessens et al., 2016).

Effect of Integrated Crop-Livestock Systems on GHG emissions

Soil Carbon

In contrast to harvested forages and grains, the risk of losing soil organic carbon (SOC) in permanently seeded pastures is negligible (Boehm et al., 2004, ECCC 2019). Sequestering additional carbon due to year-round grass cover is thought to improve the GHG budget of pasture-based systems. However, incremental SOC accumulation takes time, while livestock and manure emissions are constantly being emitted. Carbon dioxide (CO_2) fluxes in grazed systems were less than non-grazed systems based on reduced C input from grazing, which was also consistent with lower SOC when measured (Abagandura et al., 2019). Other research suggests pastured systems increase SOC (Thelen et al., 2010). However, one study suggests that more than 4 years of research is needed to understand the interactions between cropping and forage or grain production systems and their impact on yield, SOC and other factors (Tanaka et al., 2005). A Northern Great Plains study found sequestration rates between 0.39 and 0.46 Mg C per ha per year, based on 44 years of soil sampling, in permanently established pastures that were 70-87 years old (Liebig et al., 2010).

GHG Emissions

Emission intensities associated with grazing cattle are higher than from the feedlot system, due to feedlot animals having a shorter lifespan, lower stress from weather and exercise, and higher quality diets (low roughage). Feedlot systems also have high densities of cattle and risk of over-stocking, which can damage pastureland (Lyons et al., 2018, Seijan et al., 2015, Van Haarlem et al., 2008). Grazing livestock also tend to have lower feed conversion rates, requiring more metabolic activity to meet slaughter weight requirements, which causes more enteric methane production for ruminants (Dyer and Desjardins, 2021). However, CH_4 production for grazing cattle is challenging to quantify directly and is often not considered in studies considering land management practice changes.

Nitrous oxide (N_2O) emissions show similar patterns and variation between grazed and non-grazed pastures of the same crop rotation (Abagandura et al., 2019). N_2O emissions have been shown to not significantly increase when grazing livestock is added to a diversified crop rotation as grazing has been shown to enhance microbial activity and N mineralization, reducing the potential for N_2O emission (Denef et al, 2011, Sainju et al, 2012). Others have shown that N_2O emissions decreased on grazed plots (Abagandura et al., 2019).

In a study where all gases and SOC were measured, the global warming potential (total impact of all gases) of grazing beef on three systems in the Northern Great Plains was most impacted by the N_2O flux and SOC storage (Liebig et al., 2010). Enteric fermentation from different stocking densities also played a role in the total emissions but to a lesser extent than N_2O and SOC. Total GHG emissions from a heavily grazed and fertilized crested wheatgrass made the pasture a net source of emissions. In contrast, unfertilized, naturally vegetated pastures were net sinks on a per hectare and per weight gain basis at both moderate and heavy stocking densities (Liebig et al., 2010). SOC storage had a strong role in balancing out the N_2O and enteric fermentation-CH_4 emissions observed in the study, which was based on a change in SOC over 44 years of soil samples (Liebig et al., 2010). Mitigation potential values for the three treatments in this study are described in Table 36.

Potential impact of Integrated Crop-Livestock Systems on Prairie GHG emissions

Potential adoption and GHG emissions benefits

Due to the highly diverse nature of Integrated crop-livestock systems, it is challenging to estimate the current adoption or the potential adoption of these systems within the Canadian Prairies. This is further challenged by the lack of data on specific farming practices for mixed operations within Canada.

The mitigation potential of integrated crop-livestock systems varies drastically based on the management practices included and the methodology in which GHG emissions and SOC are quantified. Of the limited research conducted on integrated crop-livestock management, studies representative of the Canadian Prairies with sufficient mitigation potential data are included in Table 36. As noted below, improving tame pastures and native rangeland varies significantly on the ecological region.

*Table 36: Mitigation potential of several integrated crop-livestock systems relevant to the Canadian Prairies. * Note: Improved grazing management includes complementary grazing, reduced stocking density, continuous or rotational grazing of grass and legumes, and carbon sequestration is based on the ecological region of study. Negative values indicate a net source in GHG emissions, as positive values represent a mitigation potential in GHG emissions or SOC storage.*

Description	Mitigation Potential ($tCO_2e\ ha^{-1}\ yr^{-1}$)	Relevant GHGs included in Analysis	Reference
Integrating livestock in a corn-soybean rotation	9.34	SOC, N_2O, CH_4	Thelen et al., 2010
Native rangelands under improve grazing management*	0.007-0.096 (Mixed Grassland – Aspen Parkland ecoregion)	SOC	Lynch et al., 2005
Tame pastures under improved grazing management*	0.228-0.342 (Boreal Transition – Moist Mixed Grassland ecoregion)	SOC	Lynch et al., 2005
Heavily grazed Crested wheatgrass in Northern Great Plains	-0.397	SOC, N_2O, CH_4, (including enteric fermentation)	Liebig et al., 2010
Heavily grazed natural vegetation in Northern Great Plains	0.618	SOC, N_2O, CH_4, (including enteric fermentation)	Liebig et al., 2010
Moderately grazed natural vegetation in Northern Great Plains	0.783	SOC, N_2O, CH_4, (including enteric fermentation	Liebig et al., 2010

Barriers to adoption

The primary barriers to transitioning to an integrated crop and livestock system were farming norms, complexity of management, biophysical conditions, and financial costs for infrastructure (Hayden et al., 2018). Depending on the region, the norms for specialized farming practices provide significant barriers to adopting an integrated system in an environment driven by the dominant markets and farming systems, financing and insurance available and the regulatory environment (Hayden et al., 2018).

Farmers remarked that integrated crop-livestock systems have considerable complexity in their management as livestock farming is a long-term commitment that requires intensive management, knowledge of stocking densities and upfront infrastructure costs (Hayden et al., 2018). In addition, where managing existing soil issues and improving soil health was a priority, it was perceived to be a competing priority for those interested in adopting integrated crop-livestock systems (Hayden et al., 2018).

Financial costs are a significant barrier to adopting an integrated system due to the need for new farm infrastructure. Even when physical infrastructure issues were addressed, the long time horizon for returns on investment was a considerable challenge (Hayden et al., 2018). Farmers were discouraged when the benefits associated with an integrated crop-livestock system (animal welfare, soil health, financial returns) did not appear quickly, and the lag in financial returns in some cases caused farmers to revert to their specialized farming system (Hayden et al., 2018). Challenges beyond the farmers control included regional infrastructure for the new system, financing and insurance and long time horizons for returns (Hayden et al., 2018).

Hayden et al., (2018) also discussed the many opportunities and co-benefits that may outweigh the existing barriers for adoption. However, these opportunities exist differently for each farmer or production system.

Co-Benefits

Integrating livestock into cropping systems provides many environmental and economic benefits to farmers within the Canadian Prairies. Organic amendments such as animal manure is an economic alternative to inorganic fertilizers, and integrated systems allow for reduced costs on fertilizers for crop management. Integrated systems also often have energy and fuel savings from reduced transport of feed and manure, as commodities are harvested (forage) and applied (manure) by animals within the farm themselves, instead of transported long distances (Asgedom and Kebreab, 2011).

Integrated crop-livestock systems often improve the nitrogen use efficiency in crop production (Cicek et al., 2014). This increased efficiency is due to the use of recycled nutrients in manure in contrast to the GHG-intense inorganic fertilizer application. Improving nitrogen use efficiency has economic and environmental implications such as reducing nitrate leaching, nutrient loading in runoff, and potentially reducing indirect N_2O emissions (Asgedom and Kebreab, 2011). Improved grazing practices also contribute to improved soil, water, and air quality (Asgedom and Kebreab, 2011).

Generally, livestock integration into cropping systems has positive effects on crop yields and soil organic matter, despite the potential for soil compaction during winter grazing (Tracy and Zhang, 2008). Trade-offs for soil compaction can be mitigated by winter grazing only on frozen soils, thereby avoiding soil compaction from livestock (Clark et al., 2004). A recent global meta-analysis compared the crop yield effects of integrated crop and livestock systems to their specialized system counterparts and found that, in most cases, annual cash crop yields were similar between integrated and unintegrated systems (Peterson et al, 2020). The exception was in dual-purpose annual cropping systems, where yield was on average 20% lower than single-purpose crops (Peterson et al., 2020). In soil types and crop types relevant to the Canadian Prairies, this meta-analysis indicated that yield was the same or improved with livestock integration under cover crop grazing, forage rotation and stubble grazing (Peterson et al., 2020).

In addition, integration of livestock can also make agronomic practices like cover cropping more financially viable for farmers. Thiessen-Martens and Entz (2011) found that instead of terminating cover crops, grazing sufficient cover crop biomass can provide revenue per hectare based on the animal live weight gain rate. The gross revenue generated from grazing high biomass cover crops (5000 kg per ha) was between $385-770 per hectare based on 2011 prices (Thiessen-Martens and Entz, 2011). This approach allows for the numerous cover cropping benefits to soils and nutrient cycling while improving the cost-effectiveness of the practice.

Trade-offs

Given that this BMP is a systematic approach, the potential trade-offs depend on how the system is designed and operated. Based on simulations of crop-livestock systems, Sneessens et al., (2016), found that how the system was organized was directly responsible for the trade-offs observed. For example, higher proportions of crop production increased farm income but negatively impacted nitrogen balance, while increasing livestock production increased energy consumption.

Depending on type and nutrient composition of manure, concentrations of phosphorus and other minerals need to be strictly monitored during implementation of any crop-livestock system, as these minerals can contribute to environmental pollution. In some regions of the Canadian Prairies, animal production occurs in high densities (ex. Alberta Feedlot Alley), which can contribute significantly to water, soil and air pollution. However, this pollution can be mitigated through livestock management whereby nutrients are deposited over a large area. Influx of nutrients and sediment in water bodies can also be mitigated by reducing winter grazing and strategic practices on slopes (Monaghan et al., 2017).

Integrated crop-livestock systems may also increase the demand for land. For example, grass-fed beef and other pasture-based commodities increase the land area required per tonne of meat produced due to harvested forage production (Dyer and Desjardins, 2021). This is a considerable trade-off as other potential land uses such as conserved forests and grasslands have a higher GHG mitigation potential than integrated crop-livestock systems.

Knowledge Gaps

Many challenges exist for quantifying the GHG emissions associated with integrated crop-livestock systems. Studies report soil impacts and animal impacts separately (i.e., soil CO_2, N_2O, SOC and animal CH_4), and may only report emissions during the growing season (Abagandura et al., 2019). Mitigation of GHG emissions is regionally specific and requires data from the Canadian Prairies on livestock integration practices and their associated GHG intensity.

For systematic changes, LCA studies are best for comparison purposes, however, there is significant variation in the potential operation of integrated crop-livestock systems. Some studies discussed limitations due to their boundary of the farm-gate, and not including upstream or downstream impacts of the systemic change (Dyer and Desjardins, 2021).

Some literature is available discussing the potential trade-offs of land requirements for integrated crop-livestock systems in comparison to natural or intensified crop production systems (Dyer and Desjardins, 2021). However, many scenarios exist for land management and the most efficient or practical systems have yet to be conclusively determined.

Finally, there is significant gap in the adoption of various types of integrated crop-livestock systems, specifically regarding the practices currently adopted within mixed production style farms. Due to this gap, it is challenging to quantify or estimate the potential for integrated crop-livestock systems within the Canadian Prairies due to the variability in current GHG emissions estimates for different production systems and the lack of data on specific practice adoption for integrated crop-livestock systems within the region.

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BMPs for Future Research

This section includes seven BMPs that may have a significant opportunity to provide GHG mitigation and co-benefits to the agricultural system but lack sufficient detail on one or many of the categories included for each BMP in the sections above. Due to this lack of information within the literature, the BMPs in this section have a short description of the potential BMP and the gaps in our current understanding that are needed to adequately assess the BMP for adoption within the Canadian Prairies. Where more information was available (e.g., co-benefits, trade-offs, barriers to adoption, etc.), it was described. The types of BMPs included may be useful in annual crop production, an extension of the natural landscapes or as part of a farming system. BMPs in this section require further research to evaluate the GHG mitigation potential in the Prairies, including the current and potential adoption.

19. Integrated Pest Management

Description

Integrated pest management (IPM) describes a holistic approach to managing agricultural pests like weeds, unwelcome insects, and disease. The strategy utilizes a combination of practices that focuses on long-term or systematic prevention of pests, rather than prescriptive, short-term solutions. Examples include biological controls, modified practices, plant and animal diversity, and many others. Typically, IPM strategies have an emphasis on reducing environmental harm and risks to human health.

The two main methods of weed control on the prairies are herbicides, tillage, or a combination of both. Tillage kills weeds mechanically while herbicides do chemically, but both impact soil processes and have broader environmental implications. Novel approaches such as weed zappers, comb-cuts, and grazing are less common weed-control methods. Organic crop producers may use biological or agronomic solutions to manage weeds such as increased seeding rates or adding highly competitive bi-annual forages to a crop rotation (Caroline Halde et al., 2014). Advancements in modern technological solutions, such as John Deere's "See and Spray" equipment have the potential to drastically reduce the volume of herbicides applied by specifically targeting weeds.

In organic crop production systems where no herbicides are used, tillage events are common in the spring to prepare the seedbed for planting and in the fall after harvest. Occasionally, a third tillage event will occur either before or after the crop season. Tillage frequency occurred an average of 2.8 times over a 12-year study period in a Manitoba organic cropping system (Hoeppner et al., 2006). Research from the Natural Systems Agriculture Lab in the University of Manitoba has experimented with reduced and no-till cropping systems for organic agriculture. Incorporating highly competitive green manures into a crop rotation, rolling crimping the green manure into a mulch, and then directly seeding cash crops (wheat and flax) into the mulch proved effective for weed control for a period of 1.5 to 2 years (Halde et al., 2014). Systems that produce high levels of net-primary productivity with lower soil disturbance are more likely to sequester more carbon (Bolinder et al., 2007b).

In conventional agriculture, herbicides applications vary by crop type, pest pressure, and farmer decision-making. It is normal for conventional farmers in the prairies to apply herbicides before, during, and following harvest. Fungicides and insecticides are used less consistently and depend on the seasonal circumstances such as moisture and insect patterns. Apart from environmental effects, pesticides have an energy cost at the manufacturing, transportation, and application phases (Clements et al., 1995). A study in Iowa compared the energy costs of a traditionally managed corn-soy rotation to rotations with more crop diversity and lower input requirements. The 3-year (corn-soy-small grain with red clover) rotation and 4-year (corn-soy-small grain with alfalfa-alfalfa) produced similar economic returns and harvested crop weight as the traditional system but used far less inputs. The energy inputs of the pesticides were 1.64 Gj $ha^{-1} yr^{-1}$ for the traditional 2-year rotation, 0.63 Gj $ha^{-1} yr^{-1}$ in the 3-year rotation, and 0.48 Gj $ha^{-1} yr^{-1}$ in the 4-year cropping system (Cruse et al., 2012). These data suggest considerable potential for reduced energy and associated GHG emissions from reducing pesticide inputs. In addition, modern crop production practices such as diversifying or adding perennials to the rotation further reduce the need for herbicide inputs, and therefore the GHG emissions from the rotation.

Knowledge Gaps

Very little information exists on the link between integrated pest management strategies and climate mitigation, as IPM strategies are generally targeted at pest mitigation, improving biodiversity or human health. The trade-offs between higher pest pressure and less tillage and/or less pesticide applications need to be further investigated. Changes in herbicide and tillage use are often done through systematic changes to the farming system, such as substituting crops or integrating livestock. The net changes to soil carbon stocks and sequestration, as well as the secondary effects of management changes can either increase or reduce the mitigation potential of integrated pest management. Net primary productivity is also a critical component of soil carbon sequestration (Bolinder et al., 2007b), so a better understanding on how weed populations contribute to or inhibit soil carbon levels is needed. Generally, determining the influence of IPM from a system-level (i.e., as an addition to an organic system), would provide a deeper understanding of the system's GHG benefits.

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20. Maximize Crop Residue Production

Description

The SOC stocks in soil change due to difference between C inputs and decomposition of new C additions plus decomposition of existing SOC stocks. Therefore, maximizing crop residue production will maximize C inputs and, until an equilibrium is reached between input and decomposition, increase SOC stocks.

Practices that increase crop yields, such as improved genetics, more optimal fertilization, and management of insect pests and plant diseases, will all increase C input to the soil. Since these are all general goals of growers, they are BMPs but are not specific BMP for GHG reduction.

Weeds, although they can reduce the economic yields and their management is an agronomic BMP, do not necessarily reduce the C input. However, there is no evidence that promoting weed growth is beneficial to SOC gains so cannot be recommended as a BMP from the perspective of GHG emissions and certainly cannot be recommended as a BMP from an agronomic perspective.

Perennial pastures and forages provide a more positive C input-C decomposition balance than annual crops and so their production instead of annual crops increases SOC stocks (ECCC, 2022). However, the production of perennials is generally to provide feed for ruminant livestock so the emission from that livestock production needs to be considered when switching from annual crops to their production. Therefore, growing more perennial crops cannot be immediately judged a BMP from the GHG perspective.

Changing which annual crops are grown can affect the SOC balance. The amount of residue produced depends on total plant growth and the apportioning of growth between the harvested and crop residue (including roots) portion. Typically, potato, flax, and grain legumes of lentil, chickpea, and soybean produce relatively less crop residue than cereals or other oilseeds grown under the same conditions. Oat and canola produce more root biomass than other important Prairie annual crops such as wheat, barley, or pea.

In a modelling study, Fan *et al.* (2019) identified that feasible increases in canola and oat production at expense of other crops could increase SOC in Canada by 22 Mt CO_2e/yr compared to long-term trend. Paustian *et al.* (2016) identified using crop phenotypes with enhanced quantity of roots as important opportunity to mitigate GHG emissions. Deep rooted crops have the ability to translocate labile C to subsoil depth where decomposition effects occur at slow rates. At these depths C could become stabilized than at the C-rich surface leading to enhanced soil carbon sequestration than at the C-rich surface. Another pathway is to alter the chemistry of crop residue, so it is more slowly decomposed and/or more efficiently converted to chemically or physically protected forms of SOC. Genetic changes to various crops to increase crop residue production or increase residue C persistence has not been evaluated for the Prairies.

Impact on GHG emissions

The main impact of increasing crop residue on SOC will be increasing C input above historic levels. Whether the crop selection to increase C input is a legume or not and the amount of N additions required to provide increased C inputs determines whether N_2O emission increase or decrease.

To change the crop itself depends on the relative profitability to grow that crop. Fan *et al.* (2019) identified that the increases in SOC in Canada by a feasible increase in the proportion of canola and oat is 12 Mt CO_2e/yr in 2030 compared with the current trend. Most of that increase would occur on the Prairies. The potential for cultivars (genetics) that increase crop residue amount or crop residue C persistence will depend on how that affects the economics of the harvestable portion, i.e., grain. Paustian *et al.* (2016) suggest the potential is large both in terms of area for adoption and net emission reduction per unit area.

Barriers to adoption

The crop switching with current genetics depends on the expected profitability of the switch with any GHG-related incentives.

Changing the genetics to increase crop residue production and/or the efficiency of crop residue C is sequestered in the soil will depend on how those genetics affect the economics of grain production from those cultivars.

Trade-offs

Changing production to crops that increase C inputs may reduce returns compared to growing other crops.

Changing to cultivars that enhance C sequestration may reduce grain yields. In particular, without an increase in net photosynthesis, increasing the partitioning to photosynthate to crop residue would be expected to reduce grain yield.

Research Gaps

Breeding to increase relative residue production or changing the chemistry of crop residue to reduce its decomposition is largely unexplored.

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21. Increasing Diversity of Crop Rotations

Description

Crop rotation diversification refers to the practices of adding different species of crop types within an annual or perennial crop rotation (Messéan et al. 2021). There are several reasons for diversifying crop rotations, including reducing weather and yield risks, managing weed populations, reducing plant diseases, managing workloads, creating the proper environment for subsequent crops, reducing fixed costs per unit of production, accessing alternative markets and improving resilience in the crop rotation. Crop diversification (i.e., increasing number of cash crops grown) is included within many agricultural practices including crop rotations (Renard and Tilman 2019), intercropping and multiple cropping (Hufnagel et al. 2020), agroforestry and landscape heterogeneity (Beillouin et al. 2019). Crop diversification may include both species diversity (i.e., different species of crops in a rotation), or functional diversity (i.e., crops in the rotation have different traits). For example, adding peas to a canola-wheat rotation provides both species diversity, and functional diversity (peas are a legume and have different traits than a cereal or oilseed crop).

Functionality of diversification is an important concept within crop rotations, as ecosystem functioning would depend on the range of traits crop species posses (Tilman, 2001). An ecosystem with increased diversity would be more stable in comparison to individual species. For examples if we consider a trait such as biological N fixation mostly carried out by leguminous crops, under a cereal rotation of wheat-oat-barley rotation that does not contain a N fixer, the rotation will be functionally redundant (Fetzer et al. 2015) compared to a wheat-soybean-canola rotation (Dent and Cocking 2017) even though both rotations have the same number of species. Functional diversities are also created based on timing of the cropping event. For example, winter wheat, a cool-season crop, grown in fall keeps the ground covered during the winter, which provides numerous benefits. A similar species, spring wheat, is grown during spring through the summer. Functional diversity can also be considered under crop resource use (Tilman, 2001). The ability of plants to utilize carbon efficiently varies and thus creates diversity. For instance, wheatgrass (i.e., a C-3 grass) and wire grass (i.e., a C-4 grass) belong to the same class of species but have a different carbon pathway, and C-4 plants are more photosynthetically active than C-3 plants (Pearcy and Ehleringer 1984). Decoupling the effects of functional diversity and related species diversity when assessing crop rotations can provide better understanding on what species are needed to achieve multifunctionality and more beneficial farm outcomes.

In practice, diversification of crop rotations could be done through a temporal or spatial configuration (Hufnagel et al. 2020). In the temporal configuration, crop rotations are designed based on agronomic, biological, and ecological factors (Hanson et al. 2007). For example, annual legumes (e.g., dry peas) could be rotated with perennial legumes such as alfalfa for the added benefit of yield improvement and soil N enrichment thereby reducing N fertilizer use (Hufnagel et al. 2020). The choice of crop sequences in rotations can be influenced by economic factors such as market accessibility, grain process, and input costs (Hanson et al. 2007). In the spatial configuration, crop rotations are deployed to either disrupt weed, pathogen, and insect cycles (Kirkegaard et al. 2008), promote microbial activity and root growth (Hocking 2001), reduce allelopathy (Purvis 1990), reduce nitrate-N leaching or soil erosion (Johnston et al. 2002). For example, cover crop inclusion in polycultures offer more ecological services, e.g., weed suppression, N retention (Finney et al. 2016), than individually grown in monocultures (Finney and Kaye 2017). Crop diversification allows producers the option of flexibility to achieve an agronomically sound rotation. Many long-term studies have been conducted in Canada testing the economic, environmental, and practical impacts of diversifying crop rotations in either wheat-, canola- or corn-based systems (Bowles et al. 2020, He et al. 2021, Khakbazan et al. 2022, Bradshaw et al. 2004, Benaragama et al. 2016). In a study conducted to assess the adoption of crop diversification in the Canadian prairies between 1994 to 2002 (Bradshaw et al. 2004), it was gathered that based on the annual crop seeding data from over 15,000 Canadian prairie farms, individual farmers have generally become more routinely specialized in their cropping patterns since 1994. Recent studies have evaluated the economic impacts of more diverse rotations. Over a period from 1987 to 2015, wheat yields increased in both wheat-canola-pea and legume green manure-wheat-wheat crop rotations compared to continuous wheat cropping by 33.5 and 10.8%, respectively (He et al., 2021). Adding pulses (lentils, chickpeas, peas) in various ways had a similar economic return as lentil-based rotations and significantly higher economic return to wheat monoculture systems (Khakbazan et al., 2022). There are many ways to increase diversity in crop rotations, and producers should strive to achieve levels of diversity that are adequate to attain the goals established for their situation.

Effect of Crop Diversification on GHG emissions

Soil Carbon

Crop diversification can significantly increase and decrease CO_2 emissions by altering the balance between C inputs and outputs in agricultural systems (Lal, 2004). Crop diversification affects the aboveground plant C through the composition and abundance of plant diversity (Li et al. 2018) and the belowground plant biomass C by influencing root architecture and biomass (Liu et al. 2020). In a 25-year study on the Canadian prairie, researchers found that the gain in soil organic carbon over time played a significant role in offsetting carbon emissions ascribed to crop inputs (Gan et al. 2014). In this study, for each kg of wheat grain harvested, the wheat carbon footprint was reduced by a net of 0.027–0.377 kg CO_2e with most soil C sequestration increment resulting from a legume-included rotation.

Potential adoption and GHG emissions benefits

Diversification of crops presents an opportunity for reducing the carbon footprint of cropping systems (Tian et al. 2021). For example, the introduction of alternative crops such as N-fixing legumes in N-reliant cereals and oilseeds rotations decreased carbon footprint mainly due to a reduction in the N fertilizer needs of succeeding leguminous crops (Gan et al., 2014; Sinclair and Vadez 2012). However, in practice farmers would need to reduce the application rate of N, considering the biological N that is already in the soil. Similarly, the carbon footprint of durum wheat in rotation with a pulse crop was 10% lower compared to wheat in rotation with cereal in a study on the Canadian prairies (Gan et al. 2011a). Yet in another study in a similar area, (Gan et al. 2011b) reported a 34% carbon footprint reduction in the durum wheat grown following two consecutive pulse crops compared to a cereal monoculture. The exact emissions reductions for increasing crop rotation diversity is heavily dependent on the type of crop rotation that is adopted, as each crop has unique impacts on SOC and GHG emissions.

Barriers to adoption

Crop diversification strategies are more effective than others in supporting ecosystem services. However, the adoption of diversification strategies by producers would largely depend on realized economic benefit as well as the resilience of such practice to adverse climate conditions. Consequently, a profit-driven producer will only embrace a new cropping system if it is likely to provide a net economic return compared to currently adopted systems with lower production costs. Producers have become more responsive to relative prices since the early 1990s, an indication of greater market reliance (Smith et al. 2001). For example, Smith et al (2001) attributed the perceived major upward trend in the adoption of crop diversification in the prairies to changes in the prices of canola and pulse crops relative to the price of wheat. Barley prices had no impact on cropping diversity; canola prices had the greatest impact.

Additional factors that could influence changes in crop diversity, but could not be tested for in this study, including emerging technologies, the changing structure of prairie agriculture, government indications that the industry needs to become market-oriented because of limited treasury resources, and the development of markets for alternative crops (Smith et al. 2001).

Co-Benefits

Crop Productivity

It has been well-researched that most crop diversification strategies could potentially improve crop yields (Beillouin et al. 2019; Harker et al. 2015; Li et al. 2018; Li et al. 2019). For example, both wheat-canola and pea-barley-canola rotations increased canola yields by 0.20-0.35 Mg ha^{-1} compared to continuous cropping of canola in western Canada (Harker et al. 2015). In a longer study of 8 years, diversified rotations which included lentils, peas, and chickpeas enhanced yields of chickpeas compared to wheat under mono-cropping (Li et al. 2018).

Crop productivity in diverse crop rotations also occurs due to the increased resilience to environmental stressors like drought (Bowles et al., 2020). A meta-analysis of North American crop rotations found that more diverse rotations had positive impacts under unfavourable weather conditions, including reducing yield losses by 14-90% in drought years (Bowles et al., 2020). Providing benefits in stressful conditions is a major co-benefit of diversity in rotation for both environmental stressors as well as weed and disease pressure, which is more relevant as farmers adapt to new growing conditions under climate change.

N use efficiency

Increasing plant diversity within rotations can reduce fertilizer N application for subsequent crops. In a study done across six locations in western Canada, St. Luce et al. (2015) showed that wheat produced more grain with an optimum N rate application when sown after a diversified rotation sequence compared to continuous wheat. In another field experiment at Scott, Saskatchewan, crop diversity following the sequence Canola-fall rye-pea-barley-flax-wheat for six years appeared to store more of the excess N as soil organic matter during dry cycles thereby reducing depletion of soil N supplies while also avoiding large leaching losses (Malhi et al. 2009).

Water Use efficiency

Preceding crops and rotations can significantly affect the water use efficiency in cropping systems. Niu et al., (2017) assessed the rotational benefit of 4-year crop rotation systems on water use and results showed that soil water was greatest at wheat sowing immediately after peas or lentils than with continuous wheat. Similar findings were reported by Archer et al., (2020) where a high-input diversified rotation recorded 166% higher water use efficiency than its low-diversity rotation counterparts.

Income Stability

Some studies have documented the benefit of agricultural diversification on both household income and return on investment (El Benni et al. 2012, Harkness et al. 2021, Lawes and Kingwell 2012, Pacín and Oesterheld 2014). Harkness et al. (2021) established in their study that increasing agricultural diversity by a degree of specialisation (i.e., less diversity in crop) brought about a 20% increase in income variability. Similarly, another study provided evidence on reduced income variability as a result of crop diversification (Mzyece and Ng'ombe, 2020). Although it is not clear whether certain species will exert individual or composite effect on revenue stability (Harkness et al. 2021), drought resistant grassland species and leguminous crops have been shown to improve yield stability which may in turn lead to higher income returns (Dardonville et al. 2020). Increasing agricultural diversification leads to farm resiliency and therefore reduces potential farm risks that would have otherwise translated to financial downturns (Pacín and Oesterheld, 2014). Also, the stabilization of farm income revenue by crop diversification would be contingent on the production cost for each specific crop (Harkness et al. 2021).

Knowledge Gaps

There is a lack of research on the impact of crop diversification on GHG emissions within the prairies with little or no information on key emission contributors from life cycle assessments. For example, a DNDC model assessment provided good performance for estimating N_2O emissions from continuous corn rotations, but poor performance for rotational oats and alfalfa crops (Jiang et a., 2021). Most current life cycle assessments do not consider the complexity of cropping systems (Hufnagel et al. 2020). Studies on increasing plant diversity also need to be broadened to include research areas such as total energy, caloric, or protein yield across an entire crop rotation. These areas are critical to determining food security at the farm and regional levels (Seufert and Ramankutty 2017).

Data synthesis on the effect of crop diversification before and after the land-use change is also lacking. Furthermore, an analysis of long-term yield trends encompassing a range of crop rotations, key management practices such as fertilization, and climate and soil type is needed. This is because an assessment of diversified rotations can help agriculture adapt to increasingly stressful growing conditions while contributing to sufficient food production. Although such integrated knowledge has urgent policy relevance, it has been hindered by a lack of adequate long-term agroecosystem research networks that synthesize cross-site results (Bowles et al. 2020).

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22. Rebuilding degraded agricultural land through targeted regenerative agriculture practices

Description

Targeted regenerative agriculture involves specific practices adopted based on particular degraded soil conditions. As mentioned above, regenerative agriculture is a term used to describe practices that improve soil health, biodiversity, and system resiliency. Many of these practices include improved management practices as compared to conventional agriculture, such as cover cropping, intercropping, silvopasture, integrating livestock with crops, improved grazing strategies, and reducing soil disturbance. Just like in conventional agriculture, regenerative practices can be adopted individually or as part of a system to benefit the soil, environment, biodiversity, and agricultural production.

Paustian et al., (2016) outlines a decision tree for cropland GHG mitigation that outlines some of the potential practices that could be adopted on degraded soils. In soils where nutrients are deficient, these practices include adding nutrients in an organic or inorganic form, using lime, and growing N-fixing species. In soils left fallow, including regions where summerfallow is an adopted practice within the Canadian Prairies, cover crops can be grown, or the fallow fields could be otherwise vegetated. In soils with a history of disturbance or compaction due to intensive tillage, a targeted regenerative agriculture practice is to reduce or terminate the use of tillage while implementing residue retention.

Each targeted approach has barriers to adoption, co-benefits and trade-offs. Many are described in the corresponding BMP summaries outlined above (reducing tillage intensity, cover cropping, N management, increasing grain legumes). Generally, the co-benefits of targeted regenerative agriculture on degraded land include increasing the productivity of land that has been deemed unproductive or has had severe degradation. Although rebuilding the land takes time and labour for a farmer, it improves soil productivity and can improve yield potential for cropping systems. Rebuilding the land through the low-cost practices outlined above also improves water quality, reduce soil erosion, and improve soil health.

Knowledge Gaps

Research gaps for targeted regenerative agriculture include determining the areas within the Canadian Prairies that are currently degraded, the amount of degradation that exists, and if rebuilding the soil through targeted practices is worthwhile.

Based on current data , it is not clear what land is considered degraded and what the levels of degradation exist within the Canadian Prairies. Currently there is research to suggest SOC storage gains and losses as well as areas of high erosion risk exist within regions of Canada, but the specific soil characteristics that need to be managed on degraded lands vary. It is also not clear to what extent regenerative agriculture practices are adopted across the Prairies, nor whether it is to improve existing agricultural land or to target unproductive land.

In addition, the GHG mitigation potential of targeted regenerative agriculture is not clear. This is due to the lack of information on existing soil conditions, lack of research of SOC storage on degraded lands and the variation in degradation is not well characterized. The GHG mitigation potential for the synergistic effects of multiple regenerative practices being implemented together has not been determined. This is a challenge because it is not clear whether the time and effort for farmers to rebuild lands that are low productivity is worthwhile from an economic and feasibility perspective.

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23. Conversion of marginal cropland to permanent cover – Land set aside

Description

Degraded agricultural lands or marginal lands are caused by the loss of productive capacity within the soils through erosion, nutrient and soil organic matter losses, agricultural management, or environmental stressors (such as drought). Marginal lands often have no agricultural or industrial value as the soil has poor soil characteristics and is unproductive for crop and livestock systems. Improving marginal land for production requires targeted practices such as adding nutrients, improving pH, and growing high biomass or N-fixing species. For degraded and marginal lands, the most productive mitigation option is to convert to perennial vegetation either left unmanaged or sustainably harvested for bioenergy purposes (Paustian et al, 2016). Agricultural land conversion to permanent cover for growing trees or bioenergy crops is often referred to as "land set aside".

In cases where a small portion of the field is unproductive, it can be suitable for growing trees, shrubs, or other perennial herbaceous vegetation while a whole field may be suitable for growing high biomass perennials crops such as Switchgrass or Miscanthus that can be harvested for bioenergy purposes (Xu et al., 2022). A modelling study in the US Midwest looked at the GHG impact of different agricultural systems for bioenergy production and found that successional herbaceous vegetation established on non-forested, marginal land had the highest direct GHG mitigation compared to conventional agricultural systems, poplar, and alfalfa (Gelfand et al., 2013). Looking at areas greater than 0.4 ha, the study found around 11 million hectares across the US Midwest could be used to develop bioenergy (Gelfand et al., 2013). This potential identified in the United States suggests similar potential for marginal land to be vegetated and used for bioenergy production within the Canadian Prairies.

Knowledge Gaps

Research is needed to understand the best course of action for farmers and whether land set aside can be an agronomic practice. There is significant research available to provide detail on what types of permanent cover to use for land set aside (specifically for forestry, grassland and biomass crop species), however, which species types are best for regions of the Prairies and when set aside should occur is not well understood. Understanding at what point a poor-yielding part of the field is economic to set aside for permanent cover can help farmers use the practice.

It is also challenging to understand how to manage these lands properly. For example, the knowledge gaps section of Increasing and Managing Trees in Working Agricultural Landscapes (described earlier in this report) summarized challenges of planting and managing trees for them to properly increase biodiversity and store carbon. This includes how to best manage trees for them to be productive (e.g., how often and how to prune tree species), what species are most suitable for the land being set aside and what area has the most potential for tree growth. These challenges also exist for other permanent cover types including grassland species and biomass crops.

Other knowledge gaps include how effective land conversion can be for different regions and soil types. Research is needed to evaluate the GHG mitigation potential for a range of different marginal or degraded lands within the Prairies, and which lands are suitable for conversion based on climatic or soil factors. In addition, more research is needed to evaluate the SOC storage and GHG impacts that can occur on perennial high biomass crops that can be used for bioenergy (Xu et al., 2022).

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24. Reduce soil erosion in areas of high risk

Description

Due to soil, climate, and landform, some areas of the Prairies have inherently high erosion risk. Frequently, these areas have also experienced erosion in the past, so they contain areas of moderate and severe soil degradation from erosion. There is controversary about the effects of mitigating erosion on GHG emissions. whether there is a GHG benefit to mitigating erosion.

Due to low soil quality, the areas with degraded soil give poorer yield than areas without such degradation. Therefore, these areas have reduced efficiency of nitrogen use and are prone to higher N_2O emissions. In addition, areas with soil deposition caused be erosion have N enrichment and readily decomposable carbon (C) (Holz and Augustin, 2021) which also increases the potential for N_2O emission from these depositional areas within the same landscape.

Effect of addressing erosion on SOC and GHG emissions

The impact of soil erosion on soil carbon is controversial. It depends on whether erosion stimulates SOC creation from CO_2 more than CO_2 lost to decomposition (Doetterl *et al.*, 2016). Globally, estimates for the impact of erosion on agricultural land vary from a sink to source (Van Oost *et al.*, 2007). Lal (2003) states that soil erosion represents an important emission source of CO_2 from mineralization of eroded SOC. This potential emission needs to be considered in agricultural policy (Lal, 2014). In contrast, one global analysis indicates that any sink produced by erosion is likely larger than any potential source of emissions (Billings *et al.*, 2010). The erosion reduces SOC on eroding portions of the landscape and the imbalance between resulting SOC stocks and the larger amount of SOC that would be expected given the C input from vegetation on the eroding landscape can produce more SOC sequestration in the eroding landscapes. The SOC in soil depositional locations is less prone to SOC loss, as evidenced by morainal landscapes with erosion having increasing SOC stocks (Quijano *et al.*, 2021). Other cultivated morainal landscapes of the Prairies have shown a net loss of SOC compared to native vegetation (Pennock and Frick, 2001). However, including the SOC deposition in uncultivated wetlands increase the amount of SOC on a full landscape basis (Bedard-Haughn *et al.*, 2006).

The land management is an important factor to this controversy. Under intensive tillage, considering both impacts on cultivated land and downstream deposition, the landscape was a net sink of CO_2 under no-till but a net source of CO_2 under intensive tillage (Izaurralde *et al.*, 2007). Therefore, adopting beneficial management practices that reduce erosion such as no-till, shoulder-season cover crops, and use of perennials, will increase the ability of eroded and eroding land to sequester new C.

Preferentially applying soil amendments such as manure or compost on eroded areas with low C is a practical way of both increasing the productivity of the soil and stabilizing this C addition in soil, especially in soils that are degraded due to high erosion (low initial SOC with an unfilled capacity to store C). In addition, moving topsoil from depositional areas to eroded areas can drastically increase crop production in the Prairies (Schneider *et al.*, 2021).

Due to low soil quality, areas with degraded soil due to past erosion yield less than areas without such degradation. These areas thus have poorer efficiency of nitrogen use so are prone to higher N_2O emissions with the same application rate of N, that is typical of many cropland fields. At the same time, depositional areas are enriched in N and readily decomposable C (Holz and Augustin, 2021) and that concurrence increases the potential for N_2O emission from these depositional areas within the same landscape. Consequently, over a landscape basis, soil erosion can increase N_2O emissions.

Generally, it appears that reducing erosion will not have large effects on SOC emissions. Possibly the greatest impact will be to reduce N_2O emissions by increasing N use efficiency of eroded or eroding land and reducing N_2O emissions by reducing N and C loading in depositional areas.

Potential impact of addressing soil erosion

Due to adoption of no-till and reduced tilled fallow, the amount of land with moderate to high erosion risk across the Prairie provinces in 2011 ranged from 9% of agricultural area in Manitoba to 3% in Saskatchewan (Lobb et al. 2016). This compares to a 25% of land in Saskatchewan to 14% in Alberta, 10 years earlier in 2001. Consequently, there has been much progress on reducing erosion risk.

Without better information on the effect of erosion control on GHG emissions and removals, it is difficult to quantify the potential mitigation of GHG emissions.

Co-benefits and trade-offs

The major co-benefits are environmental degradation from deposition of eroded soil downstream and/or downwind. The eroded sediment can contain soil adsorbed pesticides (Cessna *et al.*, 2006).

There are no major trade-offs from controlling erosion.

Research gaps

Although past and current erosion is important to many prairie landscapes, there is need for research to identify the effects on GHG emissions and removals.

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25. Monitoring practice adoption, soil health, vegetation condition to identify opportunities

Description

Monitoring **practice adoption, soil health, vegetation condition** supports all BMPs and is particularly important for policy development, policy evaluation, and reporting the impact of BMP adoption to stakeholders.

To identify opportunities for initiatives to regenerate degraded areas, it is necessary to identify where there is evidence that the agroecosystem is degraded and that there is unrealized opportunity to apply BMPs to reverse that degradation. With appropriate soil health monitoring for regions and soil types, combined with knowledge of how particular BMPs affect soil health, producers will be better informed of what BMPs would be most useful to increase soil health on their land. This is obviously superior to general recommendations that do not account for site-specific situation.

Information on practice adoption is needed by policy makers and analysts who want to encourage BMP adoption and account for the impacts of current adoption and potential impact of additional BMP adoptions. Investors who want to invest in BMP adoption also want the practice adoption to estimate the size and location of the opportunity. The overlay of the soil degradation with BMP adoption is also important to both these users.

Farmers who want knowledge of the state of their land are less interested in practice adoption but highly interested in monitoring soil health and vegetation conditions. This can help identify the areas across their farm where the land can be improved. If they have adopted BMPs they want to understand how those BMPs are affecting their soils and GHG emissions. Monitoring soils and crop condition can help engage the farmers with the BMPs by showing their impact.

Practice Adoption

Quantifying practice adoption is important to the sector, the public, and policy makers to know the current state and trends in adoption. The adoption is used in quantifying environmental performance, such as GHG emissions and removals. It also supports better targeting of programs and policy to increase adoption of BMPs.

Identifying practices needs to rely on remote sensing to collect data where this is feasible. There are more satellites and sensors so that the capability to identify practices is constantly improving. This includes crop types, cover crops, tillage practices, agroforestry and others (Czerepowicz *et al.*, 2012; Johnson *et al.*, 2016; Ha *et al.*, 2019; Hagen *et al.*, 2020; Laamrani *et al.*, 2020).

However, there is a need to develop more efficient methods to collect data on practices that cannot be reliably obtained from remote sensing. The adoption of 4R nutrient management is not reliably visualized from remote sensing. There needs to be a value proposition for farmers to supply information. The use of offsets and value-chain scope 3 insets to reward farmers financially for good stewardship leading to GHG emission reduction could be a way so that the practices incentivized are identified. This would have to be done so the farmers privacy is protected.

Soil health and vegetation condition

There are a rapidly expanding range of tools and methods to collect data on soils, soil-vegetation relationships, and vegetation conditions (Mitran *et al.*, 2021). Many applications used to identify the soil and vegetation state are based on remotely sensed data (Dong *et al.*, 2019a; Dong *et al.*, 2019b; Mandal *et al.*, 2020). Comparing vegetation performance across land parcels can identify land parcels with better or poorer performance so provide relative performance as affected by land management for the variable weather conditions of the Canadian Prairies (Li *et al.*, 2013).

There are several approaches for farmer-oriented assessment of soil health for which the Haney test (Hargreaves *et al.*, 2019) and the Cornell Comprehensive Assessment of Soil Health (Moebius-Clune *et al.*, 2016) are the most well-known. These have been applied in Canada and elsewhere (Idowu et al., 2008; Van Eerd et al., 2014; Moebius-Clune et al., 2016; Hargreaves et al., 2019; Norris et al., 2020; Mann et al., 2021). Wu and Congreves (2022) developed a prairie-based Saskatchewan soil health scoring framework. It is more comprehensive and so much more costly than other approaches to measure soil health. Planned improvements to the framework are to simplify the type and number of soil attributes measured to make the framework more accessible for broader application.

The improved management of N has a large potential to reduce N_2O emissions, but that impact is not evident to farmers. For this reason, Burton et al., 2021 recommended that programming for 4R include periodic measurement of immediate post-harvest mineral N. Although not a good timing for the purposes of planning N fertilizer application, N at this time provides an indicator of N use efficiency as high soil N at harvest is related to inefficient N use. Some targeted tissue testing for N could also be used as an indicator to show farmers the fields are not being under fertilized. These tests have costs, so these costs are well suited to covered as part of programming to increase adoption of 4R practices.

Effect on GHG emissions

The monitoring of soil health, vegetation condition, or practice adoption does not have direct effects on GHG emissions. Nevertheless, by targeting BMP adoption where monitoring identifies the need and their relative absence, GHG emission reductions from BMP adoption can be achieved more effectively. Monitoring of soil health and vegetation conditions provide producers better information on the state of the land they are managing and, again, provide better information to target BMP adoption.

Current Adoption

Monitoring practice adoption is conducted by the provincial government and federal governments. The latter is done by Agriculture and Agri-Food Canada and by Environment and Climate Change Canada, the latter for GHG inventory purposes. Not-for-profit industry associations also monitor adoption of particular BMP as do some for-profit companies BMP. The degree of coordination and data sharing among players is variable. For many BMPs there are no publicly available sources of good information.

Many agronomic advisory services are using geospatial tools including remote sensing. Many farmers and agronomic advisors do soil testing and use remote sensed imagery to assess crop condition. However, much is not oriented to identify GHG benefits of different BMPs. Nevertheless, increasing agronomic consultant services are also in have business related to reducing net agricultural GHG emissions (e.g., Farmers Edge). These private firms may be the important component of the effective delivery to farmers of geospatial tools to monitor soils and crops for GHG purposes. However, the technologies are largely private IP and their delivery is done for profit.

Aspects of soil health are monitored with conventional soil testing. The use of more comprehensive tests in the Prairies is not widespread.

Potential Adoption and Impact on GHG emissions

The impact on BMP adoption will depend on governments and investors interested in developing programs to encourage adoption by having access to timely and accurate data. Connecting potential BMP adoption with land where the BMPs would have large potential effect on GHG emissions and removals is not well developed yet.

With better knowledge of BMP adoption, the quality of GHG inventory will be improved and provide better information on how agriculture is contributing to GHG emissions and their reduction.

The impact of application of tools for analyzing geospatial data including remote sensing to encourage and maintain BMP adoption for reducing GHG emissions depends on providing value to the farmer for the results and, to make effective for GHG reductions, relating the results with GHG emissions. Similarly, to use soil health monitoring to encourage and maintain BMP adoption for reducing GHG emissions also depends on providing value to the farmer for the results and, to make effective for GHG reductions, relating the results with GHG emissions. Therefore, development effort is needed to meet these requirements and have costs feasible within public- and private-sector programming to increase BMP adoption.

Barriers to adoption

The primary barrier to adoption is the cost and complexity of deploying new techniques to monitor BMP adoption and provide useful indicators of their impact on GHG emissions and removals.

Some farmers are skeptical about the benefits of providing information on their farming practices to others.

The value of using expensive comprehensive soil health monitoring is uncertain until the utility of the monitoring results for land management decision making is demonstrated for the Prairies.

Interpreting vegetation health in terms of identifying areas for BMPs for achieve net GHG emission reduction is not well established.

Co-benefits and Trade-offs

There are no major co-benefits or trade-offs for monitoring BMP adoption.

There are important potential co-benefits to better analysis of geospatial data and soil and vegetation helathmonitoring for improving agronomic management. There are no important trade-offs.

Research Gaps

Developing and providing cost-effective tools to help producers quantify how BMPs are affecting soil health are needed to optimize land management for site-specific conditions.

Research and development of cost-effect soil health monitoring methods that are demonstrated to be useful for guiding land management decision making on the prairies is needed.

There needs to be effective and efficient approaches for sharing of data collection, data, and analysis methods to enable various stakeholders to extract more value from existing and future investments in monitoring.

Developing data collection system on BMPs that provide value exceeding the burden to all participants is needed.

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Summary Table

Table 37. Summary of BMPs

Section Number	BMP Described	Qualitative GHG Mitigation in 2030 (Low, Medium, High)	GHG Mitigation in 2030 (MtCO2e per yr)	Current adoption (ha or %)	Potential adoption in 2030 (ha or %)	Cobenefits list	Trade-offs list	Barriers to Adoption list	High-level Research Gaps list	GHG mitigation (Range of Error) (tCO2e per ha per yr)
1	Reduced Tillage	Low	0.69 Prairie-wide 0.2 in AB 0.24 in SK 0.25 in MB	30-78% for no-till 16-42% for reduced till	80-85% for no-till in SK, AB 50% in reduced till in MB	Reduced soil erosion, Moisture retention, Biodiversity, Reduced wind	Increase in herbicide use and runoff	Increased herbicide resistant weeds may require tillage for weed control. Increasing crop yields and cover crop use requires additional management in no-till systems.	GHG impacts (emissions reductions and removals) need more accurate estimates.	N/A
2	Cover Crops	High	7.54 (maximum adoption) Prairie-wide 2.32 in AB 1.39 in MB 3.83 in SK 0.512 (5% increase in area adopted)	Full season: 11,000 ha Shoulder season: 104,000 ha	Full season: 860,000 ha (maximum adoption) Shoulder season: 17,228,000 ha (maximum adoption) Prairie-wide: 1,414,800 ha (5% increase in area adopted)	Soil health, additional N without fertilizer, Erosion control, Weed and pest suppressant, Reduction in Nitrate leaching, cash-crop yield improvements	Phosphorus losses and nutrient loading, Water conservation,	Costs of seeds, sowing equipment and labour required. Uncertainty regarding timing and value of benefits to farmers in the Prairie region. Full-season cover crops instead of fallow reduce insurance coverage for following cash crops.	Agronomic research for optimization of mixes, species and seeding methods. Total global warming potential impacts from practices (full season vs shoulder season), including N2O emissions. Nutrient losses and loading risks.	N/A
3	Intercropping	Medium	2.9 Prairie-wide 1.02 in AB 0.54 in MB 1.34 in SK	1-5%	5,660,000 (approx. 10% of annual cropping systems)	Height of pulse pods is improved for machine harvesting, Suitable area for pulse crops increases, Reduced pesticide use, Biodiversity, Resilience to adverse weather	Economic performance, Increased complexity or managing pests and diseases	Modifications to seeding equipment and seed costs. In-season weed control, crop rotation planning, and harvesting is more challenging. Lack of equipment and labour required for post-harvest separation of grains. Grain quality may be lower, higher risk practice and social pressure for normal practices. Lack of incentives.	Separation of grains technology, Agronomic management (reducing pests, optimizing seeding and harvesting), GHG impacts, Environmental impacts	0.5503 (0.432 to 0.582)
4	Increased legume crops	Low	0.728 Prairie-wide	6,460,000 ha (22% of annual cropping systems)	8,170,000 ha (28% of annual cropping systems)	Reduced fertilizer-N associated impacts, Reduced fertilizer costs	Slower harvesting, Profitability may decrease with increased production	Harvesting is slower than cereals and canola. Increasing production of peas and lentils may reduce the profitability of domestic production.	Disease management, Market impacts of Fava Beans, Harvestability	0.427

Section Number	BMP Described	Qualitative GHG Mitigation in 2030 (Low, Medium, High)	GHG Mitigation in 2030 (MtCO2e per yr)	Current adoption (ha or %)	Potential adoption in 2030 (ha or %)	Cobenefits list	Trade-offs list	Barriers to Adoption list	High-level Research Gaps list	GHG mitigation (Range of Error) (tCO2e per ha per yr)
5	Reduced field burning of crop residues	Low	0.051 Prairie-wide 0.02 in AB 0.001 in MB 0.03 in SK	0.2% in AB (% of cropped area) 2.3% in MB 1.5% in SK	Elimination of the practice (0% adoption)	Human health improvements, Visibility improvements on roads, Air quality improvements	None	Primarily for flax production due to high crop residue, no tillage practices and lack of market for flax straw.	None	N/A
6	Improved Nitrogen Management	Medium	4.775 Prairie-wide 1.4 in AB 1.1 in MB 2.2 in SK	30-45% in Basic 12-22% in Intermediate 6-11% in Advanced	70% of farms adopting something 4,287,000 in Basic 3,721,000 in Intermediate 12,354,000 in Advanced	Reduced fertilizer-N associated impacts, Reduced fertilizer costs	Profitability of the crop is not maximized. Some enhanced efficiency fertilizer have plastic coatings which can contribute to soil pollution.	Soil testing costs at all levels. Enhanced efficiency fertilizer costs, advanced or precise management required at intermediate and advanced levels. Often not deemed profitable because it doesn't maximize yield potential. No tangible benefits to farmers.	Uncertainty in long-term GHG impacts. Lack of information on some individual practices, most notably fertilizer placement. GHG impacts vary by EEF type due to upstream manufacturing and reduction on direct/indirect emissions.	0.234
7	Biochar addition to soil	Low	1.51 Prairie-wide 1.52 for AB 2.16 for SK 0.85 for MB	Negligible		Employment opportunities	Soil health from residue harvest, Lack of benefits for biochar application	Lack of confidence in the value and benefits from removing residue for biochar production. Lack of investment in portable reactors for residue to become biochar and used as an amendment.	Impact on soil health and crop yields, Understanding the effect of feedstock type for biochar, If biochar can be a co-product of bioenergy pathways	N/A
8	Increasing Organic Amendments Applied to Agricultural Lands	Unclear, adoption scenarios not developed	No data	Highest for manure but low for other amendments. Manure is concentrated in areas with significant livestock production.	Unclear	Crop productivity, water retention, soil structure, soil health, waste diversion	Manure can spread weeds, biosolids can spread diseases or pathogens, nutrient loading in nearby waterways, cost of transporting and distributing amendments	Requires a source of organic amendment nearby to be cost-effective which is not the case for all farmers in the Prairies. Lack of information of non-manure amendments in farm communities.	Adoption data, emission factors vary by composition and treatment, efficacy of the practice for GHG mitigation	3.85 tCO2e/t N for synthetic fertilizer 4.76 tCO2e/t N for manure or slurry 1.55 tCO2e/t N for compost 4.34 tCO2e/t N for digestate

Section Number	BMP Described	Qualitative GHG Mitigation in 2030 (Low, Medium, High)	GHG Mitigation in 2030 (MtCO2e per yr)	Current adoption (ha or %)	Potential adoption in 2030 (ha or %)	Cobenefits list	Trade-offs list	Barriers to Adoption list	High-level Research Gaps list	GHG mitigation (Range of Error) (tCO2e per ha per yr)
9	Rotational grazing	Low	0.716 for Tame pastures Prairie-wide 0.23 for Natural pastures Prairie-wide	For Natural Pastures: 50% in Basic 10% in Intermediate 7.5% in Intensive For Tame Pastures: 45% in Basic 15% in Intermediate 7.5% in Intensive	For Natural Pastures: 37.5% in Basic, 25% in Intermediate, 30% in Intensive For Tame Pastures: 27.5% in Basic, 27.5% in Intermediate, 40% in Intensive	Maintaining and increasing biodiversity, Soil health improvements, Reduced fertilizer-N demand, More resilient to environmental stressors like drought	Intensive management can cause grazing land to be converted to cropland, which results in loss of soil health, nutrient loading in the environment, GHG emissions	Investment required for fencing, water capacity and labour for pasture assessment at higher intensity. Farmers prefer to transition slowly instead of jumping from basic to advanced.	Effect of rotational grazing on soil health including SOC. Need to understand if enteric emissions are effected by rotational grazing (system comparison).	N/A
10	Rotation of Annual Crop with Perennial Forages	Unclear, adoption scenarios not developed	No data			Rotational yield improvements, N addition can reduce fertilizer-N application, weed suppression	Nutrient losses from soil (particularly Phosphorus), Reduces soil pH and increases lime application	Variability in yield and water use efficiency in annuals after perennials in rotation. Challenge in establishing and terminating perennial forage stands. Machinery required for cutting perennial forages.	Effect of the practice on Global warming potential (GHG emissions, SOC), and adoption levels. System comparisons of grazing pastures to hay removal are needed within annual rotations.	N/A
11	Increase and Manage Trees in Working Agricultural Landscapes	Low	0.06 for Riparian Zones 1.07 for Silvopasture	51,000 km of Shelterbelts in SK	14,208 for Riparian zones 921,894 for Silvopasture	Reduce agricultural runoff, reduce erosion, reduce nutrient loading in streams, improve biodiversity, reduce wind damage, improve animal health for pastured livestock	Reduced land in crop production, Costs for repairs, Habitat for livestock predators, Encroachment of wild plant species	Costs associated with tree planting and on-going management. Costs vary by size and landscape type. Some crops (drought-hardy cereals) have a low yield response to shelterbelts. Use of other management practices to improve moisture availability and erosion instead of trees.	Adoption levels, Regional uncertainties, global warming potential including SOC, biomass carbon and all GHGs, when benefits are observed to farmers	3.94 for Riparian zones 2.39 (2.2 to 2.57) for Shelterbelts 1.16 (-0.73 to 3.04) for Silvopasture

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12	Reduce Deforestation to Agriculture	High	10.28 Prairie-wide based on total avoidance based on 2020 rates	243,885 ha Prairie-wide converted in 2020 Rate of deforestation from 2001 to 2020 was 14,648,000 ha Prairie-wide	A no net deforestation policy could eliminate deforestation to agriculture	Improvements to air quality, supports biodiversity of plants, animals, soil biota, microbes. Social benefits such as hunting and hiking.	Reduced albedo, keeping land out of food production, potential leakage where crop/livestock intensity is increased elsewhere	Value of land in agricultural regions for pasture and grain production. Cultural and economic drivers of land clearing as farmers often share costs of clearing. Land obstacles like trees reduce the efficiency of large machinery used in the Prairies.	Assessing the ecological vs agricultural benefits, accurately measuring and reporting on forest conversions and the associated losses of carbon in the Prairies.	11.71 across Prairies for all current deforestation
13	Reduce Loss of woody biomass in Agriculture (Avoided Conversion of Shelterbelts)	Low	0.31 Prairie-wide	2,491 km of shelterbelts lost between 2008-2016, roughly 311 km/yr	586 km/yr could be avoided Prairie wide	habitat for biodiversity, landscape aesthetics, erosion control, storm protection, more favourable microclimate, berries and fruits. Public benefits	Loss of productive farmland and competition over resources (H2O, nutrients, light) with adjacent crops	Cost and labour associated with tree management and maintenance. Competition between shelterbelts and crops. Inconvenience when using large farm machinery.	Lack of research on how best to overcome the cost and labour barriers to preserving and planting new shelterbelts.	N/A
14	Avoided Conversion of Grassland, Pasture and Hay land	Low to High, depending on effect of BMP adoption on the size of ruminant herd and their methane emissions.	12.7 for complete avoided conversion Prairie-wide	8,920 ha of grassland were converted to cropland in 2020	2,500,000 ha 7,002,960 ha currently in grassland Prairie-wide	Provides habitat For wildlife and livestock, Supports diverse plant and insect species, Serves as sustainable feedstock For bioenergy, Improves soil quality, Provides medicinal and biotechnology materials	If the avoided conversion of grassland, pasture, and/or hay also avoids a reduction in the cattle herd that would have occurred otherwise, then the emissions of methane from that avoided loss of cattle can have larger radiative forcing increase than the decrease from the SOC conserved. Forgone profits for producers and land owners, Increased intensification or leakage on other agricultural lands, Methane emission when grazed	Economic pressure/ value of land for agricultural production. Profit differential between cropped and perennial grazed lands. Lack of understanding for climate mitigation potential and benefits of ecosystem services provided by perennial grasslands.	Lack of research on variables contributing to uncertainty of emission factors.	N/A

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15	Wetland Conservation and Restoration	Medium	4.45 Prairie-wide for Wetland Conservation only		235,000 ha for Conservation	Aquifer recharge, Sediment and nutrient retention, Floodwater attenuation, Enhanced biodiversity	Mosquito proliferation, Land use change, Associated land depression	Costs associated with conservation and restoration of wetlands, particularly restoration and maintenance. Disinterest from landowners. Lack of organizations able to conduct restoration activities. Regulatory process and disinterest from regulators (regionally dependent).	Poor quality data for sequestration and emission. Uncertainty in GHG measurements due to variation in scale of measurement types. Description of sites within the literature rarely includes wetland classes.	6.08 (-2.55 to 18.54) for Restoration (across 33 years) 18.92 (13.03 to 24.21) for Conservation (after 10 years)
16	Crop Residue Bioenergy	Medium	4.27 Prairie-wide	515 M L of ethanol (per year)	2162 M L/yr across Prairies	Additional return to growers for crop residues, Employment opportunities, reduce Canada's dependency on bioethanol imports	Soil health from residue harvest, Increased transportation-related emissions, Risk of soil compaction from field traffic	Lack of adequate value proposition for farmers to remove residue for bioenergy purposes. Lack of supply of residue from growers to biorefineries. Long-term future of bioethanol in Canada is uncertain based on federal government policies.	Soil health and crop yield impacts of crop residue removal for the Prairies. Other potential agricultural biomass substrates efficacy in bioenergy production (woody biomass). Economics of biomass-based bioenergy for the Prairies to inform government policy.	1976 tCO2 per M L of ethanol
16	Crop Residue Bioenergy and Carbon Capture and Storage	High	11.42 Prairie-wide	130 M L of ethanol (per yr)	2547 M L/yr across Prairies	Further reduces the carbon intensity of ethanol and biodiesel production from agricultural feedstocks. More incentives federally and provincially may improve the cost-effectiveness of CCS technology.	High costs for CCS	Costs of infrastructure, maintenance and monitoring of CCS facilities. Appropriate land close to biorefineries required for CCS.	Storage efficacy and concerns around permanence of CO2 stored.	4485 tCO2 per M L of ethanol with CCS

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17	Organic and Regenerative Ag Systems	Unclear, adoption scenarios not developed	No data	763,000 ha in Organic; Regenerative: 11,000 ha in Full season CC, 104,000 for Shoulder season CC, 50% farms under zero till		Increased biodiversity, Promotes innovation, Efficient nutrient cycling from grazing CC	Heavy reliance of Organic production on tillage, For Organic systems: Increased competition with weeds and limited nutrient supply	Transition period to organic can be long. Pest management is challenging in organic systems and there is social stigma against fields with weeds. Often requires new equipment for pest and nutrient management. Lack of knowledge (technical and agronomic) for both regenerative and organic systems. Lack of access to technical services (field mapping, soil testing, advanced tools).	Technical and agronomic methods (i.e., how to implement practices in a combined way best) needed for both systems. Uncertainty in GHG emission impact. Lack of definition on what constitutes regenerative ag	N/A
18	Integrated Crop-Livestock Systems (ILS)	Unclear, adoption scenarios not developed and impact depends on the effect of BMP adoption on size of ruminant herd and their associated methane emissions	No data			Organic amendment, Improved N use efficiency, Improved crop yield & SOM	Nutrient leaching, Increased land demand, Altered N balance	Social barriers like dominant markets and farming systems, complexity in management, financial costs for infrastructure.	Lack of data on specific practice adoption, Varying operation of ILS, Most efficient land mgt. systems not yet identified	9.34 for integrating livestock in annual rotation 0.783 for moderately grazed natural vegetation

*Low: less than 2 Mt CO2e/yr by 2030, Med: between 2 and 5 Mt CO2e/yr by 2030, High: greater than 5 Mt CO2e/yr

Acronyms:

CC	Cover Crops	H2O	Water
CCS	Carbon Capture and Storage ILS	ILS	Integrated Crop-Livestock Systems
CO2	Carbon Dioxide	N	Nitrogen
N2O	Nitrous Oxide	SOC	Soil Organic Carbon
GHG	Greenhouse Gas	SOM	Soil Organic Matter

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