Potential for Reductions in Greenhouse Gas Emissions from Native Rangelands in Alberta

Technical Scoping Document

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Executive Summary

Native rangelands in Alberta contain large reservoirs of organic carbon and may be sequestering additional atmospheric CO_2 through their response to elevated CO_2 levels. Mitigation practices to increase atmospheric CO_2 sequestration or otherwise reduce greenhouse gas (**GHG**) emissions were evaluated in this report. Improving range health through the effective application of rangeland management principles may increase C storage on rangelands that are currently rated as unhealthy or healthy with problems. However, the potential to reduce GHG emissions by this mechanism is small because most native rangelands in Alberta are healthy and the few estimates of C gain due to improved range health are small and inconsistent. Conversion of annual cropland to rangeland has the potential to increase C sequestration substantially, but this practice is most appropriately considered as a mitigation practice for annual cropland. Inclusion of legumes in seeding mixes and application of compost have good potential to increase C storage when annual cropland or degraded lands is converted to rangeland, but have limited potential to reduce GHG emissions on healthy rangelands. Overall, the potential to adopt practices that reduce GHG emissions on existing Alberta rangelands is small.

Introduction

Native rangeland consists of land on which the historic plant community is principally native grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing. In most cases, native rangeland is extensively managed through livestock control rather than by agronomic practices such as fertilization, mowing or irrigation. Native rangeland includes lands that have been invaded by non-native species but are managed as range and lands revegetated with native plant species by seeding or through natural recovery processes (e.g. abandoned cultivation, reclaimed industrial sites, and field-scale re-grassing to permanent native cover).

Based on the 2006 Census of Agriculture, 6.5 million hectares of farmland were classified as “natural land for pasture” in Alberta (Statistics Canada 2007). Approximately 70% of this area is public land administered by Alberta Sustainable Resource Development and other public institutions (Alberta Sustainable Resources Development (SRD) - Lands Division 2001). Alberta rangelands exist in grassland, forest and alpine ecosystems.

The dominant activity on native rangelands is grazing of beef cattle to produce calves that are sold to backgrounding and feedlot operations. Public rangelands in Alberta provide over 1.6 million Animal Unit Months (AUMs) of forage each summer for about 14 percent of all Alberta's beef cattle (SRD 2003). One AUM is equivalent to approximately

450 kg of forage dry matter. Thus, forage yield on native rangeland is about 0.25 Mg per hectare, but this varies widely among locations and years due to differences in soil, landscape position, vegetation and weather.

Native rangelands also provide numerous other ecosystem services. Healthy rangelands protect soils from erosion, capture and beneficially release water and maintain biological biodiversity (SRD 2007). Native rangelands also store a large amount of organic C: one hectare of rangeland in Alberta contains 50 to 200 Mg of C in soil organic matter (to 0.3 m), 2 to 10 Mg C in plant biomass and 1 to 2 Mg C in litter. Total C storage in native rangeland of Alberta is equivalent to about three times the current annual emissions of all greenhouse gases in Canada (based on storage of 100 Mg C ha^{-1}, 6.5 million hectares and GHG emissions of 721 million tonnes CO_2 eq in 2006 (Environment Canada 2008)).

The objective of this report is to estimate the potential of mitigation practices to reduce greenhouse gas (GHG) emissions on native rangelands of Alberta. This will be accomplished by first evaluating baseline estimates of GHG emissions from native rangelands and then assessing potential reductions from four mitigation options.

Baseline estimates of net GHG emissions from native rangelands

Baseline estimates of net GHG emissions can be evaluated on an area basis or a production basis. Estimating GHG emissions per hectare is similar to the approach used for estimating reductions in GHG emissions due to changes in tillage practice (Alberta Environment 2008b). However, production of calves can be achieved with many different feeds and production practices. If a change in production practice on native rangeland causes a change in production on other land types (e.g., tame pastures), then a reduction in net GHG emissions on native rangeland may not provide an actual reduction in GHG emissions. Vergé et al (2008) estimated GHG emissions per unit of live-weight gain (**LWG**). They estimated that the intensity of GHG emissions from the Canadian beef industry had declined from 16.4 to 10.4 kg CO_2 eq kg^{-1} LWG between 1981 and 2001. However, due to the overall increase in beef production, total GHG emissions had increased from 25 to 32 Tg CO_2 eq over this period. Thus, advantages and disadvantages exist for both approaches of estimating GHG emissions from native rangelands. Both will be considered in the remainder of this report.

The greenhouse gases emitted from native rangelands are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O).

CO_2 emissions, farm energy: One source of anthropogenic CO_2 emissions is farm energy use. Vergé et al. (2008) identified six types of farm energy use for the Canadian beef industry, but only two of these are relevant for native rangelands: energy required for hauling livestock to and from native rangelands and energy required to manufacture and supply machinery used in operations. Neither of these activities emits significant quantities of CO_2 nor is likely to be reduced by mitigation practice. Therefore, the baseline estimate of CO_2 emissions from farm energy use was assumed to be negligible.

CO_2 emissions, C storage: Another source of anthropogenic CO_2 emissions is caused by changes in C storage due to management activities. In baseline evaluations, changes in C storage are generally assumed to be negligible because management practices are not changing and C pools are likely at steady-state. However, baseline changes in C storage may not be zero in native rangelands due to historical changes in land management practices or current changes in climate. Due to the long-time period required for soil organic C (**SOC**) in native rangelands to recover from cultivation (Dormaar et al. 1990), major changes in management that occurred even decades previously may still be causing a gradual change in SOC levels. Overall, this effect is likely a factor in few sites. Current changes in climate may have a more widespread impact. Atmospheric data and transport models suggest that C storage in terrestrial systems has likely increased over the last few decades due to physiological responses (e.g., elevated atmospheric CO_2 concentrations increase photosynthesis rate and water use efficiency, particularly for C_3 plants) or shifts in plant populations and age structure (e.g., regrowth of forests in temperate latitudes or invasion of grassland by shrubs) (Houghton 2007). However, increases in temperature or drought may reduce potential C storage. Climate-induced changes in C storage have been evaluated for native rangelands in several different ways:

- Micrometeorological measurements: Continuous monitoring of CO_2 fluxes on ungrazed rangelands in good ecological condition showed that five of eight U.S. locations typically were sinks for atmospheric CO_2 during a study period of five to seven years (Svejcar et al. 2008). At the locations most similar to Alberta, average annual C sequestration was 0.5 Mg C ha^{-1} yr^{-1} in a northern mixed prairie at Mandan, North Dakota and 1.1 Mg C ha^{-1} yr^{-1} in a shortgrass prairie at Nunn, Colorado. Estimates of C sequestration were sensitive to net primary production and precipitation, which were above average at these locations for the study period. Annual variation in precipitation also contributed to large annual variations in C sequestration in a micrometeorological study conducted on a northern temperate grassland near Lethbridge, Alberta (Flanagan et al. 2002). Carbon sequestration was 1.1 Mg C ha^{-1} over the whole study period in this study due to a large net gain in 1998, small net gain in 1999 and a small net loss in 2000. Integration of CO_2 flux measurements with biophysical models (Huxman et al. 2004; Zhang et al. 2004) or remotely sensed data (Wylie et al. 2007) may allow more robust estimates of C sequestration to be obtained. For example, based on a regression model between remotely sensed data and measured CO_2 fluxes, Wylie et al. (2007) concluded that C sequestration during the years 1998 to 2001 was generally negligible in the Northern Great Plains of the U.S. (Montana, Dakotas and Colorado), though it may have been significant in some areas.
- Elevated CO_2 experiments: Based on a meta-analysis of free-air CO_2 experiments (FACE) conducted with a wide range of vegetation types, Jastrow et al. (2005) concluded that elevated CO_2 levels (usually doubled) increased soil C at a median rate of 0.19 Mg C ha^{-1} y^{-1} over time periods of 2 to 9 years. Rates of increase in soil C were greater for grassland soils (exceeding 0.4 Mg C ha^{-1} y^{-1}) due to large increases in root production. Carbon sequestration will be less for actual increases in atmospheric CO_2 concentrations. Gill et al. (2002) found that the relationship of C sequestration to atmospheric concentration was highly non-linear, with much greater response to historic than future increases to atmospheric CO_2 levels. Carbon sequestration due to

elevated CO_2 levels may be limited by nutrient deficiencies and increasing temperatures (Houghton 2007).

- Measured changes in C storage: There are currently no reliable measurements of change in C storage over time for native rangelands on the Canadian prairies. Effective monitoring requires a well-designed program conducted over several decades due to considerable spatial variability in C storage, considerable temporal variability in C fluxes, and expected small change in C storage. Possible increased C storage in shrub communities, which have increasingly encroached on grasslands in part due to climate change, may represent evidence for C sequestration in rangelands. In Mixed and Moist Mixed Grasslands of Saskatchewan, SOC was not different between shrub and grassland communities, but phytomass C in shrub communities was approximately 6 Mg C ha^{-1} higher than in grassland communities (Colberg 2007).
- Models: Negra et al. (2008) estimated that most grassland and shrubland soils in the U.S. had minimal changes in C storage in the 1980s and 1990s based on output from the Century model.

In summary, there are some grounds for suggesting that climate change may be increasing C storage in native rangelands, but supporting evidence is weak for rangelands in general and absent for Alberta rangelands in particular. If it was demonstrated that C storage in native rangelands was increasing due to climate change, then another issue that would have to be resolved is whether these increases could be used as GHG offsets or were simply baseline modifiers.

CH_4 emissions: Most anthropogenic CH_4 emissions on native rangelands come from enteric fermentation by grazing ruminants. The tier 2 methodology of the Intergovernmental Panel on Climate Change (IPCC) (used for Canada's National Inventory) calculates CH_4 emissions from enteric fermentation by multiplying calculated gross energy intake by the fraction of gross energy intake converted to methane (Y_m) for each livestock category (IPCC 2006). Y_m is assumed to equal 6.5% ± 1% for beef cattle under typical grazing conditions. Thus, CH_4 emissions can be calculated from estimated forage consumption:

$$CH_4 \text{ (ent. ferm.)} = ANPP * FG * 18.45 * (Y_m/100) / 0.05565$$

Where:

CH_4 (ent. ferm.) = CH_4 emissions from enteric fermentation, kg CH_4 ha^{-1} yr^{-1}

ANPP = above-ground net primary production (dry matter basis), Mg ha^{-1} yr^{-1}

FG = fraction of ANPP consumed by grazing animals

18.45 = Gross energy content of dry matter, GJ Mg^{-1} (IPCC 2006)

Y_m = percent of gross energy in feed converted to methane

0.05565 = energy content of methane (GJ kg^{-1} CH_4)(IPCC 2006)

To estimate global warming potential of CH_4 equivalent to CO_2, multiply by 21 (IPCC 2006).

Alternatively, CH_4 emissions could be estimated by multiplying emission rates per animal by the number of animals per hectare. Either approach requires estimates of long-term forage productivity (10+ years). Possible impacts of climate change on forage quantity and quality (Milchunas et al. 2005) may have to be accounted for in baseline estimates.

Small amounts of anthropogenic CH_4 on native rangelands come from manure excreted by grazing animals. Emissions of CH_4 from manure can be calculated from forage consumption using the IPCC tier 2 methodology (IPCC 2006):

$$VS = GE * ((1 - DE\%/100) + 0.04) * ((1\text{-}ASH)/18.45)$$

$$CH_4 \text{ (manure)} = VS * 1000 * B_0 * 0.67 * MCF$$

Where:

VS = volatile solid excretion on a dry matter basis, Mg ha^{-1} yr^{-1}

GE = gross energy intake, GJ ha^{-1} yr^{-1} (=ANPP * FG * 18.45)

DE% = % digestibility of forage consumed by beef cattle on pasture (60%, Boadi et al. 2004)

0.04 = urinary energy expressed as a fraction of GE (IPCC 2006)

ASH = ash content of manure as fraction of dry feed intake (0.08, IPCC 2006)

18.45 = gross energy content of dry matter, GJ Mg^{-1} (IPCC 2006)

CH_4 (manure) = CH_4 emissions from manure excreted on rangelands, kg CH_4 ha^{-1} yr^{-1}

1000 = conversion factor from Mg to kg

B_0 = maximum CH_4-producing capacity for beef cattle (0.19, IPCC 2006)

0.67 = conversion factor of m^3 CH_4 to kg CH_4

MCF = methane conversion factor for beef cattle on pasture (0.01, IPCC 2006)

N_2O

Of the N excreted by cattle on native rangeland in Alberta, the fraction emitted as N_2O varies from 0.16 to 1.7%, depending primarily on moisture regime (Environment Canada 2008). Cattle obtain approximately 15 kg N per Mg of consumed forage on natural pastures (Janzen et al. 2003) and incorporate approximately 2 kg N Mg^{-1} into body mass (based on feed:LWG ratio of 15: 1 and 2.7% N in cattle live-weight (Janzen et al. 2003)). Thus, cattle excrete approximately 13 kg of N per Mg of forage consumed on natural pastures and N_2O emissions can be calculated:

$$N_2O = ANPP * FG * 13 * FN_2O * 44/28$$

Where:

N_2O = N_2O emissions from N excreted by cattle on native rangeland, kg N_2O ha^{-1} yr^{-1}

ANPP = above-ground net primary production (dry matter basis), Mg ha^{-1} yr^{-1}

FG = fraction of ANPP consumed by grazing animals

13 = N excreted per Mg of forage consumed on native rangeland (kg N Mg^{-1})

FN_2O = fraction of excreted N lost as N_2O

44/28 = weight ration of N_2O to N

To estimate global warming potential of N_2O equivalent to CO_2, multiply by 310 (IPCC 2006).

Emissions of N_2O from soil nitrification and denitrification in natural grasslands are not included in national GHG inventories (IPCC 2006).

Total baseline emissions and implications for mitigation

Most GHG emissions on native rangelands are from enteric fermentation, with appreciable contributions also from N_2O (Table 1). Emissions of CH_4 and N_2O per hectare are proportional to consumed forage yield. Thus, a mitigation practice that increased consumed forage yield by 50% would also increase GHG emissions per hectare by 50% unless C storage or farm energy use changed. Put another way, C storage would have to increase by 0.16 Mg C per Mg increase in consumed forage yield to counteract the increased emission of CH_4 and N_2O (1 Mg of forage consumption emits 603 kg CO_2 eq as CH_4 and N_2O, equivalent to 0.16 Mg of C). Conversely, a mitigation practice that reduced consumed forage yield by 50% will reduce GHG emissions per hectare by 50% unless C storage or farm energy use was affected.

Estimates of GHG emissions per LWG are independent of consumed forage yield because feed conversion efficiency was assumed to be independent of forage productivity (Table 1). Some improvement in feed use efficiency may occur with increasing productivity, e.g., due to reduced net energy requirements for activity. However, many factors influence feed use efficiency, and similar feed use efficiencies can be achieved on less productive rangeland using appropriate grazing practices. Feed use efficiency is difficult to measure for livestock grazing on native rangelands and is not monitored by grazing managers. Estimated GHG emissions per LWG on native rangeland (6-10 kg CO_2 eq kg^{-1}) were slightly less than that estimated for the Canadian beef industry as a whole (10.4 kg CO_2 kg LWG in 2001; Vergé et al. 2008).

Table 1. Baseline estimates of net GHG emissions[z] on native rangelands as a function of forage productivity.

ANPP[y]	FG	AUMs	CO_2, farm energy	CO_2, ΔC stocks	CH_4, enteric fermentation	CH_4, manure	N_2O	Total
($Mg\ ha^{-1}$)		(ha^{-1})			kg CO_2 eq ha^{-1} yr^{-1}			
0.25	0.5	0.3			48-65[x]	1	1-13[w]	50-80
0.5	0.5	0.6	≈0	≈0 (?)	96-131	3	3-27	101-160
1	0.5	1.1			191-261	5	5-54	202-320
2	0.5	2.2			383-522	11	10-108	404-641
					kg CO_2 eq kg^{-1} LWG			
All productivity levels			≈0	≈0 (?)	6-8	0.2	0.2-2	6-10

[z]Positive values indicate net emissions to atmosphere.
[y]Abbreviations: ANPP, above-ground net primary production; FG, fraction of ANPP consumed by grazing animals; AUMs, animal unit months; LWG, live-weight gain.
[x]Range estimated assuming Y_m = 5.5 to 7.5%.
[w]Range estimated assuming FN_2O = 0.16 to 1.7%.

Changes in C storage can potentially have a large impact on net GHG emissions. For example, an increase in C storage of 0.1 Mg C ha^{-1} yr^{-1} would reduce net GHG emissions by 366 kg CO_2 eq ha^{-1} yr^{-1}. If this reduction was occurring on all 6.5 million hectares of native rangeland in Alberta, GHG emissions would be reduced by 2.4 million tonnes CO_2 eq, equivalent to 1% of all Alberta emissions in 2006.

Mitigation option #1: improve grazing practices

Alberta Sustainable Resource Development (**SRD**) have developed a code of practice to maintain and improve public rangelands in Alberta (SRD 2007). Rigid prescriptions and templates were not developed because rangelands are diverse and dynamic. Rather, the approach was to flexibly apply rangeland management principles to achieve objectives for a healthy rangeland: maintenance of plant vigor, protection and building up of the soil, perpetuation of the forage resource and maintenance of a stable flow of products and other societal benefits. The principles of sustainable rangeland management are 1) balance livestock demand with the available forage, 2) distribute livestock grazing impact, 3) avoid grazing during vulnerable periods and 4) provide effective rest after grazing. A system to assess the health of rangelands and associated riparian areas has been developed by the Alberta government (Adams et al. 2003). Grazing leaseholders of public lands in Alberta are required to participate in periodic health assessments and invest in management practices that result in stable health in the case of healthy rangelands or improved trend lines in the case of rangelands categorized as healthy with problems or unhealthy. Grazing leaseholders of public lands in Alberta are also required to maintain and provide stocking records annually on a field basis.

The current requirements for grazing leaseholders of public lands in Alberta represent a useful approach to implement and verify improved grazing practice. However, a number of issues would need to be resolved prior to using this approach to quantify potential GHG credits:

1. Project-based emission reductions or removals must not be required by law (Alberta Environment 2008a). Thus, this mitigation activity might only be implementable for private lands. A GHG credit might be possible for improved grazing practice on public lands if a practice could be shown to be additional and more effective than current requirements, but this possibility seems remote due to the flexible nature and probable effectiveness of current requirements.
2. Because the basis for this mitigation activity is reduced GHG emissions due to improvements in grazing practice, practitioners who currently maintain rangeland in a healthy state would not be able to participate. This may be unacceptable to many in the ranching community because good stewardship of rangeland is considered a feasible responsibility, i.e., should be business-as-usual. Currently, about 60% of public land leased for grazing in Alberta is rated as healthy, while 33% is rated as healthy with problems and 7% is rated as unhealthy (Milk River Watershed Council Canada 2007; Barry Adams, personal communication).
3. The requirement that project-based emissions be real, demonstrable, and quantifiable (Alberta Environment 2008a) is challenging due to the diversity and complexity of rangelands, the wide range of practices that could be used to improve rangeland health, and the limited number of studies that have quantified GHG emissions as affected by grazing practice or rangeland condition. The effectiveness of mitigation activities must be quantified relative to baseline estimates:
    a. CO_2, farm energy: Changes in CO_2 from farm energy depend on the practice(s) adopted to improve range health. A reduction in stocking rate or season of grazing would have negligible impact on farm energy use, while installation of fences or watering units may represent significant farm energy use.
    b. CO_2, C storage: Similarly to baseline estimates, difficulty in measurement and lack of relevant studies limit evidence for C storage change due to grazing management. Based on a global review, Conant and Paustian (2002) concluded that converting heavily overgrazed land to moderate grazing would increase soil C in North America by an average of 0.16 Mg C ha^{-1} y^{-1}, but noted less potential in drier areas and negligible over-grazing on the Canadian prairies. Protection from grazing did not affect C storage in studies conducted in southern Alberta (Henderson et al. 2004) or Saskatchewan (Colberg 2007). Canada's National Inventory assumes that there is no potential for increasing soil organic C on grassland due to improved grazing practice (Environment Canada 2008). A reliable assessment of change in C storage due to improved

range health would require implementation of an intensive, well-designed quantification effort.

   c. CH_4: Emissions of CH_4 may be affected by improvements in range health due to impacts on forage productivity and quality (McCaughey et al. 1999), but insufficient information is available to assess this impact.
   d. N_2O: Emissions of N_2O may be affected by improvements in range health due to impacts on forage productivity and protein, but insufficient information is available to assess this impact.

4. Project-based emission reductions or removals must also have clearly established ownership (Alberta Environment 2008a). This is primarily an issue for public land, but participation may still be possible. For example, the state of New Mexico joined the Chicago Climate Exchange to allow qualified grazing lease-holders of public land to obtain C credits (two thirds of revenue to lease-holder, one third to state land trust) (de Steiguer et al. 2008). The province of Alberta may be able to pursue similar opportunities in the future.

Based on the above considerations, the potential to reduce GHG emissions by improving grazing practice is likely small due to the limited area of unhealthy rangelands and the probable small difference in C storage between healthy and unhealthy rangelands.

Mitigation option #2: convert cropland to native rangeland

The initial conversion of native grassland to cropland reduced SOC by approximately 22% and increased emissions of N_2O (Environment Canada 2006). Thus, reductions in GHG emissions may be possible by converting cropland back to native rangeland.

In the wetter regions of Alberta, cropland was obtained by deforestation rather than breaking of grassland. Deforestation in western Canada generally does not reduce SOC, but reduces the amount of C stored in live and dead vegetation (Fitzsimmons et al. 2004; Environment Canada 2008). A protocol currently exists for afforestation (Alberta Environment 2007), and thus evaluation will be restricted to grassland ecosystems.

This mitigation option might be best included within a protocol to reduce GHG emission by establishing perennial forages (tame or native pastures, perennial hay) on annual cropland:

- conversion of cropland to rangeland is not a typical rangeland practice
- the baseline is the same for establishment of perennial forages, tame pastures or natural grasslands (i.e., annual cropland)
- the mechanisms for change in GHG emissions as similar for perennial forages, tame pastures and natural grasslands (i.e., increased C storage), although the magnitude of change will vary with forage type and management
- issues related to reversals are the same for perennial forages, tame pastures and natural grasslands

Quantification of baseline GHG emissions may be contentious because the products and activities on cropland are highly variable and not easily compared to those on rangeland. Cropland GHG emissions largely consist of CO_2 from cropping operations and N fertilizer manufacturing and N_2O from N fertilizer and crop residues; CH_4 emissions may be significant if crops are primarily used to feed livestock. Rangeland GHG emissions largely consist of CH_4 from enteric fermentation (Table 1).

The simplest, and probably conservative, approach is to assume that reduction in GHG emissions due to conversion of cropland is only due to increased SOC. Less CO_2 and N_2O will be emitted after native vegetation has become established and land is primarily used for grazing, and increased CH_4 emissions should only partially offset these reductions (Boehm et al. 2004).

Canada's National Inventory estimates the loss in SOC due to conversion of natural grassland to cropland using the following equation (Environment Canada 2008):

$$\Delta SOC\ (t) = 0.28 * SOC_{agric} * (1 - e^{-0.12t})$$

where:

$\Delta SOC(t)$ = change in SOC for the t^{th} year after conversion, Mg C/ha

t = time since breaking, years

SOC_{agric} = 0- to 30-cm SOC from the National Soil Database within CanSIS for the soil profile under an agricultural land use (Cropland), Mg C ha^{-1}

The same equation could be used for conversion of cropland to natural grassland, although an initial lag in C gain will likely occur due to time required for plant establishment. The equation implies that SOC losses or gains are directly proportional to steady-state SOC levels, i.e., the potential to gain C is greater for soils with a higher initial C content. The maximum change in SOC is 28% of agricultural SOC (estimated from National Soil Database), with 91% of maximum change occurring during the first 20 years after conversion.

The time period required for SOC to recover to pre-cultivation levels may be greater than estimated with this equation. In the Dry Mixedgrass Subregion, SOC was 12% lower for rangeland that had been cultivated for 10 to 15 years and then abandoned for 55 years than for rangeland under native vegetation (Dormaar and Smoliak 1985). Based on SOC composition, Dormaar et al. (1990) estimated that recovery of abandoned cropland would take at least 75 years for the Dry Mixedgrass Subregion and 150 years for the Foothills Fescue Natural Subregion.

This equation was based on studies of paired comparisons of SOC in native grasslands and annual croplands, and is therefore not appropriate for conversion of croplands with a high proportion of perennial crops. Cropland practices such as tillage and fallow frequency also impact SOC, but correction for these factors is likely not warranted due to their moderate impact. However, SOC in croplands with a high proportion of perennials may be as high or higher as that in natural grasslands (Environment Canada 2008).

A number of recent studies have evaluated the recovery of SOC due to conversion of annual cropland to grassland (Table 2). Most of these studies were not grazed, but SOC levels for grazed systems are likely to be intermediate between harvested and unharvested treatments. Harvesting native grassland did not affect SOC in the studies by Willms et al. (2008). Experiments often included fertilization, which may not represent actual practice on native rangeland. However, recovery of SOC requires significant input of N, and if these are not provided by inclusion of legumes or fertilization, then recovery of SOC will be slowed (Table 2). Establishment of non-native species had inconsistent impact on SOC gains: establishment of Russian wildrye or crested wheatgrass had a similar impact on SOC in the studies by Willms et al. (2008), but establishment of crested wheatgrass reduced SOC gains in the study by Ellert (personal communication). Overall, the observed changes in SOC due to establishment of grass were consistent with the equation used by Environment Canada (2008): SOC change in the Brown soil zone after 12 years in treatments with sufficient N fertility was 6 to 7 Mg C ha^{-1}, comparable to 12-year estimates of 6 to 11 Mg C ha^{-1} assuming agricultural SOC levels of 30 to 50 Mg C ha^{-1}, while SOC change in the Dark Brown soil zone after 12 or 13 years in treatments with sufficient N fertility was 11 to 14 Mg C ha^{-1}, comparable to 12-year estimates of 11 to 15 Mg C ha^{-1} assuming agricultural SOC levels of 50 to 70 Mg C $^{-1}$.

Reductions in GHG emissions due to conversion of annual cropland to grassland will be reduced if an appreciable fraction of land within a project is converted back to annual cropland or if appreciable rangeland outside of a project is converted to annual cropland to maintain crop production. In the 2006 National Inventory, 192 kha of natural grassland was converted to cropland, while net GHG reductions due to increased perennials was estimated at 4200 Gg CO_2 eq (roughly 5700 kha if sequestering at 0.2 Mg C ha^{-1} yr^{-1}) (Environment Canada 2008). Conversion of cropland depends on the relative profitability of livestock and crop production, land suitability, available technology (e.g., zero tillage may allow more marginal land to be cropped) and capacity of land managers to shift between livestock and crop production. An assurance factor could be obtained through consultation with industry and government experts (e.g., managers of Canada's Greencover program).

Table 2. Change in soil organic C (ΔSOC) of grass stands compared to wheat-based crop rotations.

Location	Soil	Comparison	Duration (yr)	ΔSOC[z] (Mg C ha^{-1})	ΔSOC/yr	Comments	Reference
Swift Current, AB	Brown	Unfertilized CWG vs. FWW	10	-3 ± 5	-0.3 ± 0.5	Grass harvested for hay, poor stand, N deficient	Campbell et al. (2000)
Swift Current, AB	Brown	Year 4 vs. Year 0 of native grassland established on long-term FW	4	2 ± 1	0.5 ± 0.3	Average of 4 treatments	Iwaasa and Schellenberg (2005)
Bow Island, AB	Brown	Unfertilized grass (*Thinopyrum intermedium*) vs. FW	12	3 ± 2	0.3 ± 0.2	Grass harvested for hay	Bremer et al. (2008)
		Fertilized grass vs. FW	12	6 ± 2	0.5 ± 0.2	Grass harvested for hay	
Onefour, AB (initially native grassland)	Brown	Undisturbed native grass (*Stipa-Bouteloua*) vs. FW	12	7 ± 2	0.6 ± 0.2	Grass not harvested	Willms et al. (2008)
		Disturbed native grass vs. FW	12	4 ± 2	0.3 ± 0.2	Grass not harvested	
		Undisturbed native grass vs. FW	12	6 ± 2	0.5 ± 0.2	Grass harvested at 8 cm	
		RWR vs. FW	12	6 ± 2	0.5 ± 0.2	Grass harvested at 8 cm	
		CWG vs. FW	12	7 ± 2	0.6 ± 0.2	Grass harvested at 8 cm	
Lethbridge, AB	Dark Brown	Grass (*Agropyron* spp.) vs. FW	8	6 ± 2	0.8 ± 0.3	Grass not harvested, values recalculated to equiv. mass	Bremer et al. (1994)
Lethbridge, AB (initially native grassland)	Dark Brown	Undisturbed native grass (*Stipa-Bouteloua-Agropyron*) vs. FW	13	11 ± 3	0.9 ± 0.2	Grass not harvested	Willms et al. (2008)
		Disturbed native grass vs. FW	13	14 ± 3	1.1 ± 0.2	Grass not harvested	
		Undisturbed native grass vs. FW	13	14 ± 3	1.1 ± 0.	Grass harvested at 8 cm	
		RWR vs. FW	13	13 ± 3	1.0 ± 0.2	Grass harvested at 8 cm	
		CWG vs. FW	13	12 ± 3	0.9 ± 0.2	Grass harvested at 8 cm	
Lethbridge, AB	Dark Brown	Unfertilized native grass (*Stipa-Bouteloua-Agropyron-Koeleria*) vs. FW	12	8 ± 5	0.6 ± 0.4	Grass harvested for hay	Ellert, personal communication
		Fertilized native grass vs. FW	12	12 ± 5	1.0 ± 0.4	Grass harvested for hay	
		Unfertilized CWG vs. FW	12	3 ± 5	0.3 ± 0.4	Grass harvested for	

						hay	
		Fertilized CWG vs. FW	12	6 ± 5	0.5 ± 0.4	Grass harvested for hay	
East-central SK	Dark Brown, Thin Black, Black-Gray	Restored grass (dominated by *Agropyron* and *Bouteloua* spp.) vs. wheat-based rotations	5-12		0.6-0.8	Comparison of SOC at 12 sites, grass was lightly grazed or hayed	Mensah et al. (2003)
South-central SK	Brown and Dark Brown	Restored grass (tame and native spp.) vs. FW	≈8		0.3-2.9	3 locations, higher values on shoulder positions	Nelson et al. (2008)

[z]Difference in SOC between grass (CWG=crested wheatgrass (*Agropyron cristatum*), RWR =Russian wildrye (*Elymus junceus*)) and wheat-based systems (FW=fallow-wheat , FWW=fallow-wheat-wheat). Values are means ± approximate 95% confidence interval. SOC determined to ≈15 cm, equivalent mass basis, most measurements include roots and other macro-organic matter.

Mitigation option #3: interseed legumes in native rangeland

In South Dakota, interseeding yellow-flowering alfalfa (*Medicago sativa* ssp. *falcata*) directly into native rangeland increased forage productivity, forage quality and C storage (Mortenson et al. 2004; 2005). The interseeded alfalfa persisted for a period of at least 36 years. This may represent a significant opportunity to reduce GHG emissions in Alberta rangelands if similar benefits could be obtained here. Native legumes are available that might provide similar benefits, e.g., purple prairie clover (*Dalea purpurea* Vent.).

Baseline GHG emissions estimated in Table 1 should be appropriate for this mitigation option. The effectiveness of this mitigation option depends on:

a. CO_2 emissions, farm energy: The change in CO_2 emissions from farm energy will primarily be caused by the seeding operation. For typical seeding operations on the prairies, CO_2 emissions are approximately 40 kg CO_2 eq ha^{-1} from diesel fuel (Dyer and Desjardins 2003). These emissions will be greater for native rangelands due to the rough terrain and greater risk of establishment failure. However, provided the seeded legume persists, CO_2 emissions from farm energy use will be small.
b. CO_2 emissions, C storage: In South Dakota, interseeded stands had 8 Mg ha^{-1} more SOC to a depth of 30 cm on average than control stands (3, 14 and 36 years after establishment, similar increase for all comparisons) (Mortenson et al. 2004). Interseeded stands had 1.1 Mg ha^{-1} more C in above-ground vegetation than control stands and, based on assumed root:shoot ratios of 10:1 or 27:1, 8 or 21 Mg ha^{-1} more C in roots than control stands (slightly smaller increase for 3-year stand). The lack of root C measurement and similar C gains for 3-year stands as 14- or 36-year stands make these estimates somewhat tenuous, but measured gains were statistically significant and at least 9 Mg C ha^{-1}. Based on values calculated with root:shoot ratio of 10, C gains were approximately 20 Mg C ha^{-1}, equivalent to 3.7 Mg CO_2 ha^{-1} yr^{-1} if assumed linear for 20 years (similar to that for converting cropland to grassland in the Dark Brown soil zone). However, the efficacy of this practice has not been confirmed in other studies. A study has recently been established at Swift Current, SK, to evaluate the inclusion of legumes in seed mixes when converting cropland to native rangeland (Iwaasa, personal communication). However, studies to evaluate the feasibility and C sequestration potential of seeding legumes directly into native range have not been conducted on the Canadian prairies.
c. CH_4 emissions: Interseeding alfalfa into native rangeland increased live above-ground biomass by an average of 84% in South Dakota (Mortenson et al. 2005), thus increasing potential CH_4 emissions per hectare (Table 1). Legumes may also affect the proportion of gross energy intake lost as CH_4 (Y_m): McCaughey et al. (1999) found that Y_m was 7.1% for cows grazing alfalfa-bromegrass pastures (78% alfalfa), but 9.5% for cows grazing grass-only pastures. However, dry matter intake was also greater for cows grazing alfalfa-grass pastures, and thus

cows grazing alfalfa-grass pastures produced more CH_4 per hectare than cows grazing grass-only pastures (411 vs. 374 L CH_4 d^{-1}). If we assume interseeding a legume in native range increases forage productivity by 50% and reduces Y_m by 10%, then CH_4 emissions per ha would increase by about 35% and CH_4 emissions per LWG would decrease by about 10%. Further study is required to reliably assess this impact.

d. N_2O emissions: Interseeding alfalfa into native rangeland increased protein concentration as well as productivity (Mortenson et al. 2005), and therefore will also increase N_2O emissions per hectare. For example, if we assume interseeding a legume in native range increases forage productivity by 50% and protein concentration by 20%, then N_2O emissions per ha would increase by 85% and N_2O emissions per LWG would increase by 23%. Further study is required to reliably assess this impact.

Based on the study in South Dakota, the potential for reductions in GHG emissions by seeding legumes on native rangeland depends primarily on potential change in C storage: the potential reduction in GHG emissions due to increased C storage was about 20 times greater than the estimated increase in GHG emissions due to CH_4 or N_2O emissions. However, evidence is required to document whether seeded legumes are likely to establish, persist and improve productivity and C storage in Alberta native rangelands to the extent observed in the study from South Dakota.

Mitigation option #4: apply compost to native rangeland

The application of compost to native rangelands may improve forage productivity due to improved nutrient availability and water holding capacity. Although increases in C storage due to the C applied in compost is not considered sequestered C (Schlesinger 2000), increases in C storage due to increased forage productivity may represent sequestered C.

Addition of organic wastes generally increases soil water retention at both field capacity and wilting point (Haynes and Naidu 1998; Krull et al. 2004). However, available water-holding capacity is the difference in water retention at field capacity and wilting point, and therefore is generally not affected by addition of organic wastes. In southern Alberta, Larney et al. (2000) found that compost addition did not affect available water-holding capacity at two sites and reduced available water-holding capacity at a third site. Organic additions may provide a greater benefit in sandy soils with low initial SOC because the water content at field capacity may be increased more than the water content at the wilting point (Rawls et al. 2003). High rates of organic addition (50 to >100 Mg ha^{-1}) to sandy soils have shown large increases in available water-holding capacity in some field studies, but not others (Krull et al. 2004). Organic additions must be repeated in order to maintain elevated SOC levels and detrimental impacts may occur if organic amendments are high in monovalent cations (Haynes and Naidu 1998).

Previous studies have shown that manure application can dramatically improve forage production on native rangelands, but may negatively affect species composition and may contribute to environmental contamination if manure is surface applied on land susceptible to runoff or if manure is applied at rates greater than plant requirements (McKenzie et al. 2003). In comparison to manure, composting reduces N availability by ≈50% (Eghball 2000; Helgason et al. 2007) and P availability by ≈30% (Zvomuya et al. 2006). Thus, compost has reduced nutrient availability than manure, but still represents an effective source of nutrients.

Baseline GHG emissions estimated in Table 1 should be appropriate for this mitigation option. The effectiveness of this mitigation option depends on:

a. CO_2 emissions, farm energy: The change in CO_2 emissions from farm energy will be primarily due to transportation and spreading operations, which will depend on distance and logistics. For example, CO_2 emissions from fuel consumption alone would be about 21 kg CO_2 eq Mg^{-1} of compost (assuming round-trip hauling distance of 20 km, 10 Mg compost $trip^{-1}$, 3 L diesel fuel km^{-1} and 3.5 kg CO_2 eq L^{-1} diesel fuel). GHG emissions from loading and machinery manufacturing would increase this value somewhat.
b. CO_2 emissions, C storage: The potential for compost application to increase soil C sequestration due to enhanced productivity depends on the amount and persistence of enhanced productivity and the decomposition rate of increased C inputs. If C inputs shifts toward above-ground residues and less recalcitrant plant residues (a probable shift with increasing fertility), then SOC levels may not increase with increasing productivity. Quantification of C sequestration will be difficult because the change in C storage due to increased forage productivity will likely be much less than the change in C storage due to compost C addition.
c. CH_4 emissions: CH_4 emissions from applied compost will be negligible (Larney and Hao 2007), but enhanced forage consumption will increase CH_4 emissions per hectare from enteric fermentation.
d. N_2O emissions: N_2O emissions will increase due to application of N and due to increased consumption of forage protein.

Compost is superior to manure as an amendment for rangelands due to lower levels of weed seeds, pathogens, parasites and available nutrients (Larney and Hao 2007). However, widespread application of compost on healthy rangelands would not be recommended because compost is still an effective source of nutrients, healthy rangelands have low nutrient requirements and excessive nutrient availability has negative impacts on rangeland function (McKenzie et al. 2003). Sandy soils are most likely to increase available water holding capacity due to compost addition, but generally also have the lowest nutrient demand and the greatest risk of nutrient loss to the environment. A more appropriate use of compost would be in the conversion of disturbed areas or degraded croplands back to rangeland.

Conclusions

Native rangelands in Alberta contain organic C stocks equivalent to about three times total annual GHG emissions in Canada. The most certain and effective means to preserve this stored C is to maintain rangelands in good condition.

Current GHG emissions from Alberta rangelands are relatively low because C storage is close to steady state and forage productivity is low. Climate change may be affecting plant and soil C dynamics, but few studies and little evidence are available to assess this.

The potential to reduce GHG emissions on land currently managed as native rangeland in Alberta is low. Very little native rangeland is currently considered unhealthy and maintenance of healthy rangeland is a priority for rangeland managers. Possible improvements due to seeding of legumes or application of compost are likely only feasible and effective for land that is being reclaimed as native rangeland, not land that is already managed reasonably well. Increases in C storage due to improved rangeland health are difficult to quantify because a small increase must be detected against a large, highly variable background.

Conversion of degraded cropland to native rangeland can potentially reduce GHG emission due to increased C storage. This mitigation activity is best considered as a protocol option for the conversion of annual croplands to perennial forages[1]. .

[1] This option will be considered by a parallel working group, led by Sheilah Nolan at Alberta Agriculture and Rural Development.

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