Evaluation of Avoided Grassland Conversion and Cropland Conversion to Grassland as Potential Carbon Offset Project Types

An Issue Paper prepared for the Climate Action Reserve

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Note Regarding 2015 Revision

This version of this document has been revised from the original document. The original version of the issue paper was posted to the Reserve website in December of 2012. Subsequent to that time, some of the calculations and estimates contained in this paper were evaluated by an interested stakeholder and found to be in error. The errors were confirmed and corrected by the original authors, in cooperation with Reserve staff. Below is a summary of the revisions contained in this version of the document:

Location of Revision	Page number(s)	Description of Revision
Section 1.3, paragraph 1	ix	The last sentence was edited to reflect the change in assumption of the land use transition from 20 years to 30 years.
Table 2	ix	An error was discovered in the calculation of these values. The updated values represent not only a correction of the error, but also adjustments to several inputs to the estimation to better reflect the system dynamics. The new values are, on average, an order of magnitude lower than the previously-published values.
Section 1.4, paragraph 1	xii	The last sentence was added to support the change in assumption of the land use transition from 20 years to 30 years.
Section 3.2	6	Text and equations edited to reflect the assumed transition time of 30 years.
Table 5	7	Removed rows related to “Uncultivated” and “Set aside (CRP)” from the Land Use category, as these are not relevant.
Section 4.1.1, paragraph 1	9	The last sentence was added to support the change in assumption of the land use transition from 20 years to 30 years.
Table 12	34	These values were edited based on the updated values in Table 2. In addition, a new column for the mean cost was inserted.
Table 13	35	These values were edited based on the updated values in Table 2. In addition, a new column for the mean technical potential was inserted.
Table 17	48	These values were edited based on the updated values in Table 2. In addition, a new column for the mean cost was inserted.
Table 18	49	These values were edited based on the updated values in Table 2.
Body text		The text was edited in sections which reference specific numbers from the above tables to reflect the updated numbers.
Map captions	x, xi, 35, 49	Maps were not updated to reflect the changes to the figures. Where the changes to the figures in the above tables would result in changes to the data displayed in a map, the caption was updated to reflect this.

For questions related to the revised issue paper, please contact the Reserve Policy Team at policy@climateactionreserve.org.

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1 Executive Summary

This Issue Paper was prepared in support of the Climate Action Reserve's continued exploration of potential carbon sequestration project activities in the agricultural sector. The review and analyses presented here focus on two potential land use change (LUC) project activities: (1) avoided conversion of threatened grasslands (AGC), and (2) conversion of marginal cropland to grassland (CCG). This Issue Paper characterizes and evaluates these activities as potential carbon offset project types within the United States.

1.1 Project Type Definitions

In its Request for Proposals (RFP) for this Issue Paper, the Reserve proposed a project activity for AGC defined as:

> *The conversion of grassland into production cropland or other land development can rapidly decrease soil carbon stocks as a result of soil disturbance and removal of permanent vegetation. By permanently conserving grassland that otherwise would have been converted into alternative use, substantial emissions of carbon may be avoided.*

The Reserve also proposed a project activity for CCG defined as:

> *Setting aside cropland that is otherwise capable of producing food and converting it into a permanent non-tree vegetative cover, such as grassland, can substantially increase soil carbon sequestration as a result of eliminating soil disturbance, increasing permanent belowground biomass in roots and shoots, and possibly increasing overall organic matter inputs from permanent vegetation compared to a cultivated system.*

1.2 Recent Trends in Cropland and Grassland Conversions

Two primary datasets were utilized to characterize the amount of land in the United States that transitions to and from both grassland and cropland, the National Land Cover Database (NLCD) and the National Resources Inventory (NRI). Although both these datasets use slightly different definitions to reflect various grassland and cropland land uses, they offer a generally consistent picture of conversion rates that have been observed in recent history.

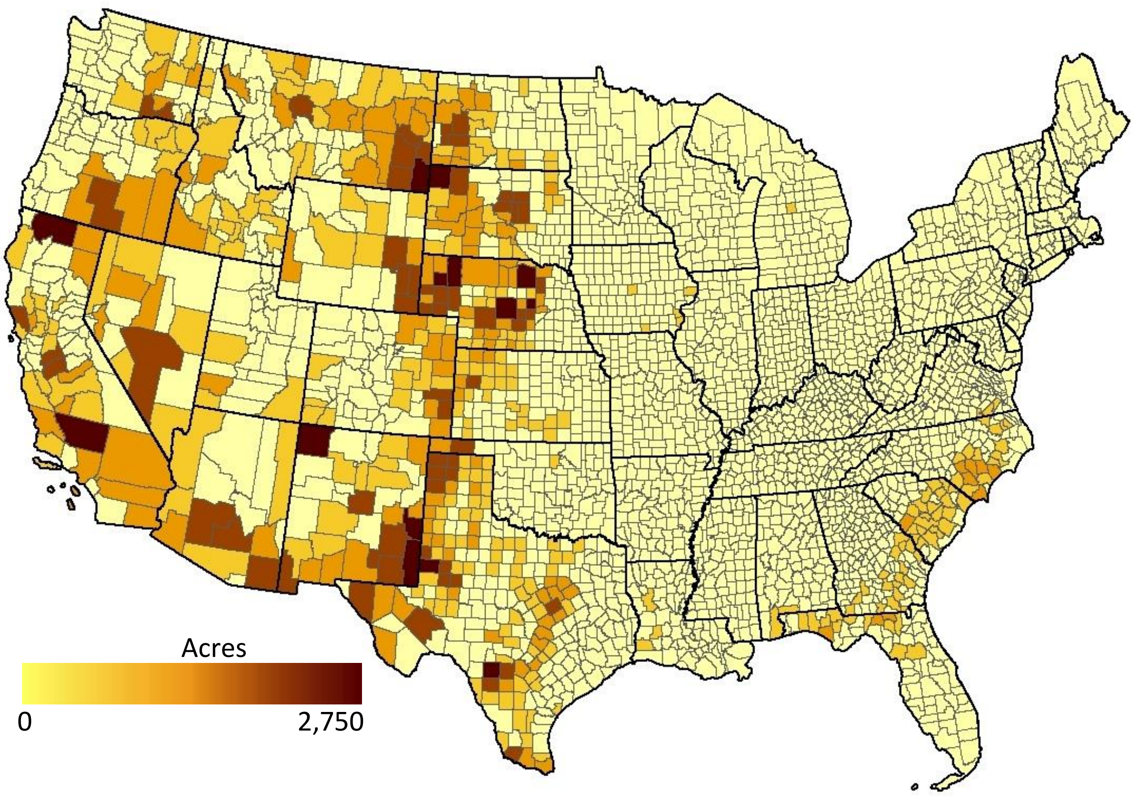
The NLCD dataset is based on land cover classified from satellite imagery. The land use changes observed from 2001-2006 are presented in Table 1.

Table 1: Conversion matrix for major land classes from NLCD data (2001-2006), 1,000s acres

Converted from:	Converted to: Cult. Crops	Developed	Grass/ Herb	Pasture/ Hay	Shrub/ Scrub	Forest	Other
Cult. Crops	307,877	698	323	19	370	284	410
Developed	0	106,151	2	0	1	1	1
Grass/Herb	598	373	280,669	120	1,878	1,488	677
Pasture/Hay	13	465	281	133,615	410	605	344
Shrub/Scrub	308	318	1,292	203	418,139	2,946	782
Forest	480	756	5,324	186	4,968	492,410	507
Other	446	289	984	161	665	410	123,910

Across the “grass/herb,” “pasture/hay,” and “shrub/scrub” land cover types, a total of approximately 929,000 acres were converted to cropland between 2001 and 2006, or about 185,000 acres on an annual basis. These land use changes are primarily concentrated in the Northern Great Plains, but additional hotspots also exist in the Southwest and in grassland areas of both California and the Pacific Northwest (Figure 1). These areas present the highest potential for AGC emission reductions.

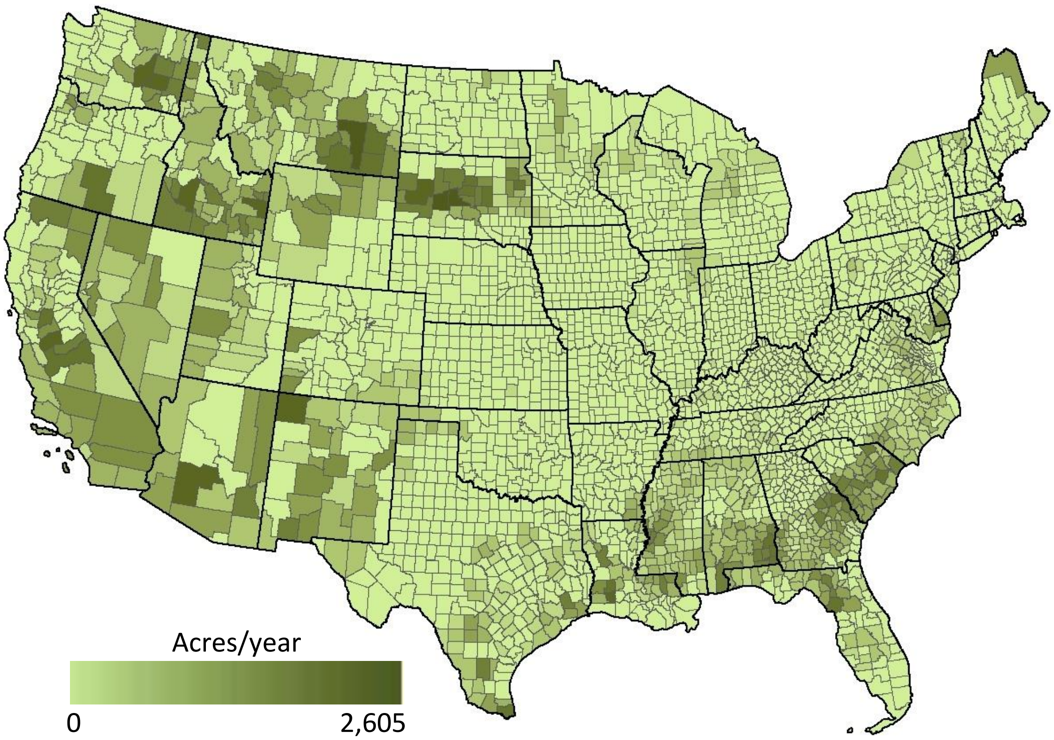
Figure 1: Gross conversion rates for grassland and shrub/scrub to cropland



Across the United States, a total of approximately 712,000 acres were converted from cropland to grassland between 2001 and 2006, representing a gross annual conversion of 142,400 acres. These conversions may be considered to represent "business-as-usual" cropland rates, although they are likely to be related to the availability of incentive programs such as payments for setting aside cropland under the Conservation Reserve Program.

The conversion of cropland to grassland occurs in a similar geographic distribution to the trends observed for grassland to cropland conversion (Figure 2). Together, these two figures suggest that land use is very dynamic in these hotspots.

Figure 2: Gross conversion rates for cropland to grassland and shrub/scrub



1.3 Estimated Cost and Technical Potential for Emission Reductions

To estimate the potential emission reductions that could be achieved by AGC and CCG activities, a simplified IPCC Tier 2 approach was employed that utilizes soil organic carbon (SOC) reference values which are multiplied by factors incorporating the effect of land use type and tillage practices. Using reference values and factors derived for the US national GHG inventory (Ogle, et al., 2003; 2006), the following emission factors were derived to represent AGC and CCG land use transitions over 30 years for each climate zone in the US:

Table 2: Estimated emission factors for AGC and CCG activities

IPCC Climate Zone	Estimated AGC Emission Factor Range ($tCO_2e\backslash ac\backslash yr$)	Estimated CCG Emission Factor Range ($tCO_2e\backslash ac\backslash yr$)
Cool Temperate Dry	(0.06) – 0.66	(0.06) – 0.40
Cool Temperate Moist	(0.22) – 0.98	(0.22) – 0.65
Warm Temperate Dry	0.16 – 0.83	0.23 – 0.60
Warm Temperate Moist	(0.17) – 1.24	(0.17) – 0.51
Tropical Dry	0.24 – 0.97	0.27 – 0.71
Tropical Moist	(0.15) – 1.66	(0.15) – 0.69

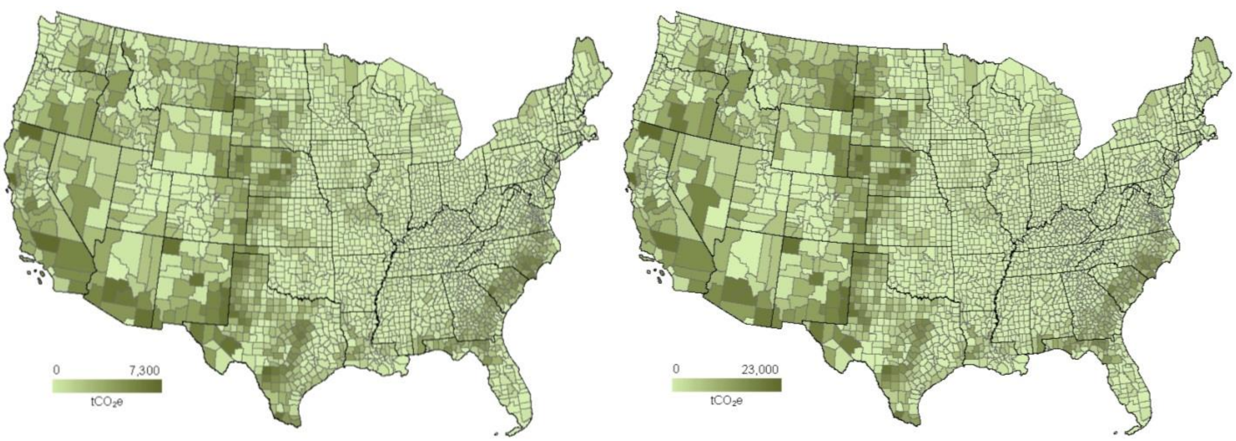
Numbers in parentheses are negative, indicating an increase in GHG emissions.

These emission factors reflect a range of changes in SOC that would be expected from AGC and CCG activities, but do not account for the additional sources, sinks, and reservoirs that would likely produce further emission reductions for these project types. For example, both project types would likely also avoid N_2O emissions in the baseline cropland management scenario following the application of nitrogen fertilizer.

To estimate the technical potential for emission reductions through AGC activities, these emission factors were applied to the 185,000 acres of grassland which annually convert to cropland. The result is an estimated technical potential for AGC in the US ranging from 3.6 to 153.9 thousand metric tonnes of CO_2 equivalent (tCO_2e) each year.

Most of the potential AGC emission reductions are concentrated along the western Great Plains, extending in a north-south corridor running from the border of Montana and North Dakota down to Texas and New Mexico. Additional hot spots for AGC technical potential appear across California and in the Atlantic coast in the Southeastern US (see Figure 3).

Figure 3: Geographic distribution of estimated technical potential for AGC activities



Note: The maps on left and right present the low- and high-end estimates for annual emission reduction potential for each county, respectively. [The magnitude of these reductions, as labeled, is incorrect. However, the geographic distribution of the potential reductions is still accurate.]

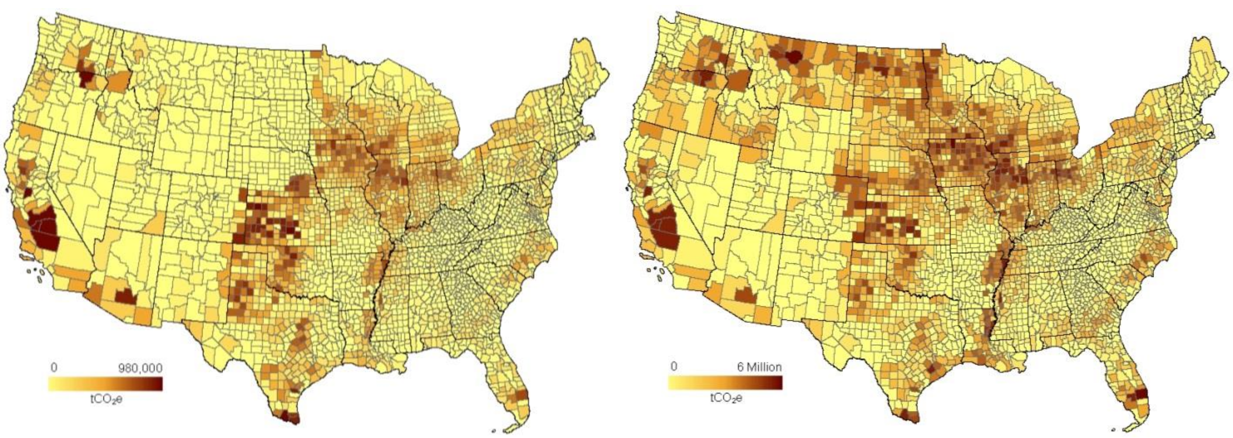
The payment rates from the USDA Grassland Reserve Program (GRP), a voluntary program which support conservation activities on grazing land in the US and that, as of 2011, had accumulated enrollment of roughly 300,000 acres. Converting from per-acre payment rates, a range from \$0.68 to \$6.10 per tCO_2e was found. It is important to recognize, however, that these payment rates reflect the payments sufficient to incentivize participation in the GRP, and that any additional commitments, costs, or restrictions on land management that would be involved with participation in an AGC offset project would justify a higher cost per ton of emission reductions achieved.

Although the limitation of CCG activities to marginal or degraded cropland is not strictly necessary to achieve emission reductions, the Reserve has stated an interest in considering such a criterion, and the estimates of technical potential below reflect two potential cutoffs for degraded cropland (see Table 18). Using NRCS Land Capability Classes to define degraded land, two estimates are presented for classes greater than 5 and for classes greater than 6. These two estimates create a range of technical potential for CCG activities ranging from -8.7 to 77 million tCO_2e per year.

The dramatically larger scale for potential emission reductions from CCG activities compared to AGC activities relates primarily to the land base that is classified as degraded in the US. Using Land Capability Classes of 5 and 6 as cutoffs for a definition of degraded lands results in the classification of 45% and 44% of cultivated cropland in the US as degraded, respectively. For contrast, CCG technical potential is estimated for 136-140 **million** acres of US cropland, while AGC technical potential is estimated only for 180 **thousand** acres of US grassland and shrubland.

CCG technical potential also follows a distinct geographic distribution from that found for AGC activities (see Figure 4). CCG technical potential appears to be distributed into several discrete zones. Considering the low end emission factor estimates, CCG hotspots appear from the Texas and Oklahoma panhandles up through western Kansas. Northern states bordering the Mississippi River also appear prime targets for CCG activities. When the high-end estimate emission factors are applied, the Cool Temperate Dry Zone moves from -0.06 to 0.40 tCO_2e/ac/yr, and hotspots appear across northern Montana and North Dakota, with a few additional clusters appearing in eastern Oregon and Washington. In both high-and low-end emission factor estimates, the southern end of California's Central Valley also appears to have substantial CCG potential stretching from Kern through Fresno counties.

Figure 4: Geographic distribution of estimated technical potential for CCG activities



Note: The maps on left and right present the low- and high-end estimates for annual emission reduction potential for each county, respectively. [The magnitude of these reductions, as labeled, is incorrect. However, the geographic distribution of the potential reductions is still accurate.]

1.4 Protocol Development Considerations

In general, there are two important qualities of AGC and CCG project activities which support a project duration and/or crediting period on the scale of 20-30 years. First, these projects would be carried out in the context of competing agricultural land management decisions. For several practical and generally self-apparent reasons, agricultural land management is applied over a timeframe that is generally shorter than the forestry sector (for which the Reserve has approved project durations up to 100 years). Second, the primary source of emission reductions for these projects is expected to be the protection of SOC stocks. Under the prevailing IPCC approach for SOC accounting, soils are generally assumed to achieve equilibrium SOC stocks after twenty years under a particular land management regime. Thus, the emission reduction benefits achieved by AGC and CCG project activities should generally be complete within this timeframe. However, more recent research supports a timeframe of thirty years or greater for certain lands to achieve equilibrium. The estimated emission factors in this paper assume that it takes thirty years to move from one equilibrium to another.

There do not appear to be suitable data to generate localized rates to represent common practice for cropland and grassland conversions. Conversion rates varied substantially between available datasets and a variety of land use change drivers exist that are likely evolve over time. Similarly, values to establish financial thresholds for a standardized additionality test (e.g., by comparing land values for a project area as cropland and grassland) are not immediately forthcoming. Publicly available data on cropland and rangeland rental rates were highly variable, and in several locations cropland rental rates were observed to be lower than rangeland rental rates. This does not necessarily preclude the potential for the Reserve to define financial thresholds for conversion (such as the approach for avoided conversion projects in the Reserve's Forest Project Protocol), but no clear values for this additionality threshold or additional conversion uncertainty discounts are apparent from publicly available data.

The sources, sinks, and reservoirs (SSRs) are generally similar for both AGC and CCG project types, with the SOC pool expected to be the primary source of emission reductions for both. In addition to accounting for changes in SOC stocks in the baseline and scenario, an AGC and CCG protocol would also need to accommodate accounting for emission from livestock (particularly emissions due to enteric fermentation) and the emission of N_2O related to the application of nitrogen fertilizers. Accounting methods for both livestock and fertilizer emissions are readily available in other current offset protocols.

To estimate initial SOC stocks, a protocol could reasonably employ carbon values derived from regional soil carbon survey data and/or field sampling. However, direct measurement of SOC stocks and changes in the field currently represent a major expense that is likely to significantly affect the financial feasibility of these projects. The authors thus encourage the Reserve to consider alternatives an exclusive reliance on field sampling for SOC stocks such as the use of soil carbon models or emission factors. In the event that lookup values for SOC stocks are used from regional soil surveys, projects should be expected to at least perform field sampling to confirm the distribution of soil types across the project area.

As both AGC and CCG project activities would remove of cropland acreage from production, it is very likely that the corresponding reduction in commodity crops supplied would be offset through some level of additional production from farms outside the project area, including those that may be developed through the conversion of grassland, forestland, or other land types to crop production. This indirect land use change effect has been studied in a substantial volume of literature regarding biofuel mandates and is sufficiently well articulated to enable a simplified approach to account for market effects leakage.

In particular, the reduced-form model described by Plevin et al. (2010) by for quantifying indirect land use change is encouraged as a transparent approach for estimating market effects leakage.

$$ILUC = \frac{NDF \times EF}{T \times Y}$$

Where:

$ILUC$ emissions due to indirect land-use change; tCO_2e per unit of yield reduction (e.g., tCO_2e/bushel)
NDF Net Displacement Factor, the ratio of land area brought into crop production to the area subject to reduced yields due to project activities; dimensionless
EF average GHG Emission Factor for the land area brought into crop production; tCO_2e/ac
T timeframe over which project-induced yield reductions are considered (e.g., project life); years
Y the annual yield reduction induced by project activities over the timeframe T; unit of yield reduction per acre per year (e.g., bushel/ac/yr)

This reduced-form model simplifies the multitude of drivers and responses simulated in global equilibrium models to comparatively intuitive factors. This approach would be followed by a multiplication of this emission level by the amount of the yield reduction, but is also amenable to a further simplified approach where the unit of supply reduction could be acres of land.

A review of several studies investigating "slippage" in the CRP suggest a maximum domestic rate of indirect land use change of 20% on a land area basis (i.e., for every 5 acres of cropland set aside, one additional acre will be brought into production elsewhere), but more recent inquiries have suggested domestic leakage rates for cropland set asides may even be as low as 3.7% (Wu, 2000; Roberts & Bucholtz, 2005; Fleming, 2010). Studies of indirect land use change in the context biofuels mandates suggest that expanding the scope of accounting internationally can produce a scale of leakage ranging from 21% to 89% on a land area basis (see Table 27 in Appendix E: Leakage Lit Review for a range of published values).

The biological component of emission reductions that could be achieved through AGC and CCG activities also correspond to a risk these carbon benefits could be reversed by natural and anthropogenic disturbances. As SOC is expected to be the dominant source of emission reductions in the project, risk management could reasonably focus on monitoring for potential soil disturbance, particularly tillage. In general, the policy options available for mitigating reversals of biological carbon storage such as the buffer pool approach, would also be suitable for managing reversal risks for AGC and CCG project types.

1.5 Concluding Thoughts

Both AGC and CCG project types present viable opportunities to achieve significant emission reductions in several geographic hotspots for these land use conversion threats across the country. One of the primary points of emphasis for protocol development will need to be finding an appropriate quantification strategy for SOC stocks and changes that does not rely solely on direct field measurement, and the authors encourage the Reserve to consider the use of biogeochemical models and/or IPCC-style emission factors as two additional potential options. It is likely that both of these project types will also be subject to substantial leakage, but transparent options are available to account for it. Although these two project types present a unique combination of carbon accounting needs, a range of baseline and with-project quantification approaches should be adaptable to these project types. Following consideration of the range of issues contemplated in the Reserve's Request for

Proposals for this Issue Paper, the authors share the opinion that both AGC and CCG project types present a strong opportunity for the expansion of current land use change activities to mitigate GHG emissions through the development of a new offset project protocol.

2 Introduction

This Issue Paper was prepared in support of the Climate Action Reserve's continued exploration of potential carbon sequestration project activities in the agricultural sector. The review and analyses presented here focus on two potential land use change (LUC) project activities: (1) avoided conversion of threatened grasslands (AGC), and (2) conversion of marginal cropland to grassland (CCG).

This document has been designed and organized to provide direct responses to questions and issues raised in the original Request for Proposals released by the Reserve in April 2012[1], as well as to additional feedback provided by Reserve staff following review of a rough draft of this Issue Paper.

This Issue Paper begins by briefly providing background information including definitions for grasslands and related ecosystem types and management practices. This section then outlines the methodological approach used to estimate the emissions reductions that may be achieved from these two project activities in the United States.

The following two chapters review and characterize each of the two potential project activities in turn. They first characterize the types of management activities that may comprise AGC and CCG projects, and then discuss available datasets and recent land use trends in grassland and cropland land use conversions. These chapters then estimate the technical potential for each project activity to achieve emissions reductions, and the estimated costs of doing so.

Both AGC and CCG activities present unique scientific and policy considerations that must be addressed through a well-designed methodological framework. The next chapter of the Issue Paper discusses these policy issues and provides some recommendations for transparent and effective solutions. These issues include eligibility, additionality, leakage, and permanence policies, as well as technical aspects including conservative science-based quantification and accounting strategies.

The major findings of this Issue Paper are briefly summarized in a final concluding section.

In the preparation of this Issue Paper, the authors compiled a significant amount of supplementary information that will provide important context for understanding and interpreting our main findings. These supporting materials are included in five appendices that provide: elaboration of the definitions and categories for land use types in relevant data sets and conservation programs; data tables on observed land use trends; a review of policies and regulations that may affect the additionality of project activities; a case-by-case review of current and proposed AGC- and CCG-related offset project protocols, methodological tools, and accounting policies; and finally a review of literature regarding leakage and indirect land use change accompanied with a discussion of lingering issues and policy options.

[1] The original RFP can be found on the Reserve website here: http://www.climateactionreserve.org/wp-content/uploads/2010/09/RFP_Issue_Paper_for_Land_Use_Change_040612.pdf

3 Background and Supporting Information

3.1 Definitions of Grassland and Related Terms

The definition of project activities comprising avoided conversion of grasslands (AGC) and conversion of cropland to grassland (CCG) hinges critically on the definitions of grasslands and croplands themselves. Ideally, existing definitions used by government agencies, land managers, producers, conservation organizations, academic researchers, and other carbon protocols are to be used. We begin by reviewing existing definitions of grasslands and examining how these can be operationalized in the context of a protocol. A short overview of regional grassland and rangeland ecosystems in the US follows the section on definitions.

As the Reserve indicated in their RFP, there are a variety of terms often associated with "grassland", including rangelands, prairie, pasture, savannah, and steppes. Though grasslands are traditionally considered a sub-group of rangelands, many of these terms often refer to the same land (Flynn, et al., 2009). It is important to clarify definitions of rangeland and related terms. Table 3 contains rangeland types and biomes present in North America.

Table 3: Rangelands in North America

Rangeland Type	Other Names	North American Biomes
Grasslands	prairie, steppe, pampas, swards, meadows, velds	Tallgrass Prairie, Shortgrass Prairie, Mixed Grass Prairie[2], Southern Mixed Prairie, Alpine Meadows, California Annual Grasslands, Palouse Prairie, Southern Mixed Prairie, Marshes, Wet Meadows, Tundra Grasslands, and Desert Grasslands
Shrublands	chaparral, cerrados, shrub-steppe, maquis, scrublands	Chaparral, Sagebrush-steppe, Salt-desert Shrublands, Tundra Shrublands, Mountain Browse, The Great Basin Shrublands
Woodlands/ Savannas	Typically called by the trees present	Pinyon-Juniper Woodland, Oak Woodlands, Aspen Savannas, Mesquite Woodlands
Deserts	---	Mojave, Sonoran, Chihuahuan, Great Basin Deserts

Adapted from the University of Idaho College of Natural Resources Rangeland Ecology and Management Program. http://www.cnr.uidaho.edu/what-is-range/Rangelands_Defined.htm

A comprehensive overview of definitions of different rangeland and grassland terms and their defining characteristics are listed in Appendix A: Land Type Definitions (Table 21). In addition, Table 22 provides further land use and land cover definitions from existing databases that may be utilized to provide baseline assessments for the protocol. These definitions may be useful to help distinguish the key

[2] See Fuhlendorf et al. (2002) for an explanation on the distinction between Shortgrass Prairies and Mixed Grass Prairies.

characteristics of these systems, which in turn will influence how to define the project activities that are acceptable within the protocol.

3.1.1 Regional Classification of Grassland and Rangeland Ecosystems

The Reserve requested consideration of several key rangeland systems. These specific systems are outlined below.

3.1.1.1 Grasslands of the Northern, Central, and Southern Great Plains

The "Great Plains" of the United States encompasses prairie, steppe, and grasslands west of the Mississippi and east of the Rocky Mountains. The Northern Great Plains is dominated by vegetation including grasses, sedges, and forbs with some shrubs (USFS, 2012). This region is characterized by several soil types[3] including Mollisols in the east and north and Alfisols and Aridisols in the south with Entisols scattered throughout the region.

Mollisols are highly fertile, rich in organic matter, and have the potential to provide significant emission reductions through carbon storage associated with AGC or CCG. Mollisols soils also have the highest potential for soil carbon sequestration associated with inorganic carbon (Goddard, et al., 2007). Alfisols, often found in glacial deposits along the Mississippi basin are also highly productive for agriculture, making them an important soil order to consider for AGC and CCG. In contrast, Aridisols are typically used for grazing but have little vegetation to add organic matter to soils (Ritter, 2012). Overall, the major soil orders located within this region provide the potential to reduction emissions through both AGC and through CCG.

The Great Plains is also a significant region for agricultural production. The fertility of Mollisols and Alfisols has made them attractive candidates for conversion to cropland. However, since Aridisols are found in arid and semiarid environments, they would most likely not be suitable for cropland unless irrigation was available.

The Great Plains is an important region for consideration in AGC and CCG project protocols for several reasons. Approximately 75% of the region is in private land ownership (Kansas State University, 2004). Furthermore, the Great Plains region, particularly the Northern Plains area, has seen significant land use change in recent years. Between 1997 and 2003, the area of grazing land (including pasture, range, and grazed forest land) in the US decreased by 1% (approximately 1 million acres a year were lost). This decline has been concentrated to a large degree in the Missouri River Basin (part of the Northern Plains), which has seen a decline of roughly 1.3 million acres of pasture and rangeland cover from 1992 to 2003 (Stubbs, 2007). Claassen et al. (2011b) found that approximately 770,000 acres of 1997 rangeland in the Northern Plains were converted to cropland by 2007 (roughly 1% of the rangeland area of the region). Compared with producers in other regions, the Northern Plains area (especially Kansas, Nebraska, South Dakota, and North Dakota) was more likely to convert rangeland to cropland. Indeed this region has the largest concentration of rangeland to cropland conversion, accounting for 57% of

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[3] The highest level categorization of soil types under the USDA classification scheme is by Soil Order. See more information about the twelve soil orders and their distribution in the United States at http://soils.usda.gov/technical/classification/orders/.

total conversion between 1997 and 2007 across the United States even though the region encompasses only 18% of the nation's rangeland (Claassen, et al., 2011b).

3.1.1.2 California Annual Grasslands

The grasslands of California are bounded by the Sierra Mountains and the California Coast Ranges running through the Central Valley. The Mediterranean climate of the region can be characterized by hot, dry summers and moist and mild winters. Though once dominated by perennial bunchgrasses, the region is now dominated by annual species.

Soils in the area are primarily Entisols and Alfisols with high fertility and good drainage. Silver et al. (2010) found that annual grasslands can sequester carbon at rates similar to perennial grasslands. Furthermore, the presence of woody plants increase carbon storage by an average of 59 metric tonnes of carbon dioxide equivalent [tCO_2e/ac] (40 MgC/ha), suggesting that wooded grasslands may provide even greater carbon sequestration potential. Given this combination of factors, California Annual Grasslands are a good candidate for AGC since they have been observed to be substantial carbon sinks.

This region is also known for its agriculture, and many annual grasslands in the Central Valley are interspersed with existing cropland. Irrigation is widespread within the Central Valley, providing the potential for cropland conversion. The cropland-dominated landscape also provides many opportunities for CCG within these regions. Since nearly 90% of California grasslands are privately owned (Jantz, et al., 2007) they provide a potentially large pool of participants.

One potential complication that may arise by including California Annual Grasslands is the presence of vernal pools. California annual grasslands contain some of the most known examples of vernal pools, ephemeral wetlands that arise seasonally. These systems provide many environmental benefits including habitat for species, many of which are often endangered (Pyke & Marty, 2005; Morgan & Calhoun, 2012). However, vernal pools behave like wetlands and release a significant amount of methane (CH_4). Globally, wetlands account for approximately 25% of all CH_4 emissions (Whalen, 2005). The sometimes sporadic and seasonal nature of vernal pools makes them difficult to track (Morgan & Calhoun, 2012). As of 2009 it was estimated that 893,000 acres of grassland habitat suitable for vernal pools existed in California (AECOM, 2009) out of a total of nearly 11 million acres of grassland in California (Jantz, et al., 2007). Since more than 8% of California grasslands may contain vernal pool habitat, greenhouse (GHG) emission calculations within these regions may be moderately more complicated.

3.1.1.3 Shrub and Bunchgrass Lands of the Great Basin

The Great Basin covers most of Nevada and parts of California, Oregon, Idaho and Utah. It is bounded by the Wasatch Mountains on the east and the Sierra Nevada and Cascade Mountains in the west. In lower elevations, the Great Basin is characterized by the dominant presence of sagebrush, perennial grasses and forbs (Utah State University, 2007). The semi-arid grasslands and shrublands of this region receive minimal rainfall but a high percentage of precipitation as snow (de Soyza, et al., 2000) and are dominated by Aridisols with lesser presence of Andisols and Mollisols (NRCS, 2012a).

The high presence of Aridisols indicates less potential for sequestration of soil organic carbon (SOC); however, arid and semi-arid environments do have the potential to accumulate significant amounts of inorganic carbon (Lal, 2002). However, there are land use pressures within this region that may make it relevant for inclusion within the AGC or CCG protocols.

While many parts of the region are not suitable for cropland, the four corners region in particular has seen an expansion of agricultural land in recent years (Grahame & Sisk, 2002). Though ecological threats including invasive species and fire are perceived major threats to Great Basin rural residents and stakeholders, development is also a notable perceived threat (Brunson & Shindler, 2008).

3.1.1.4 Pinyon-Juniper Woodlands of the West

Pinyon-Juniper Woodlands are spread throughout the West and Southwest. They are dominated by the presence of pinyon pine and juniper trees, but interspersed with annual and perennial grasses, grass-like plants, forbs, and shrubs (USFS and University of Arizona, 2002). In recent years, these trees have increased in abundance in many regions, leading to reduced grass dominance and land management practices to reduce tree cover are now being applied (USFS, 2005). Pinyon-Juniper Woodlands thus may provide some challenges when considering them in the context of CCG or AGC.

The presence and growth of woody plants in these rangeland systems can increase overall carbon storage across the landscape (Silver, et al., 2010). Woody expansion often leads to an increase in soil organic matter while management activities that reduce woody expansion through cutting or other methods can lead to rapid loss of soil carbon and nitrogen pools (Neff, et al., 2009). Since cutting juniper woodlands is a common practice within the region, it may be necessary to account for carbon losses associated with woody biomass removal within these systems. Producers may face a dilemma, however: without cutting they stand to lose significant productive rangeland; but with cutting they could be reducing the overall carbon storage potential of the system. The presence of natural fire exacerbated by woody expansion in addition to the use of burning in these systems for management can also complicate overall emissions calculations (Rau, et al., 2009).

3.2 How Potential Emission Reductions Are Estimated for this Paper

This Issue Paper presents estimates for carbon sequestration and avoided emissions based on a simplified Tier 2 accounting approach described by the IPCC for use in national GHG inventories (IPCC, 2006). This approach utilizes SOC reference values, defined based on soil type and climatic zone, which are then multiplied by factors to incorporate the effects of land use, inputs to the land, and management practices (i.e., tillage) on SOC storage. For each soil type (*ST*) and climate zone (*CZ*) SOC stocks in the top 30 cm of soil can be determined under each configuration of land use, management, and inputs as:

$$SOC_{CZ,ST,h} = REF_{CZ,ST} * F_{LU} * F_{TF} * F_{IF}$$

Where:

$SOC_{CZ,ST,h}$	SOC stocks at equilibrium for climate zone *CZ*, soil type *ST*, and land management system *h*; tCO_2e/ac

$REF_{CZ,ST}$	Reference SOC stocks at equilibrium for climate zone *CZ* and soil type *ST*; tCO_2e/ac
F_{LU}	Factor for land use type *LU* under land management system *h*; dimensionless
F_{TF}	Factor for tillage type *TF* under land management system *h*; dimensionless
F_{IF}	Factor for input level *IF* under land management system *h*; dimensionless

Soil organic carbon stocks are assumed to transition linearly from the equilibrium levels of the first land management system to another, achieving equilibrium under the new land management system after 30 years. The annual change in carbon stock using the IPCC method as applied in this Issue Paper can be represented by:

$$\Delta SOC_{h_{t=30},h_{t=0}} = \frac{SOC_{h_{t=30}} - SOC_{h_{t=0}}}{30}$$

Where:

$\Delta SOC_{h_{t=30},h_{t=0}}$	Annual change in SOC stocks produced for 30 years following a transition from current land management system $h_{t=0}$ to a new land management system $h_{t=30}$; $tCO_2e/ac/yr$
$SOC_{h_{t=30}}$	SOC stocks at equilibrium (i.e., after 30 years at time *t=30*) for new land management system *h*; tCO_2e/ac
$SOC_{h_{t=0}}$	SOC stocks at equilibrium for initial (i.e., at time *t=0*) land management system *h*; tCO_2e/ac

Ogle et al. (2003; 2006) provide SOC reference values and factors for the United States, which have been used within the US national GHG inventory. Although Ogle et al. (2003) presented SOC values for additional soil types (i.e., “Sandy,” “Volcanic,” “Spodisols,” and “Wetlands”), this Issue Paper only utilizes those values for “High-activity mineral” and “Low-activity mineral” soil types, as shown in Table 4:

Table 4: Soil organic carbon reference values used in this Issue Paper, tCO_2e/ac

	Cool Temperate		Warm Temperate		Sub-tropical	
Soil Type	Dry	Moist	Dry	Moist	Dry	Moist
High-activity mineral	60.2 - 64.4	94.8 - 98.1	53.3 - 56.5	74.2 - 77.2	58.5 - 66.2	65.3 - 103.9
Low-activity mineral	62.3 - 71.2	73.7 - 80.6	35.0 - 39.2	57.6 - 61.1	50.7 - 65.0	49.1 - 90.4

Notes: Adapted from Ogle et al. (2003). *These values were used as input for the variable* $REF_{CZ,ST}$.

Regional land use and tillage factors were drawn from Ogle et al. (2006) to estimate SOC stocks for each land management system. The range of values considered for this Issue Paper is shown in Table 5:

Table 5: Land use and tillage factors used in this Issue Paper

	Cool Temperate		Warm Temperate		Sub-tropical	
Soil Type	Dry	Moist	Dry	Moist	Dry	Moist
Land Use						
Cultivated	1.0	1.0	1.0	1.0	1.0	1.0
Tillage						
Conventional	1.0	1.0	1.0	1.0	1.0	1.0
Reduced-till	0.98 - 1.04	1.05 - 1.11	0.98 - 1.04	1.05 - 1.11	0.98 - 1.04	1.05 - 1.11
No till	1.02 - 1.08	1.11 - 1.15	1.02 - 1.08	1.11 - 1.15	1.02 - 1.08	1.11 - 1.15

Notes: Adapted from Ogle et al. (2006). *The values above are unitless. For the calculation of technical emission reduction potential, these values were used as inputs for the variables* F_{LU} *and* F_{TF}. *Ogle et al.* (2006) *found no statistically significant factor for input levels at a regional level, and thus* F_{IF} *was held constant at 1.0 for the calculations made in this Issue Paper.*

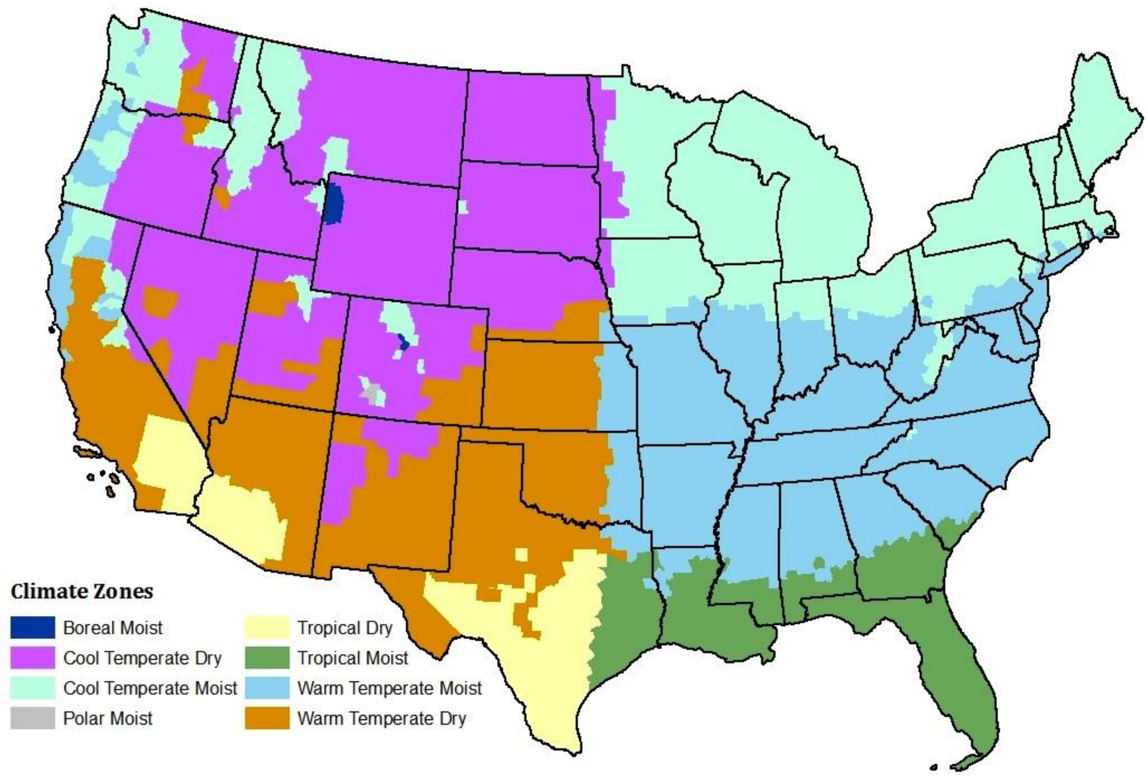
The range of emission factors corresponding to transition from cropland to set aside (as in CCG project activities) was then calculated for each climate zone, using the high and low end for each factor and SOC reference level. For AGC project activities, the high and low end values for each factor were used to estimate the emissions associated with a transition from Uncultivated to Cultivated land use types in each climate zone. The range of emissions factors for AGC and CCG project activities in each climate zone are shown provided in the discussion of the cost of these potential reductions below, in Table 12 and Table 17, respectively.

The potential for emission reductions was then calculated by determining the land area in each climate zone that could potentially be subject to AGC or CCG project activities. The distribution of IPCC climate zones for the continental United States is shown in Figure 5.

The potential land area for AGC project activities was estimated from the gross annual conversion rate observed in NLCD data between 2001 and 2006 for grassland and shrubland land covers to cropland. These data, related datasets available, and additional considerations are discussed further in Section 4.2.

The potential land area for CCG project activities was estimated from the cultivated cropland area in each county with land capability classes or slopes above 30%, indicating moderate to severe limitations for crop production (land capability classes 5 and above). The determination of "degraded" lands status was derived from the Digital General Soil Map (DGSM) of the United States, formerly known as STATSGO(NRCS, 2006) and spatially overlaid with NLCD data. The spatial intersection of cultivated cropland in the NLCD dataset with land capability classes and high-slope areas from the DGSM was calculated to determine the area in each county for potential CCG activities. See Section 6.3.1.2 for a definition of the eight land capability classes.

Figure 5: IPCC climate zones for the continental US



4 Avoided Grassland Conversion (AGC) Characterized

4.1 Scope and Definition of Project Activities

In its Request for Proposals (RFP) for this Issue Paper, the Reserve proposed a project activity for AGC defined as:

> *The conversion of grassland into production cropland or other land development can rapidly decrease soil carbon stocks as a result of soil disturbance and removal of permanent vegetation. By permanently conserving grassland that otherwise would have been converted into alternative use, substantial emissions of carbon may be avoided.*

The following discussion builds from this definition to cover the science and policy considerations regarding this potential project type.

4.1.1 Project Duration and Crediting Periods

In general, there are two important qualities of AGC project activities which support a project duration and/or crediting period on the scale of 20-30 years. First, AGC projects would be carried out in the context of competing agricultural land management decisions. For several practical and generally self-apparent reasons, agricultural land management is applied over a timeframe that is generally shorter than the forestry sector (for which the Reserve has approved project durations up to 100 years). Second, the primary source of emission reductions for AGC projects is expected to be the protection of

SOC stocks. Under the prevailing IPCC approach for SOC accounting, soils are generally assumed to achieve equilibrium SOC stocks after twenty years under a particular land management regime. Thus, the emission reduction benefits achieved by AGC project activities should generally be complete within this timeframe. However, more recent research supports a timeframe of thirty years or greater for certain lands to achieve equilibrium. The estimated emission factors in this paper assume that it takes thirty years to move from one equilibrium to another.

The length of crediting periods may also be needed to address the potential for piecemeal conversion of Project Area grasslands in the baseline scenario, which may also be affected by the choice of accounting approaches to estimate the baseline conversion extent.

4.2 Land Use Trends

4.2.1 Data Availability for Estimating Grassland Conversion Rates

There are three primary datasets with the potential to estimate grassland conversion rates: National Agricultural Statistics Service (NASS); National Resources Inventory (NRI); and National Land Cover Database (NLCD). Before reviewing each dataset, it is important to understand the dimensions over which the datasets vary and how those differences can affect estimates of grassland conversion rates. The important dimensions include:

- *Scale*– scale refers to the physical size of the smallest unit of observation (i.e., the resolution). Typical scales in land use data are aggregate (e.g., county-level, state-level), plot-level (by land use activity) and parcel-level (by ownership boundary). The scale of data can greatly affect estimates of grassland conversion rates. For example, aggregate data can show general trends over time for broad geographical areas, but cannot be used to accurately assess land conversion rates because actual conversions are not observed.
- *Frequency* – frequency refers to how often land use observations are available. Typical datasets either provide land use observation annually or at five-year intervals. Shorter time between observations implies more accurate land conversion rate estimates. With less frequent observations, multiple land conversions can occur between observations adding noise to conversion estimates. Moreover, if conversions that affect critical measures (e.g., SOC) occur between observations, their effect may go unmeasured.
- *Land Use Classes* – land use classes refer to the definitions of alternative land covers used for different datasets. Each of the three datasets uses slightly different definitions. Differences in definitions pose several problems for estimating grassland conversion rates such as constraining conversion estimates to the land class definition, and misrepresenting true conversion rates due to broad variability of land characteristics under a single land class. See Table 22 in Appendix A: Land Type Definitions for more details.

With these dimensions in mind, we briefly review the primary datasets available for estimating grassland conversion rates.

4.2.1.1 National Agricultural Statistics Service (NASS)

The USDA NASS produces a variety of datasets on private agricultural land use, including the Census of Agriculture and state-level surveys. All NASS land use data are aggregated to the county- or state-level. The Census of Agriculture is conducted every five years (latest observation in 2007), while the state-level surveys are conducted annually (latest observation 2011). NASS focuses on field crops and does not explicitly include a grassland class. Given the scale of, and land use classes in, the NASS data, it is sub-optimal for estimating grassland conversion rates.

4.2.1.2 National Resources Inventory (NRI)

The NRI is a longitudinal dataset (i.e., same plots observed over time) of land use/land cover for the contiguous US that makes state-level aggregate data available for public use. NRI data are collected on five-year intervals, beginning in 1982, with the latest data available for 2007. The data are arranged into specific-use (e.g., corn) and broad-use (e.g., pastureland) categories for all non-federal land.

The NRI data are collected using a sophisticated stratified sampling routine, where the intensity of sampling in a region is positively related to the heterogeneity on the landscape. Because the margin of error is hundreds of thousands of acres depending on land use type caution should be exercised when interpreting land use change estimates from the NRI data.

4.2.1.3 National Land Cover Database (NLCD)

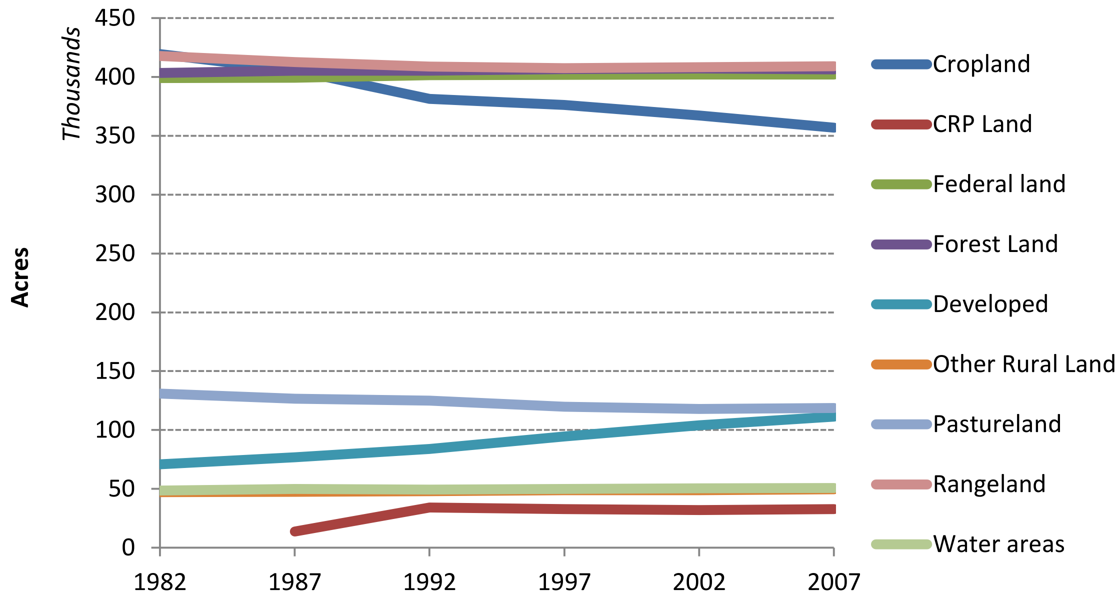
The NLCD is a Landsat-based plot-level database with contiguous US coverage is available in the NLCD data for 2001, 2006 (2011 data are forthcoming). The NLCD has 16 land cover classes, capturing all possible covers from high intensity development to perennial ice and snow. The NLCD also contains managed agricultural classes (“cultivated crops” and “pasture/hay”) and “developed land” classes, such that grassland conversions can be explicitly identified. Since the NLCD data are generated from satellite imagery, it does suffer from misclassification error. Therefore, like the NRI data, land cover observations are only estimates within a margin of error. Accuracy assessments indicate that the NLCD data has an overall accuracy (Andersen Level I) around 80%. Accuracy, however, tends to be lower at finer scales (e.g., when examining a specific plots vs. state-level aggregates), and is highly variable across land classes (e.g., some land classes are identified more accurately than others).

Given the datasets described above, we focus on the NRI and NLCD data for this issue paper. Both are readily available, are reasonably easy to manipulate, and can be used to measure grassland and cropland land-use change.

4.2.2 Grassland Conversion Trends

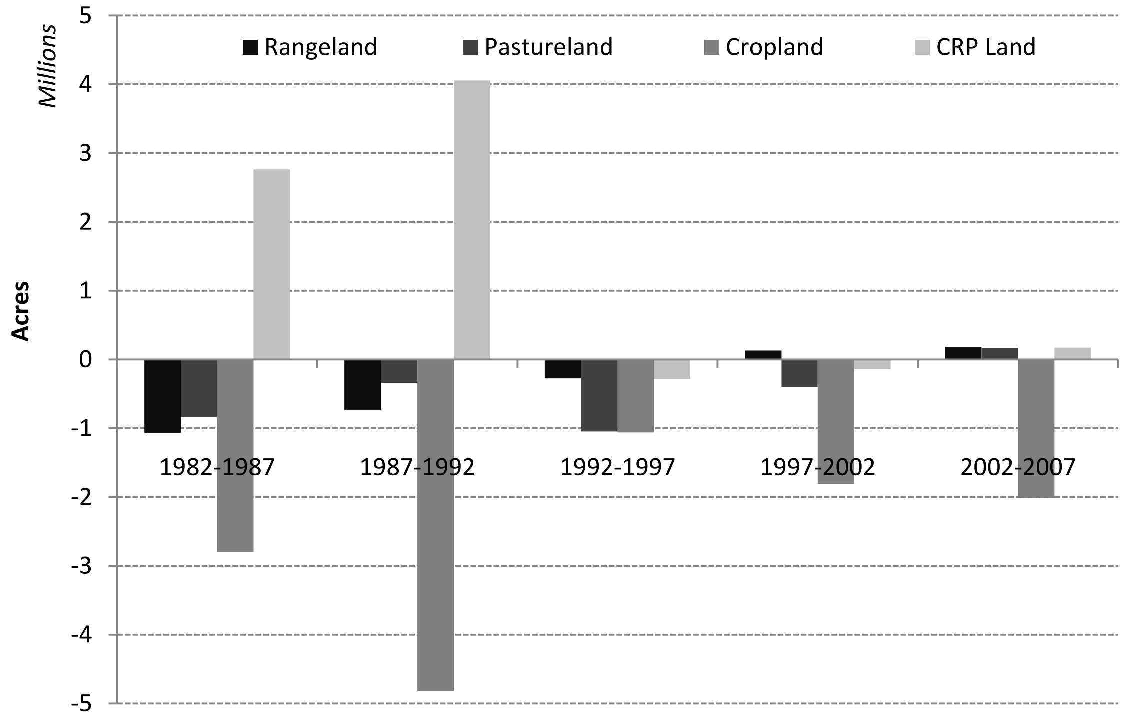
The NRI state-level data provides the longest time series for exploring land use trends. From 1982 to 2007 many land uses at the national-level remained relatively constant (Figure 6; Table 6). The exceptions are: “cropland” (lost 62 million acres); “pastureland” (lost 12.3 million acres); “rangeland” (lost 9 million acres); “CRP” (gained 32 million acres); and “developed land” (gained 40 million acres). Thus, nationwide over this time period, “developed land” appears to be growing at the expense of “cropland,” “pastureland,” and “rangeland.” With respect to grassland (best captured by the “rangeland” and “pastureland” classes), significant acreage was lost; however, the loss was more than offset by gains in “CRP.”

Figure 6: National-level land use trends from NRI dataset



The NRI data, though indicating important broad changes from 1982-2007, also indicate that land use trends have not been constant over time (Figure 7). "Rangeland" and "pastureland" decreased substantially during the early NRI survey periods (loss of 21.5 million acres, 1982-1997). During more recent survey years, however, "rangeland" and "pastureland" acres show modest increases. The effect of the first sign-up of the Conservation Reserve Program (CRP) in 1985 is also evident; however, acreage begins exiting "CRP" during the 1992-1997 period. Finally, the NRI data over time indicates a consistent long-term decline in "cropland" acres, which appears to be largely explained by CRP enrollments during the early periods but not during the later periods. Thus, more recent decreases in "cropland" are not explained by increases in other agricultural land classes, and therefore must be explained by increases in non-agricultural land classes (i.e., "developed," "forest," and "federal land").

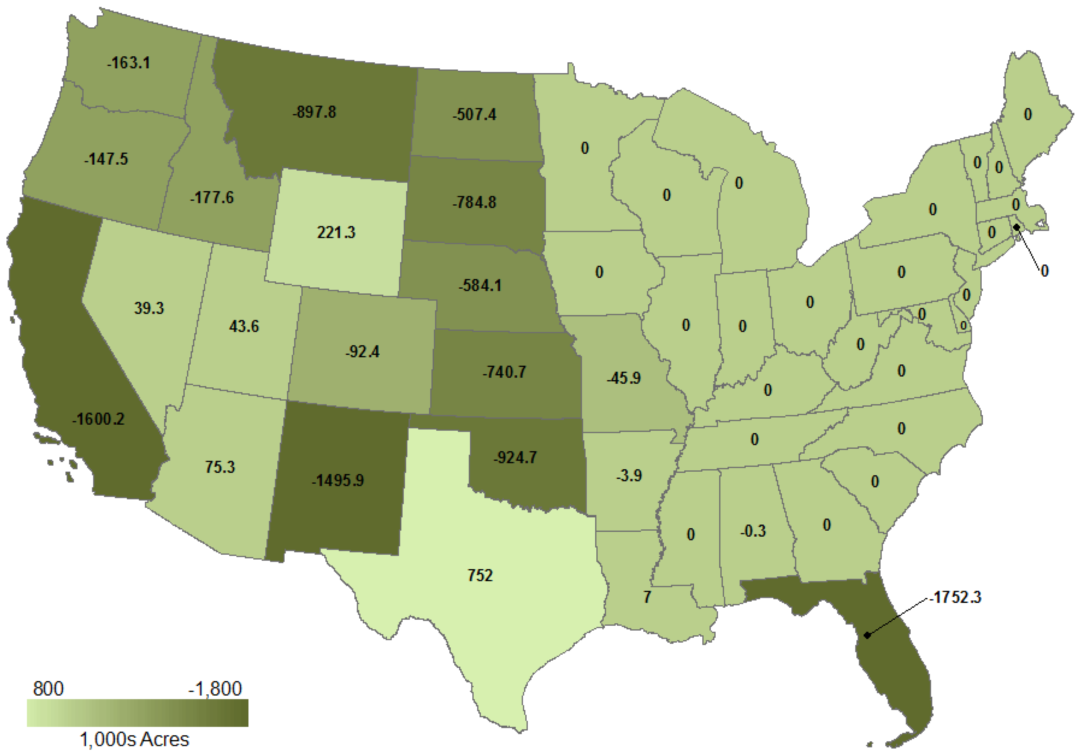
Figure 7: NRI acreage change over time



Changes in "rangeland" (most representative of native grassland) are also heterogeneously distributed across space, with some regions of the country gaining acreage while other regions lose acreage[4] (Figure 8). In general, grassland acreage in the Northeast, Mid-Atlantic, Corn Belt and Northwest remained relatively constant or increased from 1982 to 2007. In contrast, the northern Great Plains, along with Florida, New Mexico and California, experiences net losses in grassland acreage. Given the distributions of regions where grassland decreased, it is clear that the end-point of converted grassland likely differs across space. California, for example, experienced substantial increases in "developed land" (2.1 million acres) during the same time period that grassland and "cropland" decreased (by 1.6 million and 0.9 million acres, respectively). Florida shows a similar pattern to California; however, much (roughly two-thirds) of the grassland loss in New Mexico appears to be related to increases in "CRP" and "federal land" (i.e., may not actually represent grassland loss). In contrast, the Plains states have conversion trends that are much more difficult to discern. States such Nebraska, North Dakota, South Dakota, and Montana experienced significant decreases in "rangeland," "pastureland" and "cropland," with the only significant acreage increases occurring in the "CRP" class. Because of the aggregate nature of the data, it is impossible discern the actual conversion pathways. We therefore cannot, for example, determine if any "cropland" acreage that was converted to "CRP" was replaced by converting some grassland to cropland (i.e., slippage).

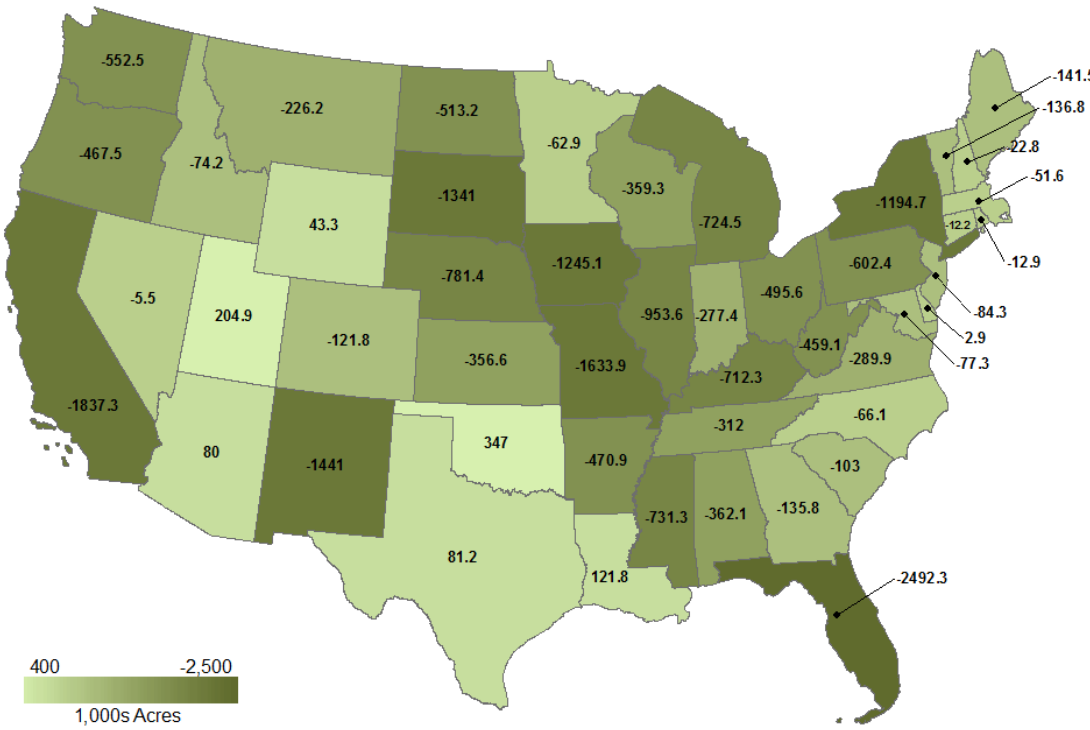
[4] We focus on absolute acreage changes rather than percent changes because acreage changes can identify policy-relevant "hotspots" (regions with significant change), and because absolute changes in acreage can potentially be converted to changes in carbon pools.

Figure 8: Change in NRI rangeland (i.e., native grassland) acres by state, 1982-2007



The spatial distribution of land-use trends, however, looks very different if we consider the NRI's "pastureland" category as grassland. NRI "pastureland" can be managed (e.g., introduced forage that is fertilized/replanted), but may largely represent un-plowed non-native grass. If "pastureland" is counted as grassland (i.e., grassland = "rangeland" + "pastureland"), then we see significantly more dispersed long-term declines in grassland (Figure 9). In particular, the Northeast and Corn Belt (which have no "rangeland") show greater losses in grassland, which is consistent with the growth in development and with the intensive cropland expansion of the 1980s and 1990s.

Figure 9: Change in NRI rangeland and pastureland acres by state, 1982-2007



Although the publicly available NRI data does not allow users to estimate explicit changes between land cover/use categories, NRI summary reports do provide a national-level land change matrix (Table 6). At the national-level, the NRI data from 1982 to 2007 indicates that most of the converted pastureland converted to cropland or forestland. Converted rangeland acres had a much more diverse set of endpoints, with "cropland," "developed land," "pasture" and "forest land" all-consuming significant "rangeland" acreage. National-level changes, however, do not allow us to isolate hot spots for specific grassland conversions.

Table 6: NRI state-level land use/cover change matrix, 1,000s acres

	Land cover/use, 2007							
Land cover/use, 1982	**Cropland**	**CRP**	**Pasture**	**Rangeland**	**Forest Land**	**Other Rural Land**	**Developed Land**	**Water Areas & Federal Land**
Cropland	326,196.4	30,168.6	30,344.7	6,895.4	8,922.7	4,136.4	11,117.5	1,765.2
CRP	-	-	-	-	-	-	-	-
Pastureland	18,526.6	1,351.6	78,372.2	5,085.3	17,760.5	2,036.1	6,845.0	919.0
Rangeland	7,430.8	1,124.5	3,369.1	391,615.0	3,379.4	2,272.5	5,201.0	3,507.2
Forest Land	2,121.7	144.4	4,847.6	2,175.6	371,660.4	2,229.1	17,083.5	3,117.3
Other Rural Land	1,685.2	56.4	1,159.0	915.5	3,310.2	38,734.9	1,077.8	304.1

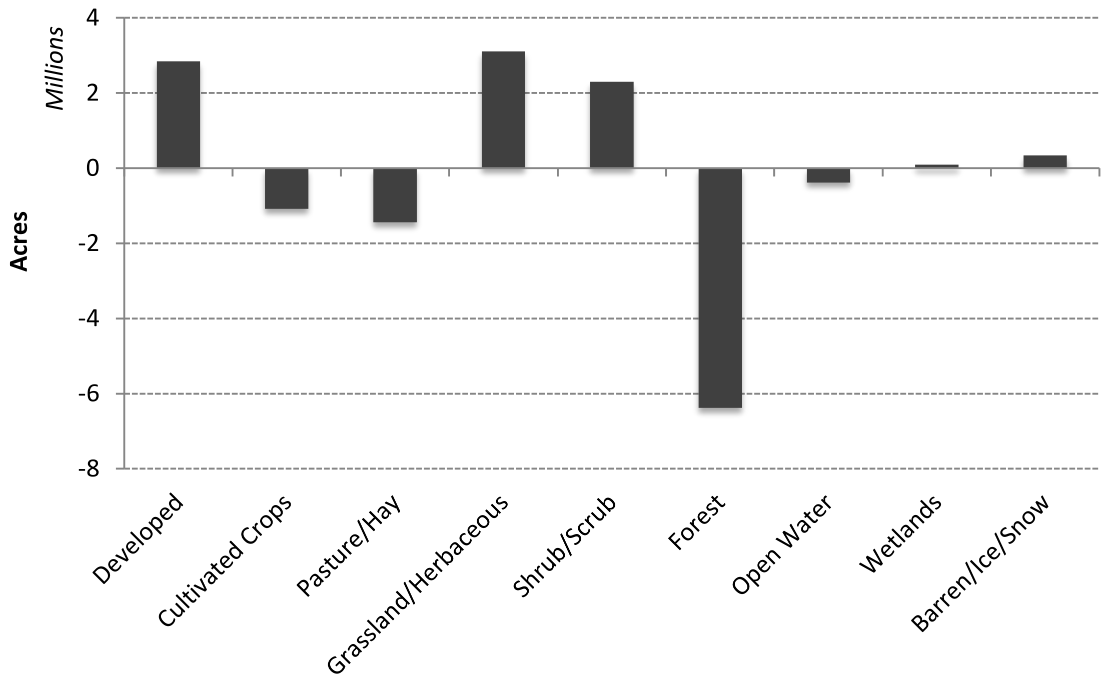
Land cover/use, 1982	Land cover/use, 2007							
	Cropland	**CRP**	**Pasture**	**Rangeland**	**Forest Land**	**Other Rural Land**	**Developed Land**	**Water Areas & Federal Land**
Developed Land	264.1	0.0	163.7	176.6	442.6	18.4	69,896.9	1.8
Water Areas & Federal Land	798.7	32,850.2	118,615.7	409,119.4	406,410.4	49,639.6	111,251.2	452,754.2

It is much easier to discern explicit land use changes in the NLCD database, which tracks individual plots over time. We cannot, however, display the plot-level data because of computational limitations (there are approximately 8.9 billion plots in the NLCD database). We therefore calculate a plot-level change matrix, which calculates the acreage change between all land classes from 2006 to 2011, and then we aggregate the results to the county-level. Recall that "grassland" in the NLCD data is best captured with the "grassland/herbaceous," and "shrub/scrub" land classes.

From 2001 to 2006 grassland and shrubland in the NLCD data increased nationwide by 5.4 million acres (Figure 10)[5]. "Developed land" also increased substantially (2.8 million acres). These increases were largely achieved by decreases in "forest land" (6.3 million acres), "pasture and hayland" (1.4 million acres) and "cropland" (1 million acres).

[5] Values in Figure 10 are net changes; thus, they account for all land that converted from and converted to each land class.

Figure 10: Net change by NLCD land cover class, 2001-2006



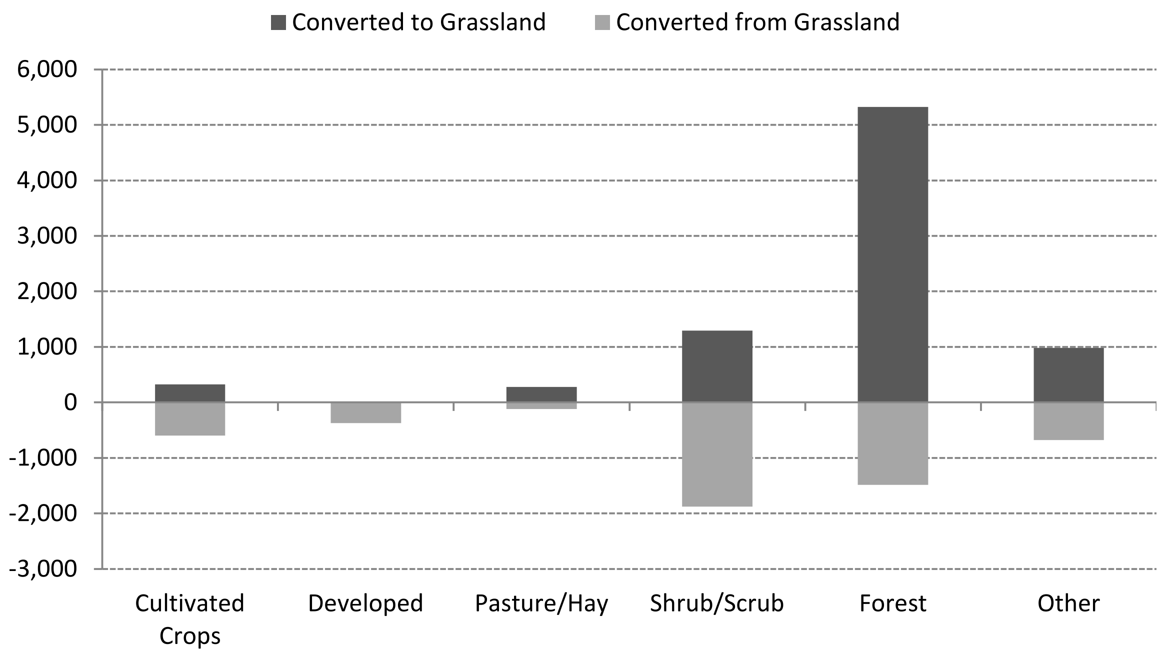
The aggregated trends depicted in Figure 10 suggest a somewhat smooth transition between major land classes. Observations at the parcel-level, however, reveal a much more dynamic land-use environment (Table 7). Over the five-year period, most land remains in its initial use, as expected. The remaining changes in land area by class are not, however, the result of simple one-to-one transitions. Instead, the net changes result from converting back-and-forth between multiple land classes. The three million acre increase in grassland, for example, results from significant net-positive conversions of "forest," "shrub/scrub," and "cropland" to grassland (Figure 11). This does not imply, however, that no grassland was lost. To the contrary, significant grassland acreage converted to "cropland" (598,000 acres), "developed land" (373,000 acres), "pasture/hay" (120,000 acres), "forest" (1.5 million acres), and to "other" uses (677,000 acres – primarily wetlands).

Table 7: Conversion matrix for major land classes from NLCD data (2001-2006), 1,000s acres

		Converted to:						
		Cult. Crops	Developed	Grass/ Herb	Pasture/ Hay	Shrub/ Scrub	Forest	Other
Converted from:	Cult. Crops	307,877	698	323	19	370	284	410
	Developed	0	106,151	2	0	1	1	1
	Grass/Herb	598	373	280,669	120	1,878	1,488	677
	Pasture/Hay	13	465	281	133,615	410	605	344
	Shrub/Scrub	308	318	1,292	203	418,139	2,946	782
	Forest	480	756	5,324	186	4,968	492,410	507

	Converted to:						
	Cult. Crops	Developed	Grass/ Herb	Pasture/ Hay	Shrub/ Scrub	Forest	Other
Other	446	289	984	161	665	410	123,910

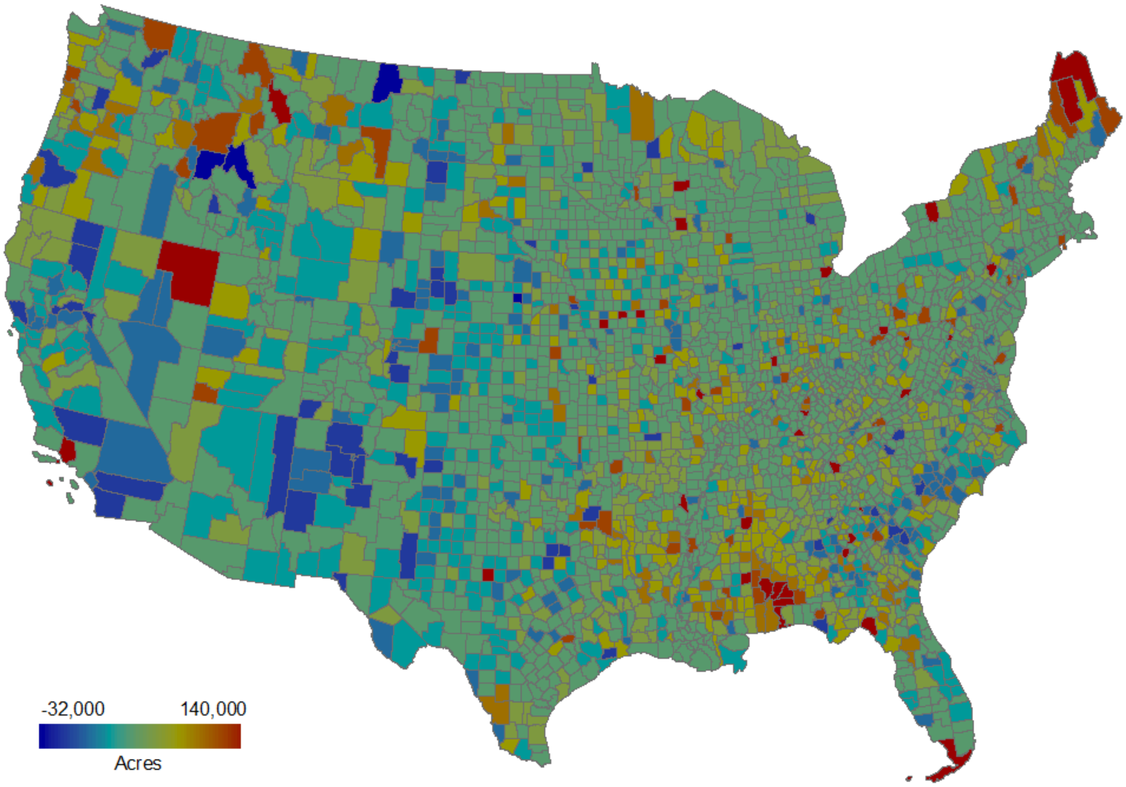
Figure 11: Grassland conversion by NLCD major land classes, 1,000s acres



Grassland and shrubland captured in the NLCD database follows a less clear spatial pattern than the NRI data (Figure 12 and Figure 13)[6]. There has, similarly, been relatively little conversion (i.e., no net change) in the eastern US, with the exception of some hot spots in the Southeast and scattered counties throughout the Midwest. Grassland loss occurred broadly along the western edge of the Great Plains and Southwest. Some, but not nearly all, of the grassland losses appear to be shifts between “grassland” and “shrub/scrub” (i.e., not really grassland loss). These “transitions” may be explained, in part, by misclassification errors from the satellite imagery (e.g., in some of the desert Southwest counties). The remaining spatial patterns of “grassland” and “shrub/scrub” losses align coarsely with the history of land development (both residential and cropland development). Thus, much of the current “action” appears to be occurring on the extensive margin around current areas of cropland and development expansion.

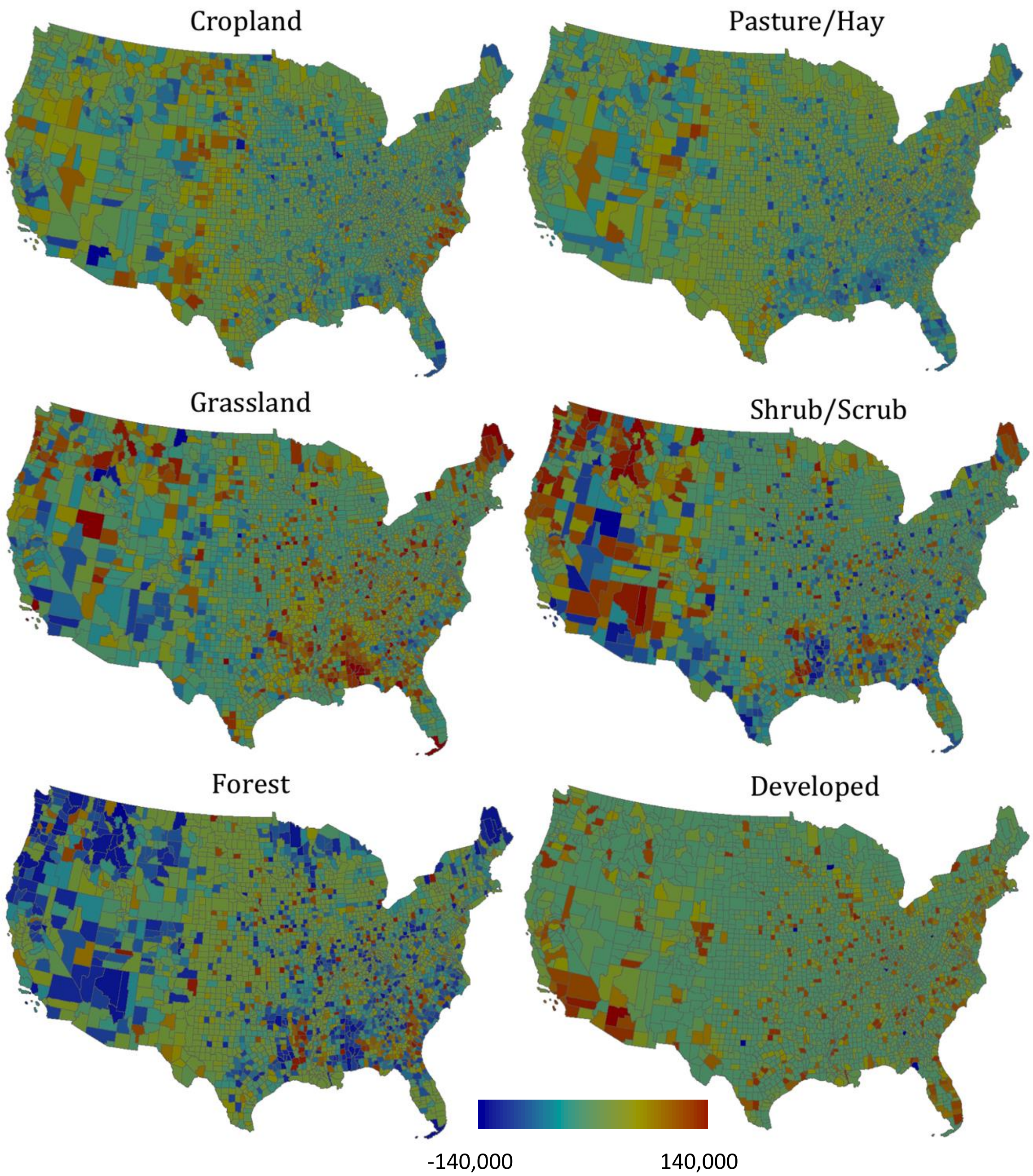
[6] As noted in footnote 4 (pg. 15), the maps in this Issue Paper present absolute acreage changes, which we believe to be more informative than proportional (i.e., percent-based) changes in land use types. Figure 12 presents the acreage change by NLCD land cover between 2001 and 2006. The county-level acreage under each NLCD land cover and the percent change from 2001 to 2006 has been provided in a separate spreadsheet to the Reserve.

Figure 12: Net change in NLCD grassland class by county, 2001-2006



Conclusions about general land use trends are difficult to draw from the NLCD data. These difficulties arise because of the highly dynamic nature of land use change across the country. Thus, understanding the spatial patterns of land use requires considering many simultaneous conversions to and from multiple land classes, and considering the unique characteristics and drivers across different regions (Figure 13).

Figure 13: Net change in major NLCD land classes, 2001-2006, acres



4.2.2.1 Conversion Rate Estimates

We estimate annual grassland conversion rates from the NRI using a simple rate of change calculation:

$$Conversion\ Rate = \frac{Acres_t - Acres_{t-k}}{Acres_t} \times \frac{1}{k}$$

where:

$Conversion\ Rate$	Annual grassland conversion rate; % per year
$Acres_t$	Area of grassland in year *t*; acres
$Acres_t$	Area of grassland in year *t* minus *k*; acres
k	Time since previous area measurement; years

Conversion rates calculated from the NRI data indicate the general decreasing trend in grassland, with a much higher rate of conversion for pasture (-0.39%/year) than rangeland (-0.08%/year) (Table 8). On the opposite side of the ledger, the NRI conversion rates show the consistent growth in CRP land (7.16%/year) and developed land (1.89%/year) that was observed over the time period (1982-2007). Because of the aggregate characteristic of the NRI data, however, we cannot derive land class specific conversion rates (e.g., rate of conversion from grassland to cropland).

Table 8: Annual conversion rates by transition period calculated from NRI data, %/yr

	Transition Period					
Land Cover/Use	**82-87**	**87-92**	**92-97**	**97-02**	**02-07**	**Average 82-07**
Cropland	-0.67%	-1.19%	-0.28%	-0.48%	-0.55%	-0.63%
CRP Land		29.36%	-0.82%	-0.43%	0.54%	7.16%
Federal land	0.02%	0.11%	0.01%	0.01%	0.00%	0.03%
Forest Land	0.10%	0.00%	0.06%	0.03%	-0.04%	0.03%
Developed	1.66%	1.83%	2.55%	2.00%	1.39%	1.89%
Other Rural Land	0.13%	0.22%	0.28%	0.08%	0.29%	0.20%
Pastureland	-0.64%	-0.27%	-0.84%	-0.33%	0.14%	-0.39%
Rangeland	-0.25%	-0.18%	-0.07%	0.03%	0.04%	-0.08%
Water areas	0.48%	-0.17%	0.20%	0.21%	0.16%	0.18%

We use the NLCD data to derive land class-specific rates of change between land class categories. Since we observe actual transitions between land plots, we can explicitly estimate the gross and net conversion rates in acres/year. The gross and net annual conversion rates between land classes *i* and *j* are calculated as:

$$Gross\ Conversion\ Rate_{ij} = Acres_{ij,t-k} \times \frac{1}{(t-k)}$$

$$Net\ Conversion\ Rate_{ij} = \left(Acres_{ij,t-k} - Acres_{ji,t-k}\right) \times \frac{1}{(t-k)}$$

where:

$Gross\ Conversion\ Rate_{ij}$	gross conversion rate from land class *i* to land class *j*; % per year
$Acres_{ij,t-k}$	area of land converted from land class *i* to land class *j* between year *t* minus *k* and year *t*; acres
$Net\ Conversion\ Rate_{ij}$	net conversion rate between land class *i* and land class *j*; % per year

$Acres_{ji,t-k}$ area of land converted from land class *j* to land class *I* between year *t* minus *k* and year *t*; acres

As depicted in the figures above, there is significant variability in grassland conversion rates (Table 9). The average gross conversion rate implies that counties lost, on average, 58 acres of grassland to cropland annually between 2001 and 2006. Some counties, however, lost grassland at rates over 2,000 acres per year. The net conversion rate implies, as expected from the trends described above, that many counties are gaining grassland from cropland at greater rates than they are losing it (see Table 23 and Table 24 in Appendix B: Conversion Rate and Land Value Tables, for state-level total conversion rates).

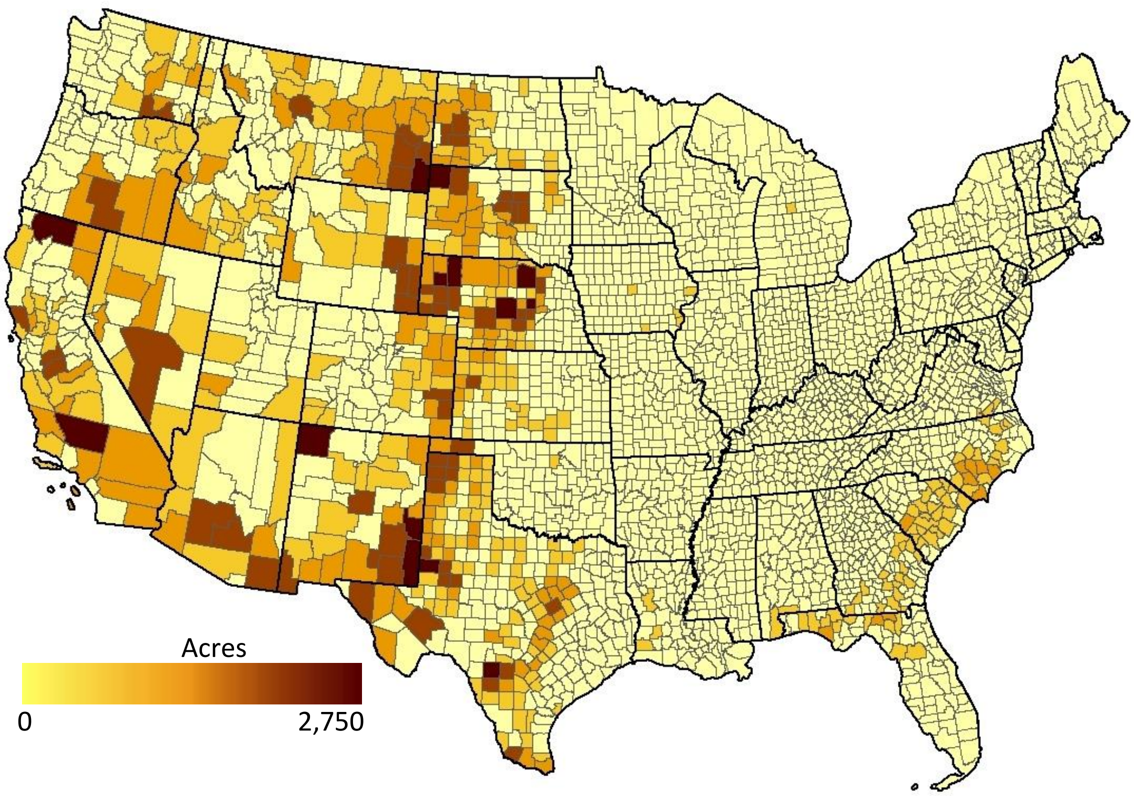
Table 9: Characteristics of county-level NLCD grassland to cropland conversion rates, ac/yr

	Avg	Max	Min	St.Dev.
Gross Rate	58.26	2,748.21	0	183.32
Net Rate	13.65	2,735.67	-2,414.54	207.31

Note: Grassland conversion includes grassland + shrub/scrub classes

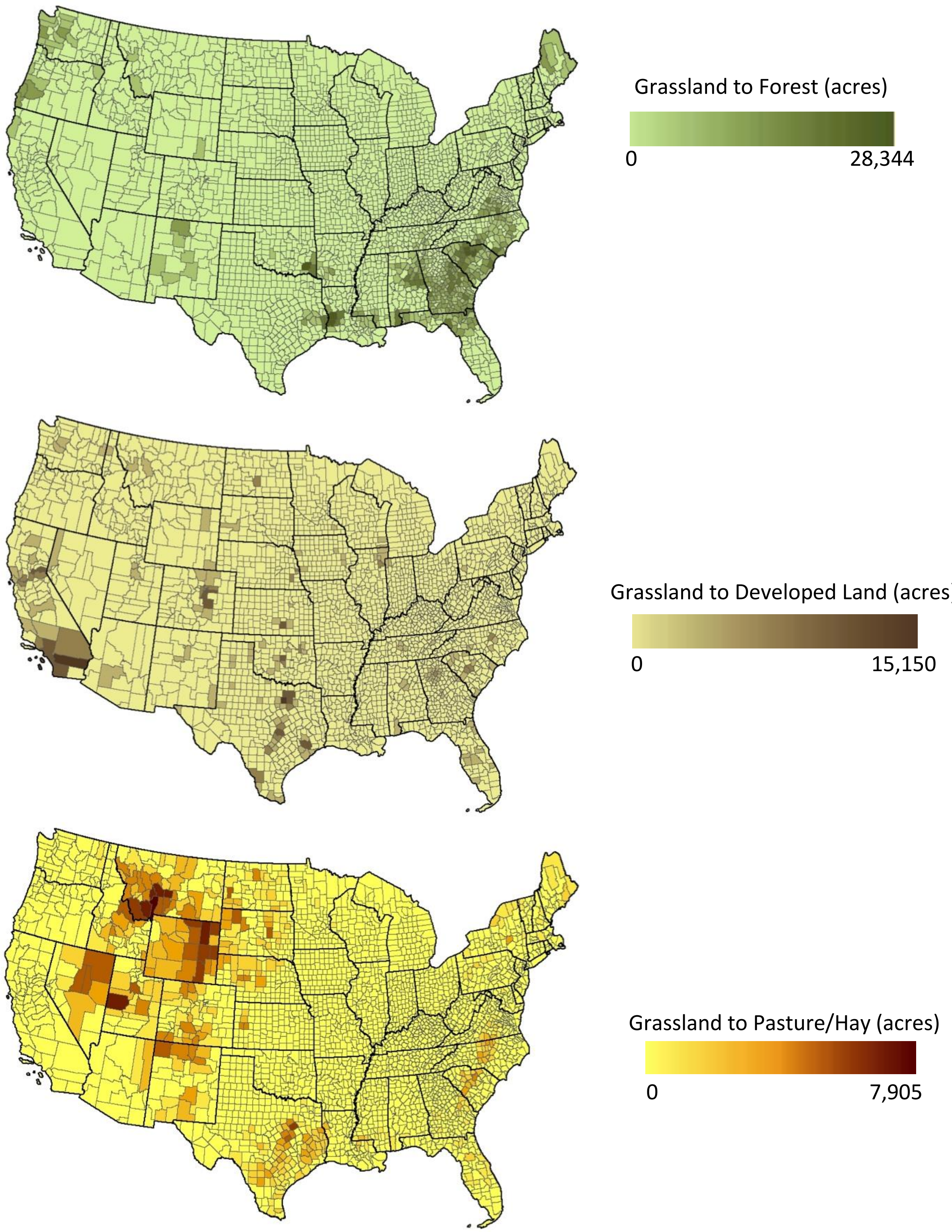
"Grassland" to "cropland" conversion rates are generally highest across the western US grassland regions (Figure 14). The high conversion rates include scattered "hot-spots" along the western boundary of the northern Great Plains, in the Southwest, and in the Pacific Northwest (east of the Cascade Mountains). There is also an additional "warm-spot" of relatively high "grassland" to "cropland" conversion rates stretching along the southeastern coast from North Carolina to the Florida panhandle.

Figure 14: Gross conversion rates for grassland and shrub/scrub to cropland



“Grassland” conversion rates to other uses, particularly “forestland” and “developed land,” are often much higher than the rates of conversion to “cropland.” “Grassland” conversions to these classes, however, are much more spatially concentrated. High “grassland” to “forest” conversion rates are concentrated in the Southeast and Pacific Northwest, while high “grassland” to “developed land” rates are highly concentrated in southern and central California, the Front Range of Colorado, and around the major metropolitan areas in Texas (Figure 15). Significant grassland acreage also converted to the “pasture/hay” land class, with these conversions concentrated in the greater Rocky Mountain region.

Figure 15: Gross conversion rates from grassland to forest, development, and pasture/hay



4.2.2.2 Regional Grassland Conversion Rates

We also aggregate grassland conversion rates by IPCC climate zones to generate a broader perspective on conversion trends. Here we focus on the total annual grassland to cropland gross conversion rates. The total conversion rate is calculated by summing the county-level conversion rates to generate the total acres/year converted (Table 10).

Table 10: Gross conversion rates for grassland and shrub/scrub to cropland

IPCC Climate Zone	**Grassland to Cropland** (ac/yr)	**Shrub/Scrub to Cropland** (ac/yr)
Warm Temperate Moist	6,335.84	7,253.04
Warm Temperate Dry	33,211.68	22,158.77
Cool Temperate Moist	4,406.75	2,298.00
Cool Temperate Dry	64,544.62	12,991.62
Polar Moist	0.00	0.00
Boreal Moist	37.50	28.91
Tropical Moist	3,653.76	5,064.15
Tropical Dry	7,402.40	11,754.62

4.2.3 USDA CRP Participation Rates and Related Programs

As of October 2011, 29.6 million acres of cropland were enrolled in the CRP. Program enrollment grew rapidly after the program's inception in 1986, and generally continued to increase nationwide through 2007 (Figure 16). Beginning in 2008, however, cumulative CRP enrollment began declining and has decreased an average of 4%/year since 2007 for a total decrease of approximately 5.64 million acres. The timing of the recent decreases in CRP acreage coincides with the recent rise in crop prices. Though we lack data to determine causality between the changes in CRP and crop prices, previous research suggests that rising crop prices have impacted CRP enrollment (Hellerstein & Malcolm, 2011). Decreases in absolute acreage have been largely concentrated in the Corn Belt and Northern Great Plains (Figure 17). These are also regions where the NLCD data indicates substantial grassland and shrub/scrub conversion to cropland.

Figure 16: Enrollment in the Cropland Reserve Program (CRP), 1986-2011

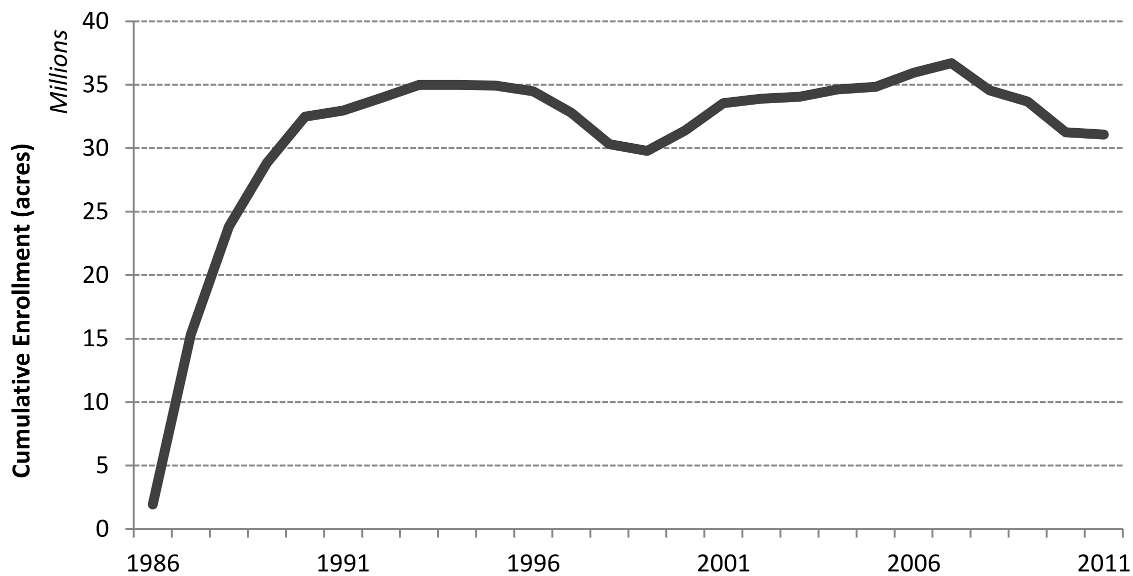
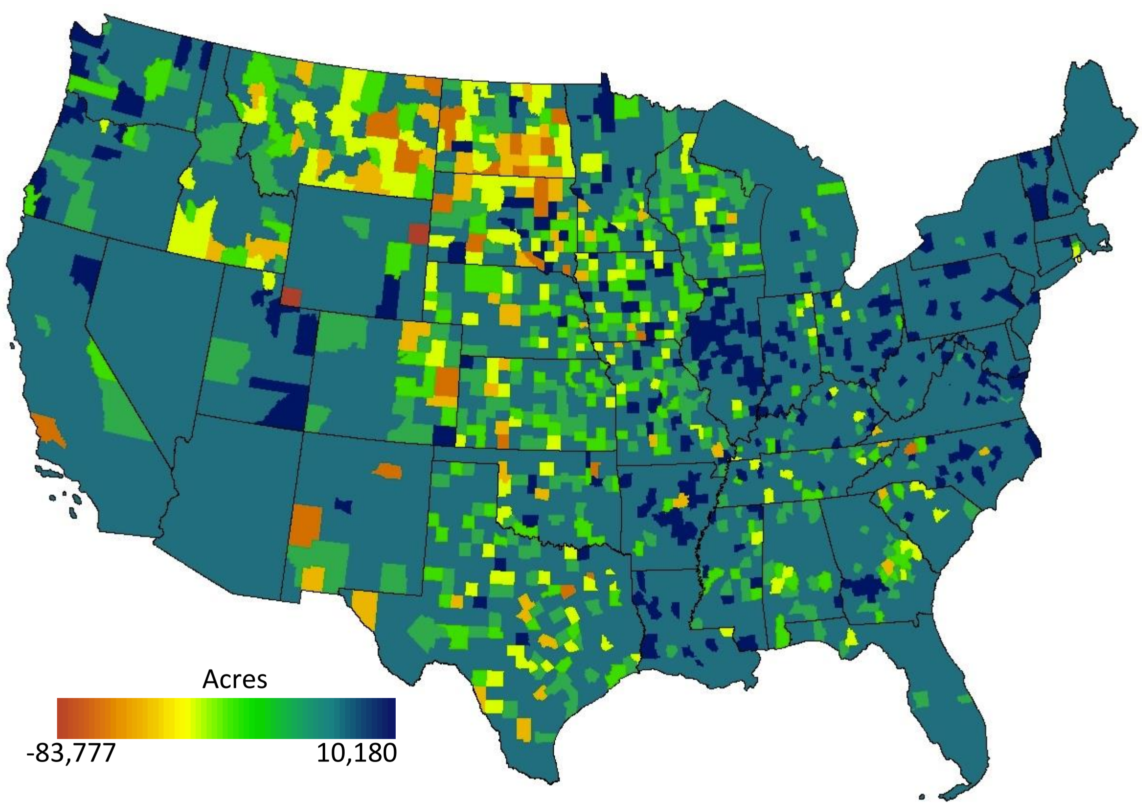
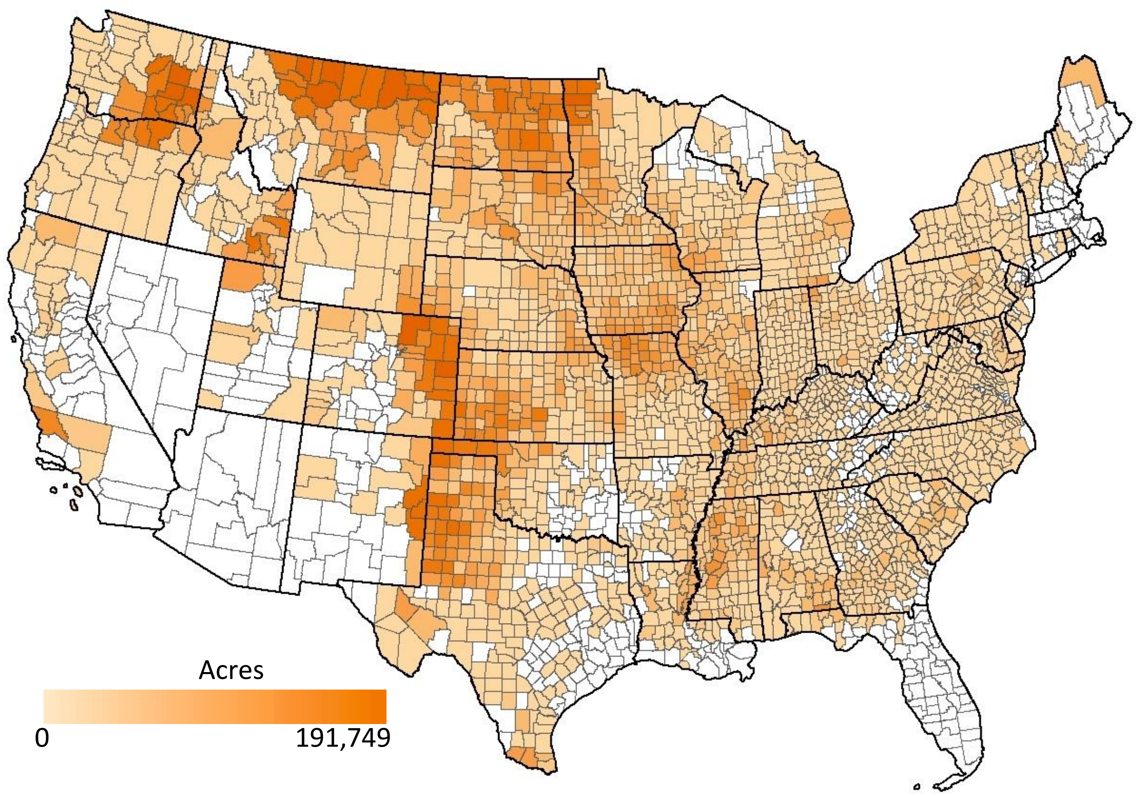


Figure 17: Change in CRP enrollment from each county-level peak (1986-2010) to 2011



In the absence of re-enrollment, approximately 18.8 million currently-enrolled acres are scheduled to exit the program by 2018 (USDA FSA, 2012a). Much of this acreage is concentrated along the Canadian border of the Northern Great Plains and the Southern Plains from Colorado to Texas (Figure 18). It is not currently clear what will happen to this acreage if it does exit the CRP. Studies from the early years of the CRP suggested that most farmers holding acres that expired in the early 1990's planned to convert at least some the area back to cropland (Heimlich and Osborn, 1993). Additionally, the apparent negative correlation in the data above between CRP and crop prices since 2007, and the rate of grassland conversion to cropland in areas where CRP has decreased, suggests that at least some of the acres exiting CRP are likely to be converted back to cropland. If the driver of CRP conversions back to cropland is the high opportunity cost of foregoing crop production, then a carbon payment that reduces the opportunity cost of grassland could prevent some expired CRP acres from converting. In a related study, Hellerstein and Malcolm (2011) concluded that a robust carbon market could offset the impacts of increased commodity prices on the cost of maintaining CRP enrollment (i.e., reducing the opportunity cost could increase enrollment).

Figure 18: CRP acres scheduled to expire between 2012-2018



Though the CRP is the largest federal program affecting grassland and cropland acreage, there are several other relevant federal programs. The Grassland Reserve Program (GRP) is a voluntary conservation program that provides support for working grazing operations to enhance plant and animal biodiversity, and to protect grasslands under threat of conversion to other uses. GRP contracts can range from restoration cost-share agreements, or 10- to 20-year rental contracts, to perpetual easements owned by the United States. As of 2011, 318,036 acres were enrolled in some type of GRP contract nationwide (NRCS, 2012b). Because of the variety of contract types, it is difficult to determine GRP acreage that may exit the program in the near future.

There are also several other federal programs that can affect grassland/cropland conversion rates, including the Conservation Reserve Enhancement Program (CREP), Wetlands Reserve Program (WRP), and the Farm and Ranch Land Protection Program (FRPP). CREP is a voluntary land retirement program that targets environmentally sensitive land. CREP contracts, which require cooperation between producers and local, state, and federal agencies or non-governmental organizations (NGOs), require a 10-15 year commitment to keep lands out of agricultural production. The WRP offers incentives to landowners to restore, protect or enhance wetlands (typically focusing on wetlands areas that have been previously altered for agricultural purposes), which can affect grassland areas that surround or support wetland complexes. FRPP provides matching funds to help purchase development rights in order to protect active farm and ranchland. Thus, FRPP can protect grazing land from conversion to non-agricultural uses. These other federal programs, however, affect a very small fraction (typically less than 1%) of agricultural land as compared to the CRP.

4.2.4 Laws, Regulations, or Programs Influencing Rangeland/Grassland Protection

Several federal laws/regulations indirectly influence rangeland and grassland protection. Most notable are the conservation compliance provisions of the Farm Bill. The so-called sodbuster and swampbuster provisions require that agricultural producers meet a minimum level of conservation on highly erodible lands and do not convert wetlands to crop production in order to qualify for benefits from a variety of USDA assistance programs (e.g., Stubbs, 2012). Starting with the 1985 Farm Bill, sodbuster provisions required farmers to meet conservation compliance (e.g., an approved management plan on newly tilled highly erodible land) in order to qualify for most of the major federal assistance programs, including price supports, crop insurance, and disaster payments. Despite some strengthening of compliance requirements in 1990 (e.g., graduated penalty), the 1996 Farm Bill weakened compliance requirements by, most notably, removing crop insurance from the list of programs for which non-compliant producer’s could lose benefits. Nonetheless, research suggests that conservation compliance has reduced soil erosion and wetland loss since its inception (Doering & Smith, 2012; Claassen, et al., 2004). The extent to which sodbuster has prevented likely grassland conversions (i.e., by reducing the potential benefits of converting grassland to cropland) is less clear. Sodbuster provisions, recall, do not disallow grassland conversion; they only require that appropriate conservation practices (e.g., conservation tillage) be put in place on newly converted acres. The future of conservation compliance is also unclear. The pending re-authorization of the Farm Bill has yet to reveal the new conservation compliance standards (including whether crop insurance benefits will be tied to compliance).

There are also a large suite of other federal, state and local laws/regulations that may indirectly protect grassland. There are several grassland endemic endangered species, for example, whose habitat may be protected from conversion under the Endangered Species Act (e.g., greater sage-grouse, lesser prairie chicken, black-footed ferret, prairie dog, giant kangaroo rat, and Wyoming toad). Additionally, a variety of state and local land-use ordinances, such as zoning restrictions or open space preservation programs, may provide local regulatory mechanisms that indirectly conserve existing grasslands (e.g., California’s Rangeland, Grazing Land and Grassland Protection Program).

4.2.4.1 Land Valuation

For AGC projects, a land use change protocol would require an additionality test to provide assurance that project areas would have likely been converted out of grassland in the absence of the project. One additionality test option is a fair market value test (similar to Reserve Forest Protocol V3.2) to determine if the value of a project area as grassland is significantly less than the value in an alternative use. Such a test requires estimates of grassland and alternative land use values. In practice, the demonstration of land values for specific parcels under defined land uses can be directly quantified through a formal appraisal. For this Issue Paper, however, such direct quantification is is not available. Below we discuss the methods and data that may be used to assess the value of grassland and alternative uses without direct appraisal data.

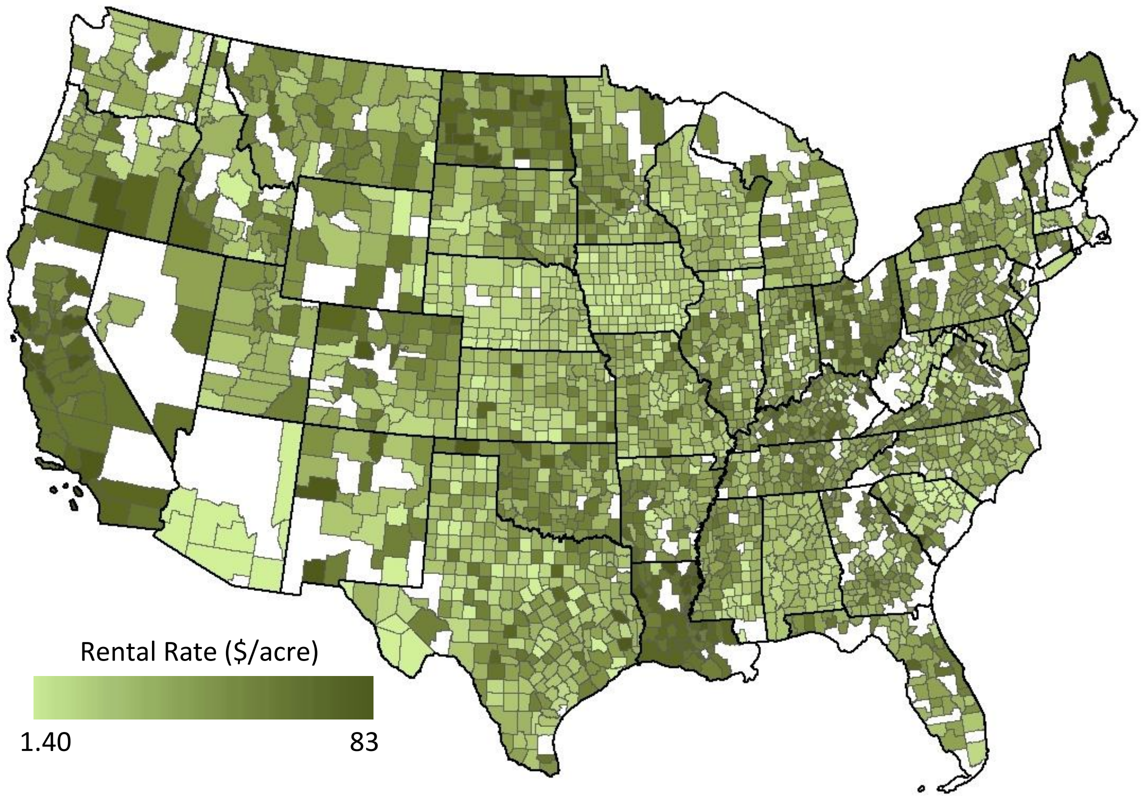
4.2.4.1.1 Grassland Valuation

There are many potential values that could be applied to grassland. Across much of the US, the primary market use of private rangelands is livestock grazing. As such, values associated with livestock production could reasonably serve to estimate the value of grassland (i.e., pasture and rangeland) similar to the use of timber values in forest carbon offset protocols. Annual NASS data on pasture and cropland rental rates by county are also readily available. Thus, one potential method for valuing grassland is to use county-level average pasture rental rates (Figure 19; Table 25 in Appendix B).

There are several limitations, however, to using rental rates to proxy for rangeland values:

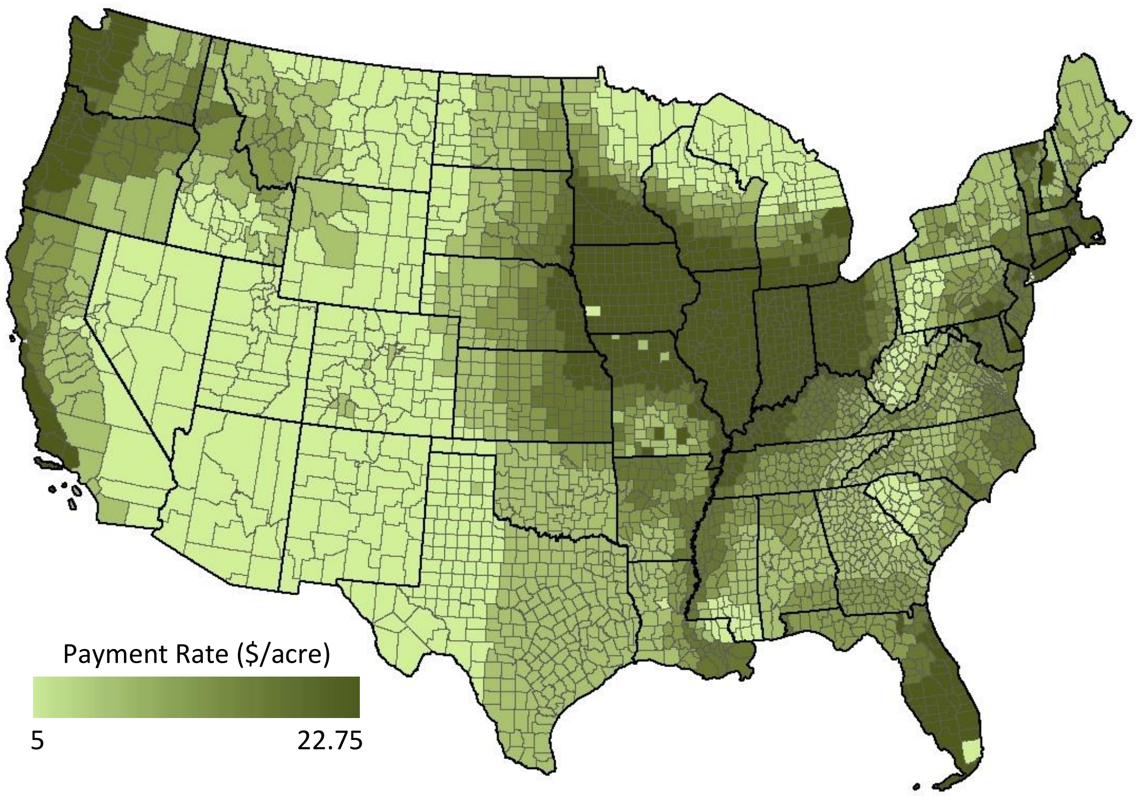
- The quality of rental rate data is highly variable. NASS county-level average rental rate estimates are based on survey responses. As such, the validity of the average rate estimate is dependent, in part, on the number of responses received.
- Individual rental agreements represented in survey responses are themselves highly variable. Individual rental agreements, for example, have different management stipulations (e.g., whether the owner or renter repairs fences), which influence the reported rental rate.
- Pastureland quality and hence productivity can be highly variable within a county – high quality pasture can have significantly higher rental rates. Thus, unless a county has many responses across many different types of agreements, representing many different land quality types, the average rental rate may not be representative of typical grassland values in that county.
- Several counties in the conterminous US do not have available rental rate data – some counties have no, or too few, survey responses – thus, rental rates cannot be used to generate a value for all US grassland.
- Rental rate data does not distinguish between different pasture land cover types. Thus, there is no way to differentiate values between, for example, grassland dominated vs. shrubland dominated pastures, except to the extent that a single cover type dominates specific counties.
- Rental rates only capture the grazing related use values of grassland. They do not capture other use values (e.g., hunting and ecotourism) or non-use values that can co-exist with grazing.

Figure 19: Average pasture rental rates



There are no readily available datasets to assess other important values (e.g., ecosystem services, such as wildlife habitat or flood control) associated with grassland. Rashford et al. (2012a) estimated that ecosystem goods and services account for 37% to 68% of the value of western US rangelands (with livestock forage values accounting for the remainder). One alternative to account for ecosystem service values is to use payment rates associated with the Grassland Reserve Program (GRP). GRP is a voluntary program that compensates landowners to restore and protect grassland, including rangeland and pasture. The intent of the program is to enhance plant and animal biodiversity and protect grassland under threat of conversion. Thus, GRP rental rates could serve as a proxy for many of the non-market ecosystem services provided by rangelands (Figure 20). Since GRP also targets grassland with high conversion risk, rental rates tend to be higher in locations facing development pressure. These higher rates do not necessarily reflect greater ecosystem services and may overestimate grassland values. On the other hand, areas with high conversion risk may also have a lower supply of ecosystem services, which suggest ecosystem services in these areas should be assessed at a higher value.

Figure 20: 2011 Grassland Reserve Program payment rates



4.2.4.1.2 Cropland Valuation

Similar to grassland values, cropland values can vary widely over space and time depending on cropland net returns, which are driven by soil quality, climate and crop prices. Thus, a careful assessment of cropland values across the country should account for these major factors. Doing so, however, requires significant data collection. One alternative is to value cropland according to NASS cropland rental rates (Figure 21 and

Figure 22). For reasons similar to those presented for grassland rental rates above, cropland rental rates may not be wholly representative of the value of cropland. Nonetheless, rental rate data are readily available, and theoretically should capture the productive (not asset) value of cropland.

Figure 21: Average rental rates for non-irrigated cropland

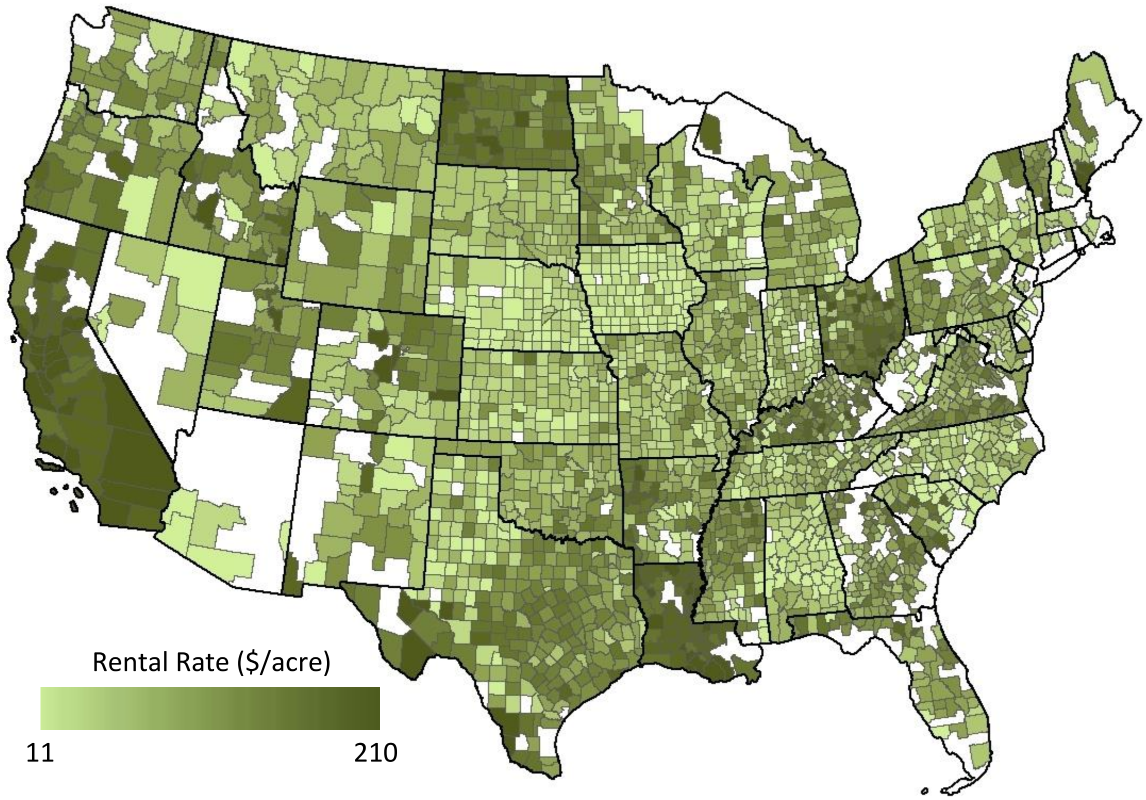
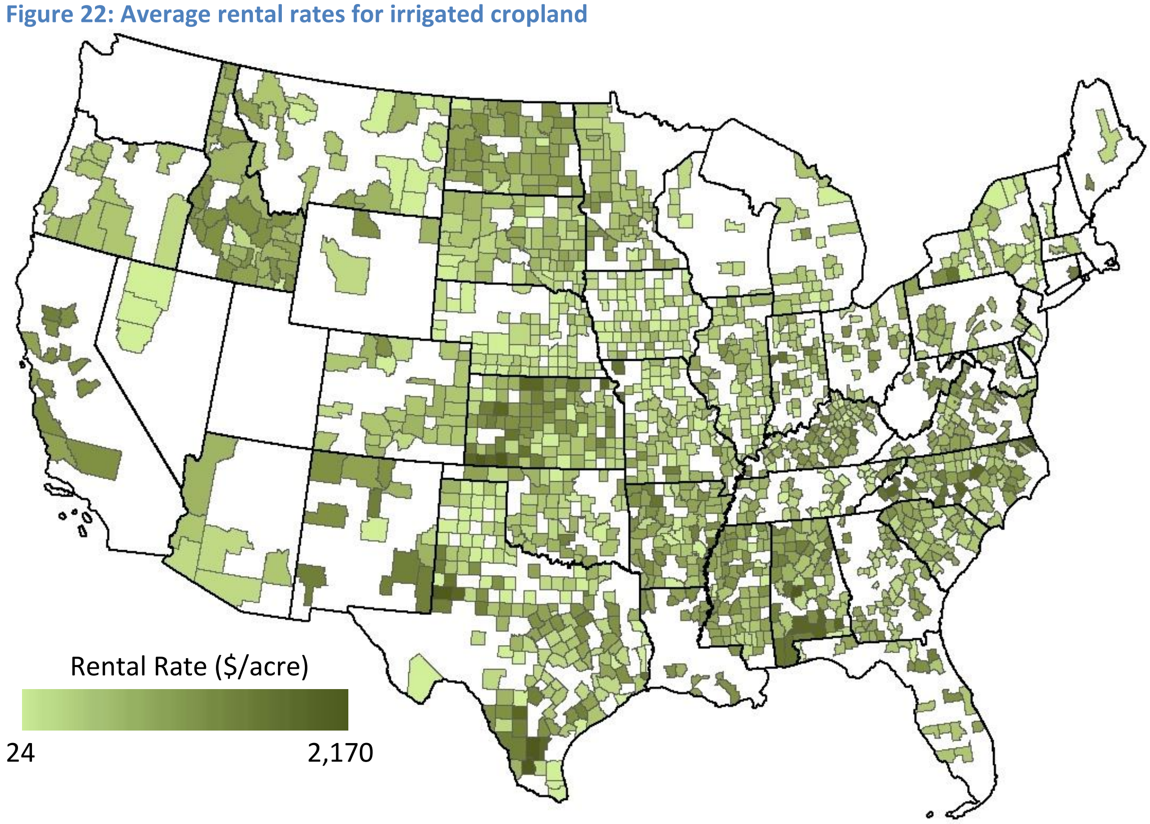
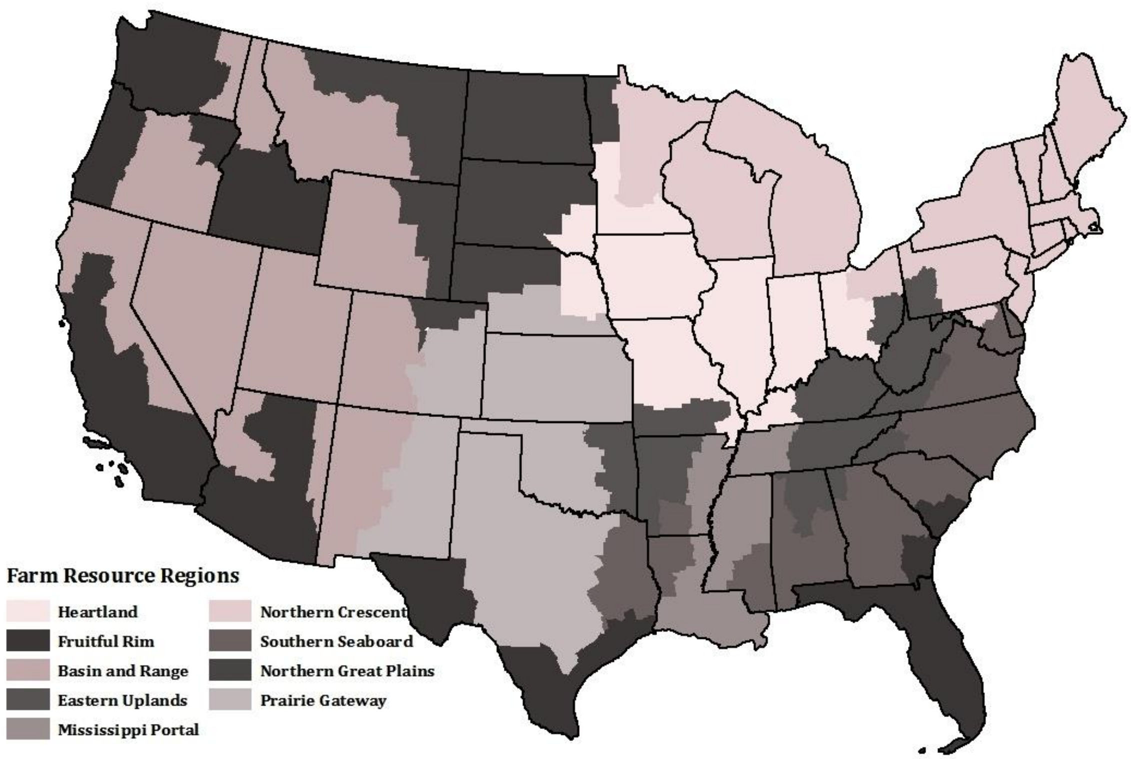


Figure 22: Average rental rates for irrigated cropland



An alternative to rental rates is Economic Research Service (ERS) cost and returns data. The ERS data provides estimates of the net returns per acre for major field crops across nine resource regions (Figure 23). Though ERS estimates lack realistic spatial variation (i.e., are highly aggregated), they do incorporate typical yield and price differential across the broad regions. Thus, they may capture cropland values better than rental rates, without requiring substantial data collection (e.g., yields by county by crop). ERS provides production costs and returns for major commodities, including corn, wheat, soybeans, sorghum, barley, oats, peanuts and cotton. However, not all crops are grown in every region.

Figure 23: Economic Research Service (ERS) Farm Resource Regions



The best proxy for cropland production values is "value of production less operating expenses", commonly referred to as returns over variable cost (ROVC). ROVC excludes fixed costs, which are highly variable across individual producers. ROVC tends to be highly variable across regions and commodities (Table 11). In general, ROVC tends to be higher than the typical non-irrigated cropland rental rates reported above, but can also be significantly lower than the irrigated cropland rental rates in some areas.

Table 11: Returns over variable costs by commodity and farm resource region, $/ac

	Commodity							
Region	**Corn**	**Wheat**	**Soybeans**	**Sorghum**	**Barley**	**Oats**	**Peanuts**	**Cotton**
Heartland	472.60	214.55	475.21	255.20	51.67	139.89		396.48
Northern Crescent	546.11	277.12	410.24		201.38	160.04		
Northern Great Plains	358.65	173.02	265.93	198.99	110.13	147.49		
Prairie Gateway	415.51	99.83	300.05	74.63		67.14	335.50	-60.88
Eastern Uplands	312.40		309.51					
Southern Seaboard	454.68		292.93				574.34	259.09
Fruitful Rim		366.10		89.35	220.64		288.41	313.30
Basin and Range		312.50			111.71			
Mississippi Portal	300.41		245.92					361.69

Source: USDA Economic Research Service (2012)

4.2.4.1.3 Other Land Values

Grassland may also convert to uses other than cropland. The most common end-uses for grassland (other than cropland) include developed land and forestland. It is difficult, however, to reasonably assess the values for these other land uses, though they both have clear use-values (e.g., residential development and timber). The difficulty arises because of the heterogeneity in developed land and forestland types likely to replace grassland. Development in previously grassland-dominated areas could range from commercial development (e.g., mine or energy extraction sites) to large-lot ex-urban residential development. Neither is easy to value across the conterminous US. Residential land values, for example, are determined by local land-market conditions; thus, they are highly variable over space and time. Even if we assumed that rural residential development was the primary use of concern, values would be difficult to assign. Every county in the US regularly collects land value data for tax assessment purposes. This data, however, is not collected in any standard format, nor is it stored in any central repository. Thus, collecting assessors' data on residential land values often requires a county-by-county individual data collection process. The US Census Bureau does provide annual county-level estimates of median home values, but these values do not isolate the value of vacant residential land that would most accurately capture the potential value of converting grassland to residential development.

4.3 Estimated Cost of Reductions

In this Issue Paper, we utilize land rental rates and conservation program payments as a proxy for the cost to incentivize AGC and CCG project activities. For AGC project activities, this analysis relies on the Grassland Reserve Program's (GRP) county level rental rate data. The GRP is a voluntary enrollment program designed to financially encourage landowners to protect grassland. The Farm Service Agency (FSA) establishes rental rates for all counties, which are based on the relative productivity of the soil (i.e., the soil's ability to stimulate plant growth) and the pre-established cash rental rate estimates for the county. The program offers several enrollment options including term contracts between five and 20 years, a permanent easement, and a cost share option to underwrite restoration work (USDA, 2010). Given the voluntary nature and enrollment options, GRP rental rates offers a useful proxy for evaluating what minimal carbon payment levels may have to be to encourage landowners to participate in an AGC project. As Table 12 indicates, the base cost of sequestering carbon through AGC activities ranges from \$5.59 per tCO_2e to an effectively infinite cost where emission factors are negative.

It is critical to recognize that these cost estimates presume that a landowner would be indifferent between GRP payment and a carbon payment. Considering the nascent nature of carbon markets compared to a well-established government program where the obligations are clearly laid out for the landowner (and less burdensome than carbon offset rules), this assumption is likely to underestimate the actual cost to incentivize AGC activities through carbon offset projects. For example, in contrast to GRP rental rates, carbon market participation would also introduce additional costs that would be reflected in the per-unit cost to develop offset credits. Cost and risk factors affecting the per-unit carbon price demanded by offset credit developers would include, for example: expenses from project development and documentation, monitoring, and third-party verification; carbon credit delivery risk; price risk; and additional opportunity costs due to longer contract/commitment periods. Considering these additional risks and costs together, it should be reasonably expected that the carbon payments would need to be substantially higher than GRP rental rates in order to induce significant landowner enrollment in AGC project activities.

Table 12: Estimated carbon cost range for AGC

IPCC Climate Zone	**Est. EF* Range** (tCO_2e\ac\yr)	**Avg. GRP Rate** (\$\ac)	**Est. Cost Range** (\$/tCO_2e)	**Est. Mean Cost** (\$/tCO_2e)
Cool Temperate Dry	(0.06) – 0.66	9.15	13.77 - ∞**	23.64
Cool Temperate Moist	(0.22) – 0.98	12.90	13.15 - ∞	26.22
Warm Temperate Dry	0.16 – 0.83	8.40	10.14 – 51.43	15.64
Warm Temperate Moist	(0.17) – 1.24	11.16	9.03 - ∞	14.16
Tropical Dry	0.24 – 0.97	8.00	8.24 – 33.81	11.41
Tropical Moist	(0.15) – 1.66	9.30	5.59 - ∞	10.33

**Emission Factors derived as discussed in Section 3.2 above.*
***In cases where the minimum emission factor is negative (as indicated by parentheses), there is no cost of carbon which would generate revenue, hence the use of the "∞" symbol.*

4.4 Potential Reduction Opportunity

The annual technical potential estimate for AGC activities is calculated along the climate zones in the continental United States, as described in Section 3.2. As such, the calculation draws on the grassland and shrubland conversion rates illustrated in Table 10, and Tier II factors to account for land uses, tillage and cropping practices (Ogle, et al., 2003; Ogle, et al., 2006). The low end to high end of values from the

Tier II assessment are used to illustrate the potential range of annual emissions associated with avoided grasslands conversion.

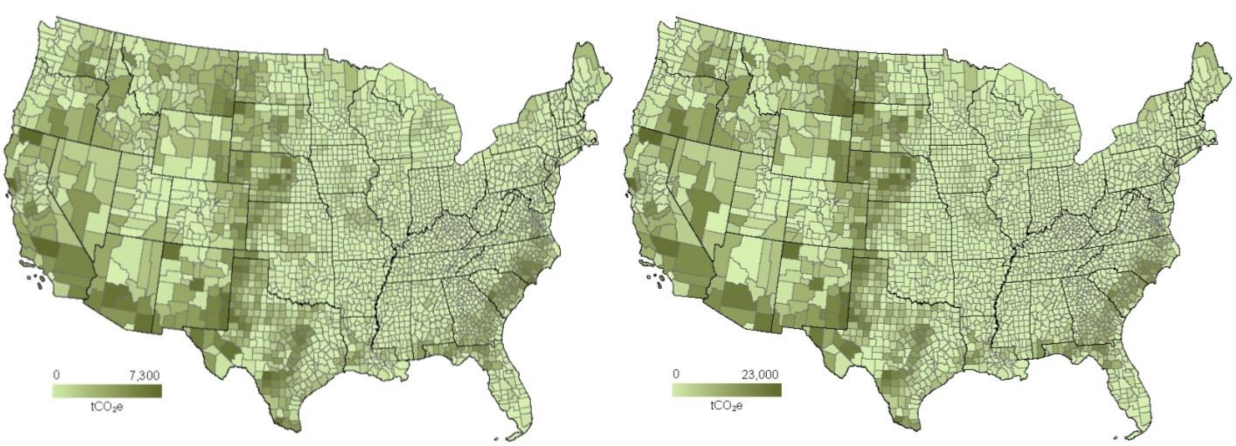
Based on this approach, the domestic technical potential for avoided grassland conversion is between 3.6 and 154 thousand tCO_2e per year. This assessment does not include the sequestration potential associated with carbon stock losses in plant biomass. As such, it offers a conservative approximation of the technical potential for AGC activities. Not surprisingly, Table 13 indicates the greatest potential for avoiding SOC losses due to conversion is in the cool temperate dry region and warm temperate dry region. Figure 24 shows the geographic distribution of these potential emission reductions by county. Most of the potential AGC emission reductions are concentrated along the western Great Plains, extending in a north-south corridor running from the border of Montana and North Dakota down to Texas and New Mexico. Additional hot spots for AGC technical potential appear across California and in the Atlantic coast in the Southeastern US.

Table 13: Estimated technical potential for AGC emission reductions

IPCC Climate Zone	Acres	Low End ($tCO_2e\yr$)	High End ($tCO_2e\yr$)	Mean ($tCO_2e\yr$)
Cool Temperate Dry	77,536	(4,830)	51,525	30,006
Cool Temperate Moist	6,771	(1,498)	6,642	3,331
Warm Temperate Dry	55,370	9,044	45,883	29,734
Warm Temperate Moist	13,589	(2,353)	16,785	10,708
Tropical Dry	19,157	4,533	18,600	13,429
Tropical Moist	8,718	(1,328)	14,493	7,846
Total	**181,142**	**3,567**	**153,929**	**95,055**

Numbers in parentheses are negative, indicating an increase in GHG emissions.

Figure 24: Geographic distribution of estimated technical potential for AGC activities



Note: The maps on left and right present the low- and high-end estimates for annual emission reduction potential for each county, respectively. [The magnitude of these reductions, as labeled, is incorrect. However, the geographic distribution of the potential reductions is still accurate.]

5 Conversion of Marginal Cropland to Grassland (CCG) Characterized

5.1 Scope and Definition of Project Activities

In its Request for Proposals (RFP) for this Issue Paper, the Reserve proposed a project activity for CCG defined as:

> *Setting aside cropland that is otherwise capable of producing food and converting it into a permanent non-tree vegetative cover, such as grassland, can substantially increase soil carbon sequestration as a result of eliminating soil disturbance, increasing permanent belowground biomass in roots and shoots, and possibly increasing overall organic matter inputs from permanent vegetation compared to a cultivated system.*

The following discussion builds from this definition to cover the science and policy considerations regarding this potential project type.

5.1.1 Project Duration and Crediting Periods

In general, there are two important qualities of CCG project activities which support a project duration and/or crediting period on the scale of 20-30 years. First, CCG projects would be carried out in the context of competing agricultural land management decisions. For several practical and generally self-apparent reasons, agricultural land management is applied over a timeframe that is generally shorter than the forestry sector (for which the Reserve has approved project durations up to 100 years). Second, the primary source of emission reductions for CCG projects is expected to be the additional SOC sequestration due to the establishment of grassland cover. Under the prevailing IPCC approach for SOC accounting, soils are generally assumed to achieve equilibrium SOC stocks after twenty years under a particular land management regime. Thus, the emission reduction benefits achieved by CCG project activities should generally be complete within this timeframe.

The length of crediting periods may also be needed to address the potential for piecemeal conversion of Project Area grasslands in the baseline scenario, which may also be affected by the choice of accounting approaches to estimate the baseline conversion extent.

5.1.2 Activities Required to Transform Cropland to Grasslands

Undoubtedly, many current cropping areas have sufficient grass seed sources and/or rootstock and may revert to grasslands in a reasonable amount of time once cropping is ceased. However, on some areas, natural succession will only occur very slowly and a more active approach to grassland restoration is warranted[7]. Therefore, we will separate the approaches to transform cropland to grassland into two broad categories: grassland establishment not requiring full re-seeding and grassland establishment with full re-seeding. However, it is clear that these two approaches represent opposite ends of a spectrum and that, in practice, some targeted re-seeding of selected areas is appropriate. In addition, whether natural succession suffices to revert cropland to grassland, or a more active restoration

[7] In general, grasslands from warmer and more humid areas tend to regenerate more consistently and rapidly whereas more arid and cold areas take longer or get stuck in undesirable states unless actively restored. This set of factors is also drives the expected amount of carbon sequestered through restoration. The type of restoration may be a secondary factor, and more determinant of the length of time required to achieve the benefits.

approach should be implemented depends to a large extent on the project-specific conditions and a financial analysis of specific farm characteristics.

Natural succession requires less inputs and costs but will occur more slowly than active restoration under most circumstances. As a consequence, the land will only allow light grazing for many seasons until the grass gets well established. Active grassland restoration may enable higher-intensity grazing even the first year after restoration. Whichever approach is taken, it is important to consult regularly with a certified rangeland specialist to optimize the establishment of the grass and forage species. Finally, it is clear that grassland restoration is a process that can take many years and that will lead to carbon sequestration for decades.

5.1.2.1 Grassland Establishment Not Requiring Full Re-Seeding

Re-seeding marginal croplands to grasslands entails a series of management and ecological decisions that can affect overall ecosystem function, grassland health and persistence and total capacity for carbon storage. Sufficient seeds and roots of persistent perennial grasses such as bermudagrass are present at the time of conversion in some cropping fields, which would enable the establishment of grassland without the need to completely re-seed. Instead, fencing and light grazing can be coupled with adequate recovery time to encourage the establishment of new grass stands (Rinehart & Sullivan, 2010). A close collaboration with a certified rangeland specialist is advised to optimize the grassland establishment and grazing through management.

In addition to fencing and light controlled grazing, producers can utilize a variety of practices to assist them in the persistent establishment of grasslands including adjustments in fertility and pH levels to those desired for the grass species of interest after cessation of cropping. For example, legumes typically require medium to high levels of lime, phosphorus, and potassium. If undesirable and aggressive weeds are threatening to suppress the emergence of existing seeds, the planting of a so-called “nurse crop” such as hay, silage, or grain can be considered to suppress (annual) weeds (Rinehart & Sullivan, 2010).

5.1.2.2 Grassland Establishment with Full Re-Seeding

Croplands that once were native grassland are prime candidates to establish grassland. However, many typical U.S. croplands, especially degraded croplands, are low in organic matter, have poor soil structure and will only slowly revert to grassland without re-seeding. When insufficient existing seeds or rootstock is present, an area can be reseeded in whole. Grasses should be selected with care and consideration for local ecosystems, economics, cattle, and biodiversity (Rinehart & Sullivan, 2010). A variety of factors affect grass selection for long term conservation plantings including regional adaptation, soil type, climate, seed availability, stand characteristics, maintenance needs, costs and returns and invasiveness (Bidwell & Woods, 2010). Seedbed preparation operations will generally entail the use of herbicides and other chemicals, mechanical site preparation (e.g., tillage), or a combination of the two (NRCS, 2009a).

5.1.3 Potential for Retiring Cropland from Production to Increase Soil Carbon Stocks

5.1.3.1 General Sequestration Rates of Grassland Restoration

In general, established grasslands sequester more carbon than croplands. Comparisons of cultivated versus forage seed sown grassland demonstrated that SOC mass was usually significantly higher in grassland restorations, gaining between 0.9 and 1.2 tCO_2e/ac/yr (0.6 and 0.8 Mg C/ha/ yr) (Mensah, et al., 2003). Schuman et al. (2002) suggest similar rates of sequestration with new grasslands storing as much as 0.9 tCO_2e/ac/yr (0.6 Mg C/ha/yr) compared to 1.5 to 4.5 tCO_2e/ac/yr (0.1 to 0.3 Mg C/ha/yr). Furthermore, in a comprehensive literature review Guo and Gifford (2002) indicate an average SOC change of 18% to 22% across 76 studies for "crop-to-pasture" transitions, though their analysis did not separate re-seeded grasslands from naturally reverting grasslands. In this sense, conversion to grasslands can provide an upfront increase in SOC sequestration (Schuman, et al., 2002). Following conversion some data also suggests that sandy soils accumulate greater carbon amounts than silt loam soils (Su, 2007). Additional data shows that the accumulation of carbon in the soil following conversion of cropland to grasslands is continual. In a study that replicated a 40-year timescale following cropland conversion to perennial grasslands, McLauchlan et al. (2006) found that SOC accumulated at a constant rate of 0.9 tCO_2e/ac/yr (62.0 g C/m^2/yr). At this rate, they determined that the grassland would have equivalent SOC contents to an unplowed native prairie within as few as 55 years.

5.1.3.2 Seeded vs. Natural Succession Sequestration Rates

A large body of literature discusses the carbon benefits associated with conversion of cropland to grasslands as indicated above. Post and Kwon (2000) provide an extensive review of data showing carbon sequestration rates across regions and ecosystem types in conversion to grasslands (Figure 25). Their data suggest that for cool temperate steppe grasslands (those that would be typical and relevant of this protocol) there are notable differences in carbon sequestration depending on grass species and conversion method. Shifting from cultivated land to seeded grass yielded no average SOC increase according to Robles and Burke (1998)[8], while shifting from cultivated to perennial grass systems resulted in an average SOC increase of 1.65 tCO_2e/ac/yr (110 g C/m^2/yr). Other studies have found that a transition from cropland to seeded grasslands can result in higher SOC gains than combinations of fallow, hay, wheat, and legume/green manure combinations (Bremer, et al., 2002).

Furthermore, species types also greatly affected the total rate of SOC increase in cultivated to improved pasture systems (with equal time since agriculture). Russian wildrye resulted in an average rate of SOC change of 0.10 tCO_2e/ac/yr (6.86 g C/m^2/yr), while alfalfa grass mixtures resulted in up to 0.51 tCO_2e/ac/yr (34.15 gC/m^2/yr) and crested wheatgrass fell in between with 0.28 tCO_2e/ac/yr (18.87 gC/m^2/yr) (Bremer, et al., 2002). These results are compatible with other research also suggests that greater grass species richness and biodiversity can result in higher rates of carbon sequestration than less diverse species compositions (Le Roux & McGeoch, 2008). Interseeding alfalfa in mixed-grass rangeland systems has also proven to increase SOC sequestration rates above those of native rangelands (Mortenson, et al., 2004). In addition, some evidence suggests that native species can have

[8] It should be noted that the authors found that following 6 years of grass-seeded Conservation Reserve Program management carbon and nitrogen pools in fine particulate organic matter and total soil organic matter did not increase. However, the research did find that some pools of SOM may increase at the microsite and field scale.

more extensive root systems than introduced species and more root biomass below ground, resulting in higher SOC levels compared to introduced species (McConnell & Quinn, 1988). Evidence also suggests that grazing can help to maintain species diversity in re-established native systems (Schellenberg et al., 2012; Schellenberg and Iwaasa, 2008; Wedin and Tilman, 1996).

Figure 25: Literature review of soil carbon sequestration potential under different conversion patterns and management

Rates of soil carbon accumulating during pasture establishment.

Site History	Years since Agriculture	Soil sample Depth (cm)	Rate of C Change ($g\ m^{-2} yr^{-1}$) max.	avg.	Reference
Cool temperate steppe					
cultivated to perennial grass	12	300		110.0	Gebhart et al. 1994
cultivated to abandoned field	50	10		3.1	Burke et al 1995
cultivated to seeded grass	6	5		0.0	Robles and Burke 1998
cultivated to improved pasture					White et al. 1976
Russian wildrye	8	7		6.86	
crested wheatgrass	8	7		18.87	
B-I-ALF(full)	8	7		14.01	
B-I-ALF(short)	8	7		34.15	
mine tailing to grass-forb meadow	5-80	10	60.0	4.01	Titlyanova et al. 1988
coal mine spoil to dry grassland	28-40	120		28.2	Anderson 1977
Subtropical moist forest					
cultivated to pasture					Lugo et al. 1986
Atlantic	37	18		-16.22	
Caonilas	37	18		-48.65	
Culebrinas	37	18		100.0	
Northwest	37	18		8.11	
West	37	18		37.84	
East	37	18		35.14	
Southeast	37	18		10.81	
Southwest	37	18		67.75	
South	37	18		113.51	
Turabo	37	18		24.32	
Tropical dry forest					
forest to unimproved pasture	23	10		-17.4	Trumbore et al. 1995
forest to improved pasture	23	10		-13.0	Trumbore et al. 1995
Tropical moist forest					
native forest to pasture	10	40		-30.0	Desjardins et al. 1994
mature forest cleared to pasture					Neill et al. 1997
Purto Velho	7	10		83.18	
Calcaulandia	8	10		-4.02	
Nova Vida-1	81	10	342.72	15.23	
Nova Vida-2	20	10	174.81	24.10	
Ouro Preto-Benjamin	20	10	115.54	39.27	
Ouro Preto-lenk	20	10	-84.13	2.37	
Vilhena	12	10	114.83	91.91	
Purto Velho	7	30		-14.29	
Calcaulandia	8	30		-90.00	
Nova Vida-1	81	30	460.0	21.85	
Nova Vida-2	20	30	410.0	59.0	
Ouro Preto-Benjamin	20	30	110.0	74.5	
Ouro Preto-lenk	20	30	49.17	17.0	
Vilhena	12	30	134.0	32.5	
Tropical wet forest					
native forest cleared for pasture					Veldkamp 1994
Eutric Hapludand	25	50		-87.2	
Oxic Humitropept	25	50		-6.0	
native forest cleared for pasture					van Dam et al. 1997
Andic Humitropept	18	60	142.03	34.93	

Note: Reproduced from Post & Kwon (2000). *1 $gC/m^2/yr$ is equivalent to 0.01 MgC/ha/yr, or about 0.015 $tCO_2e/ac/yr$.*

5.2 Land Use Trends

5.2.1 Data Availability for Estimating Cropland to Grassland Conversion Rates

The data available for estimating cropland to grassland conversion rates is the same as the data for estimating grassland to cropland conversion rates. The issues of scale, frequency and land use classes also apply in the case of CCG. Please see Section 4.2.1 above for a broader discussion of the available data sets and data issues. In brief, NASS county-level data provides good estimates of cropland area (which can be disaggregated by the major field crops) at five-year intervals for a long time series. The NASS data, however, has a fairly coarse definition of "grassland" ("Permanent pasture and rangeland, other than cropland and woodland pastured"). Additionally, since all NASS data are aggregated to the county-level, it cannot be used to estimate explicit conversion rates between cropland and grassland. The NRI data also provides estimates of cropland and grassland area at five-year intervals (beginning in 1982). Beginning in 1997, however, NRI data are only available at the state-level. Thus, it can be used to estimate long-term trends in state-level cropland and grassland area, but not to estimate explicit conversion rates. Lastly, the NLCD provides plot-level (30m resolution) data on land cover, including "cropland," "grassland/herbaceous" and "shrub/scrub" classes, at five-year intervals beginning in 2001[9]. Though the time period is shorter, the longitudinal structure of the NLCD data (i.e., the same plots observed at multiple points in time) allow us to explicitly calculate rates of change between land classes.

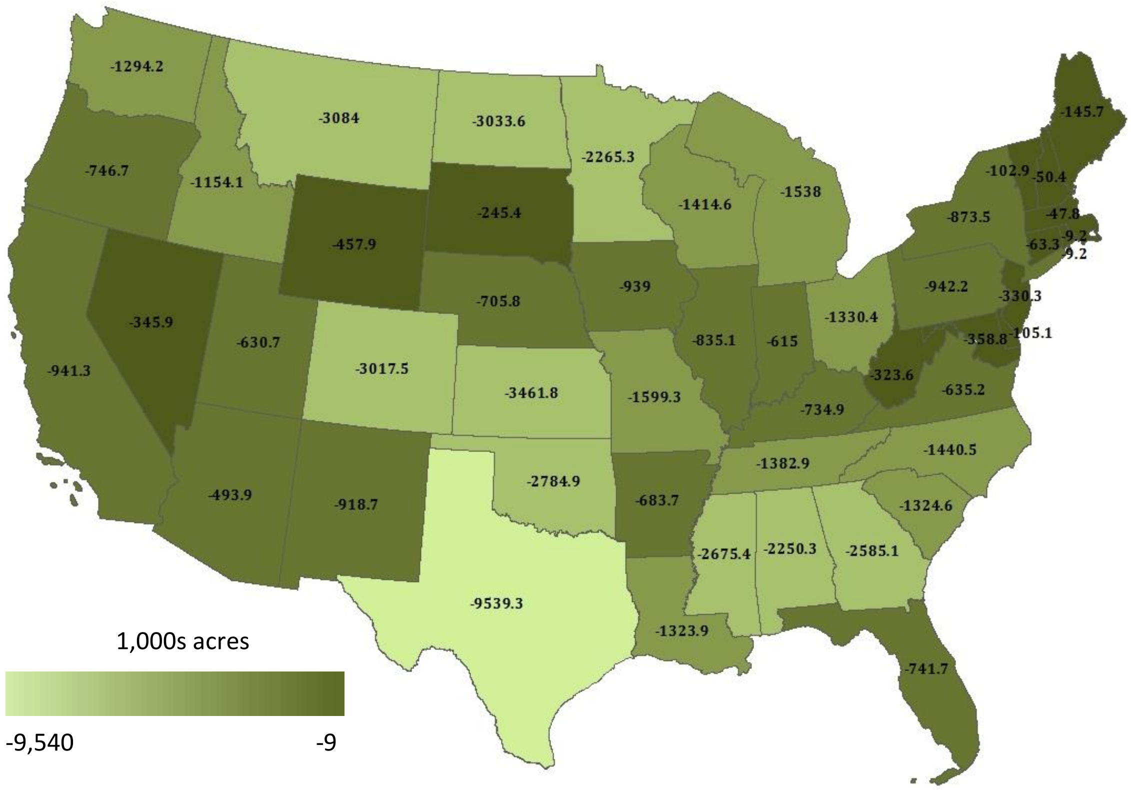
5.2.1.1 Cropland Conversion Trends

Long-term trends from the NRI (1982-2007) indicate a significant decline in "cropland" area nation-wide (62.5 million acres). On average, "cropland" has decreased by 2.5 million acres per year since 1982. The decrease in "cropland" occurred concurrently with large increases in "CRP" (32.9 million acres) and "developed land" (40.3 million acres), suggesting that "cropland" largely converted to these uses (see table 2.1 above). Additionally, "pastureland" and "rangeland" also decreased over the time period (12.3 million acres and 8.8 million acres, respectively), suggesting that "cropland" is not generally converting to these uses. Since the NRI data are only available at the state-level, however, we cannot determine explicit conversion paths for "cropland."

The decrease in 'cropland" depicted in the NRI data is also widely distributed across the country (Figure 3.1). Every state experienced a decrease in "cropland" area over the time period. Relatively large decreases are also spatially dispersed, with decreases of approximately 9.5 million acres in Texas, 3.5 million in Kansas, 3.1 million in Montana, and 3 million in North Dakota. The states with the lowest absolute decreases are all small New England states – i.e., they have few "cropland" acres compared to other states. But even the low absolute "cropland" decreases in these states represent relatively large relative decreases (e.g., the Rhode Island decrease of 9,000 acres represented 30% of the 1982 "cropland" area). The lowest acreage decreases in cropland ranged from 1% to 4% and were concentrated in highly crop-centric states (South Dakota, Iowa, Illinois, and Indiana).

[9] NLCD data are also available for 1992, but differences in class definitions make it difficult to compare the 1192 data with the 2001 and 2006 data.

Figure 26: Change in NRI cropland acres by state (1982-2007)



Even comparing the state-level changes across NRI categories does not make fully clear the transition paths for "cropland" (Figure 26). The data suggest that "developed land" is the likely end-point for much of the converted "cropland" in certain regions (Figure 27), while "CRP" dominates in other regions (Figure 28). The NRI national-level conversion matrix (see Figure 6) indicates that "pastureland," "CRP," and "developed land" were the dominant end-points for converted "cropland" between 1982 and 2007. It is impossible, however, to determine explicit sub-national conversion pathways using the NRI data.

Figure 27: Change in NRI developed acres by state (1982-2007)

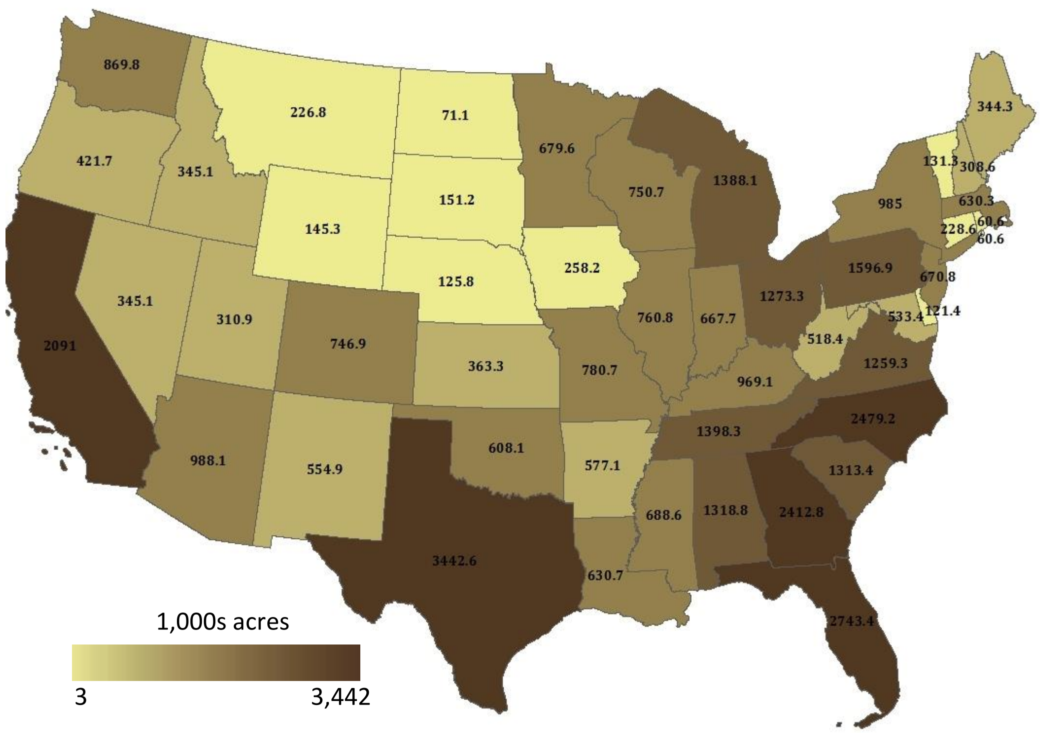
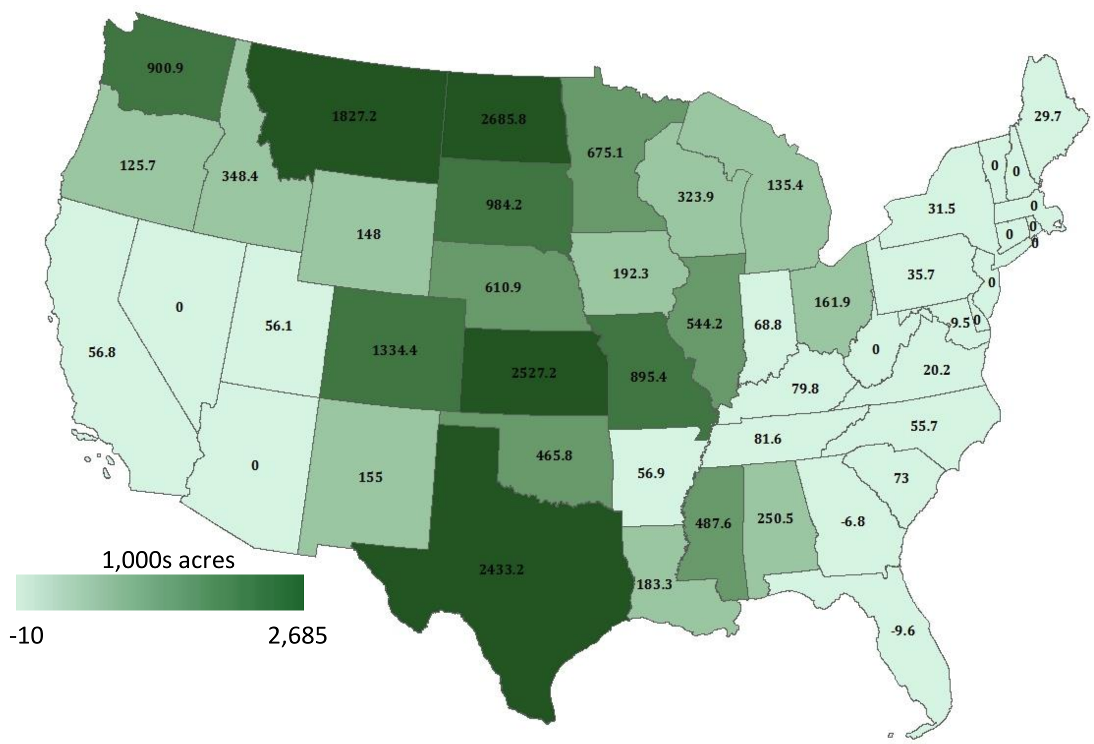


Figure 28: Change in CRP acres by state (1982-2007), 1,000s acres



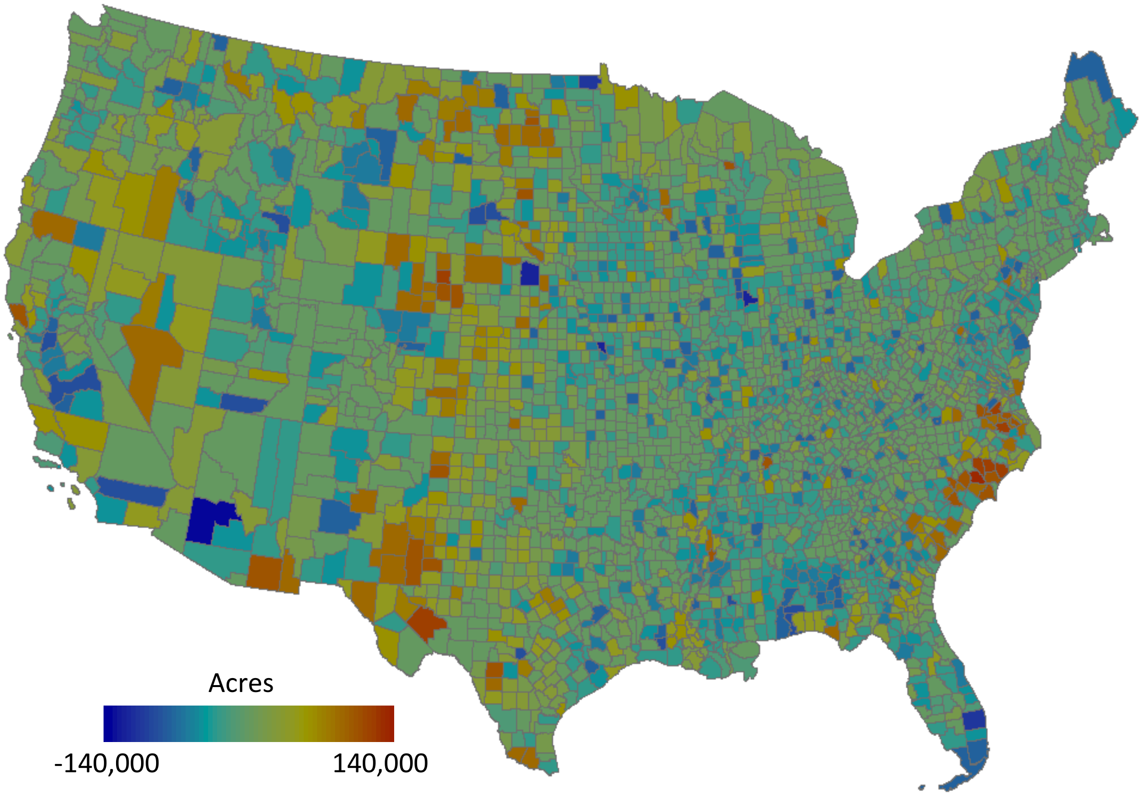
From 2001 to 2006, “cropland” area in the NLCD dataset decreased (net of conversion to “cropland”) by approximately 1.1 million acres. “Cropland” that converted had a broad array of end-points (Table 14). As suggested by the NRI data, significant “cropland” has converted to “developed land” (698,000 acres, 2001-2006). Significant “cropland” area, however, also converted to “grassland” and “shrubland” (693,000 acres), which may include some cropland-to-CRP conversion, ‘forest land” (284,000 acres) and “other land” classes (410,000 acres; primarily water areas). “Cropland” conversions from 2001 to 2006 do not, however, indicate that significant “cropland” area is converting to “pasture/hay.” It is difficult with the NLCD data, however, to assess whether “cropland” is converting more frequently to non-grazed vs. grazed grassland. Clearly “cropland” conversion to “pasture/hay” is relatively small; however, the NLCD “grassland/herbaceous” class can also include grazed land.

Table 14: Conversion matrix for cultivated cropland from NLCD data (2001-2006)

	Thousands of Acres Converted to:						
	Cult. Crops	**Developed**	**Grass/ Herb**	**Pasture/ Hay**	**Shrub/ Scrub**	**Forest**	**Other**
Cult. Crops	307,877	698	323	19	370	284	410

Similar to the NRI data, the NLCD indicates widely disbursed net changes in “cropland” area (Figure 29). Net of conversions to “cropland,” changes in county-level “cropland” area are nearly uniformly distributed across the country – with many counties gaining “cropland” area as neighboring counties lose.

Figure 29: Net change in NLCD cultivated cropland acres (2001-2006)



5.2.1.2 Conversion Rate Estimates

We estimate cropland to grassland conversion rates using the same procedures described for AGC in Section 4.2.2.1 above. Here we focus on the gross conversion rates between the NLCD "cultivated cropland" class, and the "grassland/herbaceous" and "shrub/scrub" classes. Also, as noted in the previous section, we are unable to distinguish conversion rates between grazed and non-grazed grassland.

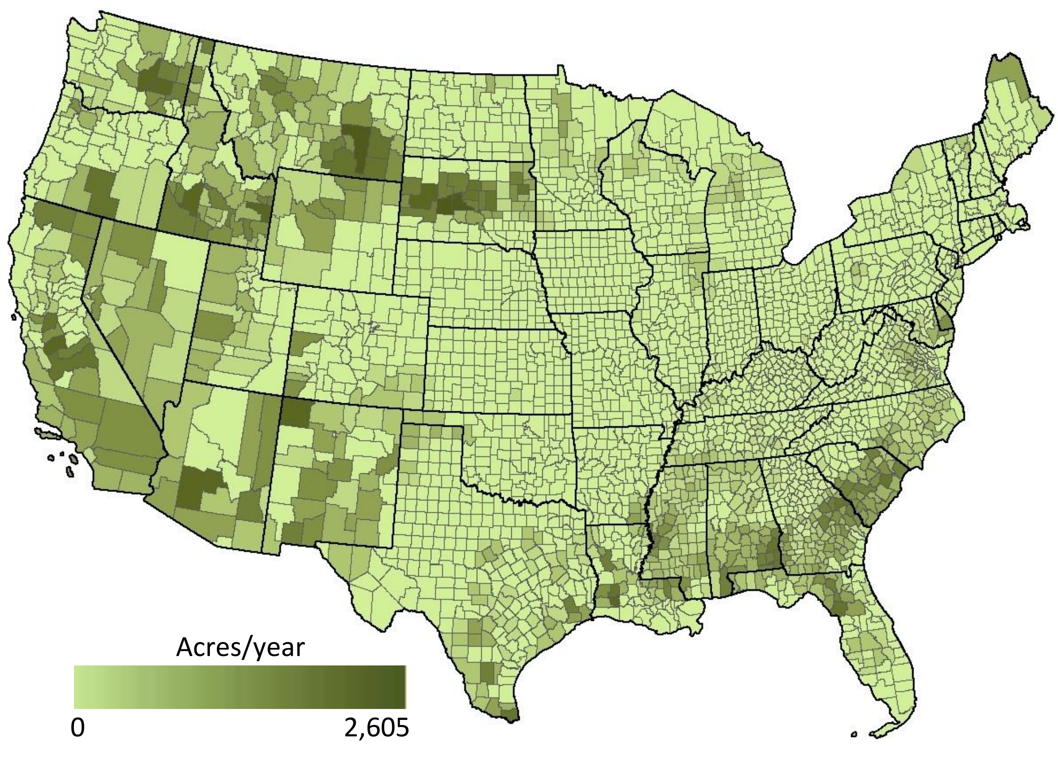
Gross "cropland" conversion rates are highly variable, with ranges across counties from 0 to 2,605 ac/yr (Table 15). While average annual conversion rates are relatively low, the large range and high standard deviation indicates that some counties are experiencing significant conversion of "cropland" to "grassland" and to "shrub/scrub." See Appendix B: Conversion Rate and Land Value Tables for state-level total net and gross conversion rates (Table 23 and Table 24).

Table 15: Characteristics of county-level NLCD cropland conversion rates, ac/yr

	Avg	Max	Min	St.Dev.
Cropland to Grassland	20.79	2,605.49	0	110.7
Cropland to Shrub/Scrub	23.82	1,773.86	0	92

“Cropland” conversion to “grassland” and “shrub/scrub” is generally concentrated in the western US and along the southeastern coasts (Figure 30). The highest conversion rates between 2001 and 2006 were scattered across the western US, with conversion rates of approximately 1,000 acres/year or more occurring in counties in South Dakota, Montana, Idaho and California. Rates of conversion from “cropland” and “grassland” are relatively small (fewer than 50 ac/yr) across much of the rest of the country, with rates of essentially zero dominating in the Northeast, and most of the Midwest and interior Southeast. The spatial distribution of “cropland” conversion to “grassland” is largely similar to the spatial distribution of “grassland” conversion to “cropland.” These similarities suggest that land-use is highly dynamic in certain areas – with many different types of conversion – and relatively static in other regions.

Figure 30: Gross conversion rates for cropland to grassland and shrub/scrub



5.2.1.3 Regional Cropland Conversion Rates

We also aggregate grassland conversion rates by IPCC climate zones (Table 10) to generate a broader perspective on conversion trends. See Figure 5 (above) for a map of climate zones.

Table 16: Gross conversion rates for cropland to grassland and shrub/scrub

IPCC Climate Zone	Cropland Area in 2001 (ac)	Conversion rate (ac/yr)	
		Cropland to Grassland	Cropland to Shrub/Scrub
Warm Temperate Moist	1,158,850,670	9,476.36	17,371.77
Warm Temperate Dry	975,708,689	7,120.94	10,924.15
Cool Temperate Moist	1,175,521,038	3,505.65	5,142.61
Cool Temperate Dry	1,340,244,285	30,680.79	16,733.32
Polar Moist	1,195	0.00	0.85
Boreal Moist	28,911	12.37	43.81
Tropical Moist	211,910,613	12,435.01	18,433.48
Tropical Dry	100,927,847	1,434.09	5,398.54

5.3 Estimated Cost of Reductions

The carbon cost assessment for CCG relies on Conservation Reserve Program (CRP) county level rental rates. Like the GRP, the CRP payments offer a useful proxy for assessing the financial incentive necessary to engender significant participation in a carbon project that converted cropland to grassland. The Farm Service Agency sets the CRP rate by multiplying the county average rental rate for dry land cropland by the grouped soil productivity factor. The soil productivity factor is derived from the Natural Resources Conservation Service (NRCS) soil survey, which shows the individual maximum soil rental rate for groups of soils. Therefore, one soil rental rate may be used for multiple soils; however, each soil grouping will have only one soil rental rate.

Using CRP rental rates and the estimated emission factors for cropland set asides, the corresponding cost per tCO_2e would range from $51.34 per tCO_2e to an effectively infinite cost where emission factors are negative (see Table 17). The high range of prices is indicative of the substantial variation in CRP rental rates, which range from $14 to $400 per acre. The low level rates for the majority of the climate zones indicate that double-digit carbon prices would be required to incentivize significant landowner participation in CCG projects.

Table 17: Estimated carbon cost range for CCG

IPCC Climate Zone	**Est. EF Range*** (tCO_2e\ac\yr)	**Avg. CRP Rate** ($\ac)	**Est. Cost Range** ($/$tCO_2e$)	**Est. Mean Cost** ($/$tCO_2e$)
Cool Temperate Dry	(0.06) – 0.40	$47.15	116.86 - ∞**	274.13
Cool Temperate Moist	(0.22) – 0.65	$91.70	140.21 - ∞	451.72
Warm Temperate Dry	0.23 – 0.60	$38.75	64.30 – 166.07	105.30
Warm Temperate Moist	(0.17) – 0.51	$70.50	136.98 - ∞	446.20
Tropical Dry	0.27 – 0.71	$36.25	51.34 – 132.78	75.36
Tropical Moist	(0.15) – 0.69	$43.05	62.15 - ∞	239.17

**Emission Factors derived as discussed in Section 3.2 above. These factors are on the higher end of similar available research, but are generally consistent[10].*
***In cases where the minimum emission factor is negative (as indicated by parentheses), there is no cost of carbon which would generate revenue, hence the use of the "∞" symbol.*

5.4 Potential Reduction Opportunity

There are important distinctions to be drawn between the estimates of AGC and CCG technical emission reduction potential. AGC, by its nature, is limited by the total amount of grassland to cropland conversion observed. In contrast, CCG activities are only technically limited by the area of cropland considered eligible for conversion into grassland.

Although the limitation of CCG activities to marginal or degraded cropland is not strictly necessary to achieve emission reductions, the Reserve has stated an interest in considering such a criterion, and the estimates of technical potential below reflect two potential cutoffs for degraded cropland (see Table 18). Using NRCS Land Capability Classes to define degraded land, two estimates are presented for classes greater than 5 and for classes greater than 6. These two estimates create a range of technical potential for CCG activities ranging from -8.7 million to 77 million tCO_2e per year.

The dramatically larger scale for potential emission reductions from CCG activities relates primarily to the land base that is classified as degraded in the US. Using Land Capability Classes of 5 and 6 as cutoffs for a definition of degraded lands results in the classification of 45% and 44% of cultivated cropland in the US as degraded, respectively. For contrast, CCG technical potential is estimated for 136-140 million acres of US cropland, while AGC technical potential is estimated only for 180 thousand acres of US grassland and shrubland.

CCG technical potential follows a distinct geographic distribution from that found for AGC activities (see Figure 31). CCG technical potential appears to be distributed into several discrete zones. Considering the low end emission factor estimates, CCG hotspots appear from the Texas and Oklahoma panhandles

[10] Paustian et al. (1995) observed a SOC gain of 0.9-3.6 tCO_2e/ac/yr for CRP. In estimating the total mitigation potential for US grazing lands, Follett et al. (2001) used an annual sequestration rate for converting cropland to pasture at 0.59 to 1.8 tCO_2e/ac/yr (400 to 1200 kg C/ha/yr) and enrolling land to CRP to sequester 0.9 to 1.3 tCO_2e/ac/yr (600 to 900 kg C/ha/yr).

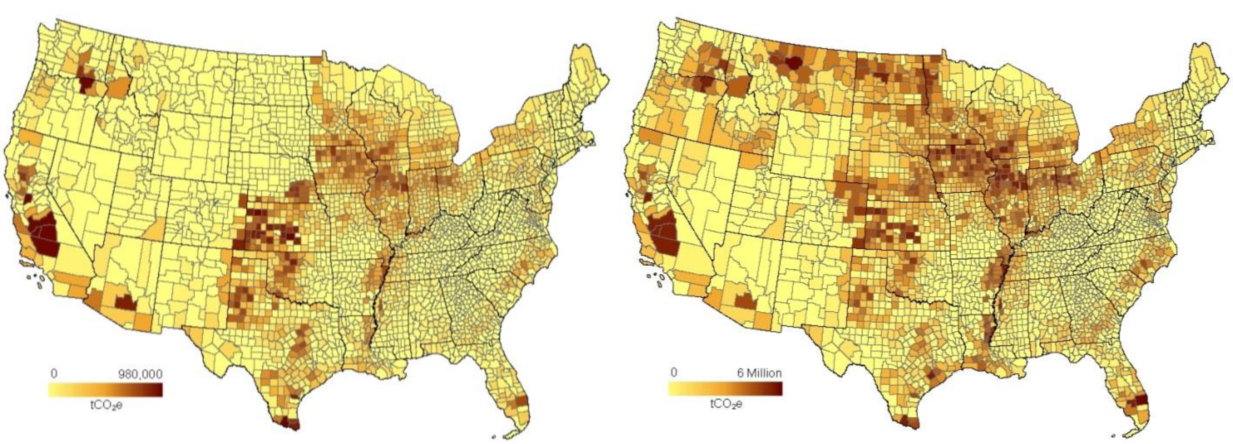
up through western Kansas. Northern states bordering the Mississippi River also appear prime targets for CCG activities. When the high-end estimate emission factors are applied, the Cool Temperate Dry Zone moves from -0.06 to 0.40 tCO_2e/ac/yr , and hotspots appear across northern Montana and North Dakota, with a few additional clusters appearing in eastern Oregon and Washington. In both high-and low-end emission factor estimates, the southern end of California's Central Valley also appears to have substantial CCG potential stretching from Kern through Fresno counties.

Table 18: Estimated emission reductions for CCG activities with two cut-offs for degraded lands

	Land Capability Classes 5+			Land Capability Classes 6+		
IPCC Climate Zone	**Acres** *thousands*	**Low** *thousand tCO_2e\yr*	**High**	**Acres** *thousands*	**Low** *thousand tCO_2e\yr*	**High**
Cool Temperate Dry	38,520	(2,319)	15,542	37,235	(2,242)	15,023
Cool Temperate Moist	34,977	(7.737)	22,875	33,964	(7,513)	22,212
Warm Temperate Dry	27,454	6,406	16,546	26,450	6,172	15,941
Warm Temperate Moist	28,789	(4,984)	14,817	28,317	(4,903)	14,574
Tropical Dry	3,498	955	2,470	3,507	957	2,476
Tropical Moist	6,915	(1,054)	4,790	6,778	(1,033)	4,695
Total	**140,154**	**(8,733)**	**77,039**	**136,251**	**(8,561)**	**74,921**

Notes: Where Land Capability Classes were provided for irrigated and non-irrigated soils, the lower of the two values was used, so all acres shown in this table had their highest land capability classification of 5+ or 6+. See Section 6.3.1.2.1 for a description of each class.

Figure 31: Geographic distribution of estimated technical potential for CCG activities



Note: The maps on left and right present the low- and high-end estimates for annual emission reduction potential *for each county, respectively. [The magnitude of these reductions, as labeled, is incorrect. However, the geographic distribution of the potential reductions is still accurate.]*

6 Project Protocol Considerations

In light of the potential for substantial emissions reductions for AGC and CCG project activities across the continental United States, this section reviews several important topics that would be considered as part of the process to develop a carbon offset protocol for AGC and CCG projects. This section moves through the primary carbon accounting and policy considerations, including the GHG accounting boundaries, additionality criteria, reversals, leakage, co-benefits, and finally, monitoring and verification issues.

6.1 Sources Sinks and Reservoirs

Accounting for the total net changes in GHG emissions will require comprehensive accounting across all sources, sinks, and reservoirs (SSR) associated with the implementation of the project. In contrast to the estimates of technical potential provided above, which considered changes in SOC stocks only, an offset protocol for AGC and CCG projects would need to include a broader set of GHG SSRs. In order to calculate these emissions across various projects, GHG assessment boundaries are defined for the specific projects. Importantly, the SSRs associated with the implementation of a project are not necessarily geographically local to the project area. This is particularly true when considering leakage, the potential of GHG emissions to be created in areas in response to baseline practices being transferred to another region, and therefore not resulting in an overall reduction in GHGs (just a displacement to another region). Within each project, SSRs are considered if they are expected to have a significant influence on the overall net GHG emissions associated with project activities. Not all emission sources are included and these emission sources can be accounted for within the uncertainty boundaries set for each projects emissions. Emissions will be excluded from a project in general if they are:

- Excluding the SSR is conservative, i.e., exclusion would result in a smaller volume of the total net GHG reductions)
- The total increase in GHGs from excluded SSRs is likely to be less than five percent of the total GHG reductions achieved by the project.

With regards to potential AGC and/or CCG protocol(s), the Reserve may consider defining a physical boundary for the project. The Reserve may find that GHG accounting across these climates and soils is complicated and may decide to limit the protocol to certain regions, states or land-use types that encompass the majority of sensitive grassland systems in the US. For example, excluding rangelands in Alaska may be necessary, as there is not currently NRI data on Alaska, there are few if any grassland regions in the area, and GHG emissions accounting associated with permafrost and tundra may complicate calculations given the potential for CH_4 emissions from these systems (Gershenson, et al., 2011).

Leakage accounting for unintended emissions that may result from the project will be necessary to consider. With this protocol in particular, leakage may result from the conversion of grasslands or other land cover types to accommodate forgone crop production, particularly as world food demand increases. A standardized leakage factor may be applied to the GHG calculations to account for this potential.

The following tables consider the likely up-stream, on-site, and down-stream emissions that should be considered in the quantification of net GHG emissions. Up-stream emissions include those that result from changes to input use or fuel use. These may include the associated emissions (storage, transportation, production) of farm inputs like manure or fertilizers. On-site emissions are those that result at the farm level in conjunction with the implemented practice. These may include emissions that are avoided through a reduction of fertilizers or inputs or emissions that occur such as the production of enteric fermentation emissions from livestock. Down-stream or off-site emissions result from indirect or past farm-gate impacts as a result of the change of practice.

Table 19: Analysis of sources, sinks, and reservoirs (SSRs) for AGC projects

Up-stream	On-site	Down-stream
• Avoided emissions from transportation (production?) of lime, inorganic N or manure to the field (avoided conversion to cropland) • GHG emissions from storing manure or any other off-site manure management (avoided conversion from grazed land) • Avoided GHGs of other inputs such as fertilizers and pesticides (avoided conversion to cropland) • Changes in emissions from fuel (avoided conversion to any land use)	• Avoided N_2O emissions from fertilizer and manure application (avoided conversion to cropland) • Avoided emissions from soil carbon changes (avoided conversion to cropland and development) • Avoided emissions from soil carbon due to tillage (avoided conversion to cropland or development) • Changes in emissions from machinery use (avoided conversion to cropland or development) • Livestock enteric fermentation (avoided conversion from grazing land) • On farm manure use (avoided conversion from grazed land) • Changes in above and below ground biomass (avoided conversion to cropland or development)	• Avoided emissions from leaching and run-off of applied nitrogen, followed by denitrification into N_2O (avoided conversion to cropland) • Avoided GHG emissions from indirect land-use changes (avoided conversion to any land use) • Emissions from storing, handling and transporting of manure

6.1.1 Sources, Sinks, and Reservoirs for AGC

The AGC project activities require consideration of upstream impacts, which will also depend upon the conversion threat. If the grassland is under threat for conversion to cropland, emission must be considered that are associated with on farm inputs, manure storage and process fuel or transport emissions of manure and inputs. At the farm level, avoided emissions from fertilizer application associated with cropland conversion, tillage, machinery and other cropland inputs should also be considered. If the grassland is under threat for conversion to development or other land use types, emissions associated with development may be included. If pasturelands are to be included within the defined project activities on farm inputs associated with pasture (fertilization, irrigation, lime, herbicides) should be included. Soil carbon benefits associated with grassland preservation should be considered. Downstream impacts from AGC include potential effects from indirect land use changes off site (i.e., leakage), avoided emissions from leaching (dependent on soil type and precipitation in various rangeland ecosystems), and off-site manure storage and processing. Non-CO_2 emissions associated with livestock including enteric fermentation and N_2O emissions associated with manures should be considered and are discussed more thoroughly in Section 3.2.

Table 20: Analysis of sources, sinks, and reservoirs (SSRs) for CCG projects

Up-stream	On-site	Down-stream
• Transport of labor to farm to restore the grassland • Emissions from transportation of seed and input materials to establish grassland • Production of inputs? (if pasture-based systems including fertilization and other inputs may be relevant)	• N_2O emissions from fertilizer and manures (avoided and realized) • Increases in soil carbon content • Changes in emissions from establishment practices (organic amendments, herbicides, tilling) • Avoided emissions from tillage/ soil carbon • Irrigation Emissions (avoided and realized) • Pasture inputs (if relevant) • Changes in above and below ground biomass • Fire emissions where prescribed fire or natural fires occur	• GHG emissions from indirect land-use changes • Emissions from storing, handling and transporting manure • Avoided emissions from leaching and run-off of applied nitrogen, followed by denitrification into N_2O

6.1.2 Sources, Sinks, and Reservoirs for CCG

The CCG project activities will need to consider a number of practices that may vary significantly depending on the site for grassland establishment. Upstream impacts will largely be from inputs including grass seed, herbicides or other inputs used in the establishment of grass species. On farm SSRs may be notably different depending on regional characteristics. As previously discussed in Section 5.1.1, a variety of practices can be utilized to help establish a grass species including organic amendments, herbicides, tillage, or irrigation. Nevertheless, it is likely that baseline emissions of N_2O can be reduced because there are potentially fewer nitrogen-based fertilizer applications and therefore less nitrogen to volatilize. However, project N_2O emissions may increase due to greater soil compaction and subsequent changes in soil moisture dynamics in intensively grazed grasslands (Bhandral, et al., 2007). The transition from cropland to pasture may be accompanied by one-time increases in fuel consumption for grass seeding, as well as other preparation activities that may be required to remediate marginal croplands in order to sustain healthy grasslands.

For CCG projects, the introduction of grazing animals in the pasture system may also cause increased N_2O emissions due to manure and urine inputs. In addition, when the grazing animals are ruminants, CH_4 emissions will likely increase relative to the baseline, i.e., a cropping system. Nevertheless, CH_4 emissions from ruminants will vary due to diet, variety of cattle, intensity of grazing, and dietary supplements. Enteric fermentation and N_2O emission changes associated with fertilization (organic and inorganic) will account for changes to livestock emissions. While some pastures are irrigated, it is unlikely that a non-irrigated cropland would be converted to an irrigated pasture since irrigation decisions are driven by cost and climate, which should remain largely constant. However, if pasture systems that include substantial nutrient inputs and irrigation are included within the scope of the protocol the Reserve will need to consider how to quantify native grassland systems compared to managed pasture systems in a way that accurately accounts for their varied inputs and subsequent emissions. There may also be increased energy required to house animals and GHG emissions from manure management at the housing location.

Changes in off-site emissions include possible reductions in the use of chemical inputs for cropping, though this may be accompanied by increases in input-related GHGs associated with livestock, such as those used for feed and transportation, and for the establishment of grasses. While some intensively grazed pastures are fertilized, the conversion from cropland to pasture may reduce fertilizer use by 25% (Eagle, et al., 2012). Additionally, there may be increased upstream emissions due to seed production (critical in the case of alfalfa), when converting marginal croplands to pasture. There may also be changes in off-site downstream emissions, including emissions from cattle in later production and processing stages (Pitesky, et al., 2009) Changes in on-farm below and above ground biomass will also need to be accounted for in order to determine SOC storage rates. These are treated as separate and unrelated pools, even though they are highly correlated and often completely related for the purpose of carbon accounting through the use of a shoot-to-root ratio. Off-site or downstream emissions may result from indirect land use changes (i.e., leakage), emissions associated with storing, handling and transporting manures or the avoided emissions from nitrogen leaching.

6.1.3 GHG Emissions Associated with Livestock

Livestock, by definition, are generally an inherent part of rangeland systems. Their presence can also have implications for the potential of AGC and CCG activities to provide an overall net reduction in GHGs. According to the US Environmental Protection Agency (EPA), in 2010 CH_4 emissions from enteric fermentation and manure management were respectively 21% and 8% of all human-caused CH_4 emissions. Within this, beef dairy cattle were the largest emitters of CH_4 from enteric fermentation, accounting for 72% of all enteric emissions. However, soil management associated with cropping practices and fertilizer application were the largest source of N_2O, accounting for 68% of total N_2O emissions in the United States. N_2O emissions from manure management accounted for only about 4% of agricultural emissions in the United States.

Within the proposed project activities, livestock emissions from manure management and enteric fermentation should be accounted for (See Section 6.1 above for a complete list of potential emissions to consider). In AGC activities where livestock are present, livestock will remain on the grassland, likely not creating significant changes in net livestock emissions in the with-project scenario. However, these livestock would generally be absent from the baseline AGC scenario (i.e., conversion to cropland) and the difference between these livestock emission in the baseline and with-project case would need to be accounted for. Furthermore, carbon sequestration can provide a net emissions benefit. The avoidance of carbon losses from tillage and increased emissions from N_2O can be expected in association with AGC. For CCG, the inclusion of livestock non-CO_2 GHGs may be more complicated to address. Since livestock would be a new addition to the landscape over previous cropping strategies, enteric fermentation and manure additions (not used or present in cropping strategies) would be considered new sources of potential emissions. To the extent that livestock emissions are less than the negative emissions associated with carbon storage in the grassland system, a net reduction in emissions can be achieved.

6.1.3.1 Measurement of Livestock Emissions

A variety of methods exist to directly measure livestock emissions, particularly from enteric fermentation. A complete review of this topic and available methodologies is provided by the Technical Working Group on Agricultural GHGs (T-AGG, 2012). Chamber measurements or enclosures can estimate enteric fermentation emissions accurately (Johnson & Johnson, 1995). Gas tracer methods, most commonly using sulfur hexafluoride (SF_6), which is assumed to be the same as CH_4 emissions, can be used to estimate emissions. This method can be especially useful for rangeland cattle since it allows for cattle to be estimated while still grazing, while chamber methods would require cattle to be housed or stationary for a period of time. Additional micrometeorological methods can measure the flux of gas in the atmosphere relative to animal emission fluxes. However, given the large expense associated with many of these direct measurement options (T-AGG, 2012), it is unlikely that verifiers and the Reserve will be able to undertake direct measurements on all farms.

6.1.3.2 Modeling Non-CO_2 Livestock Emissions

Modeling and quantifying the GHG impacts from livestock can be achieved through several methodologies. The United States EPA uses the Cattle Enteric Fermentation Model (CEFM), which estimates cattle CH_4 emissions from enteric fermentation using livestock population, feeding practices and production characteristics. The CEFM is based on IPCC Good Practice Guidance Tier 2 approach. Uncertainty estimates associated with these estimates are ±10% or lower (Mangino, et al., 2003).

In addition to national inventories like the CEFM, mechanistic models use detailed dietary inputs and base CH_4 emissions to estimate livestock enteric fermentation emissions. This approach could allow for the Reserve to estimate livestock enteric fermentation emissions based on different dominant species present in various rangeland systems. MOLLY (Baldwin, 1995) and COWPOLL (Dijkstra, et al., 1992) are the most commonly used mechanistic models. Thermodynamic models are an emerging concept (Kohn & Kim, 2011) while whole-farm models can input data based on a variety of management decisions. The Cool Farm Tool, Agricultural and Land Use National GHG Inventory, and the USDA's Integrated Farm Systems Model are examples of this strategy.

Reviews by Kebreab et al. (2006; 2008) have examined how various methods can effectively predict livestock emissions; though these reviews have looked at dairy cattle and feedlot cattle, with less attention to free ranging systems. Tier 1 emissions models were acceptable for general CH_4 inventories though Tier 2 models can provide better quality predictions when feed intake data are available For mechanistic models, MOLLY has had the lowest error for feedlot cattle systems (Kebreab, et al., 2006).

6.1.4 Carbon Sequestration and Grazing

Grazing practices, both the intensity and frequency, are generally considered to be the primary factors affecting SOC storage in grasslands, but past research has often reported inconsistent results (Derner & Schuman, 2007). Grazing directly and indirectly affects SOC through various mechanisms, the individual effects of which are easily confounded with the interaction of other factors, i.e., climate and management, and these interactions are generally poorly understood (Pineiro, et al., 2010). In a survey of the international literature, Pineiro et al. (2010) found three common generalizations of how grazing affects SOC:

1) Root contents, which are a primary control of SOC formation, were higher in grazed systems than in ungrazed systems in the wettest and driest sites but lower in the intermediate sites (precipitation levels of approximately 400mm to 850mm);
2) Soil organic matter C:N ratios consistently increased under grazing conditions; and
3) Bulk density either increased or did not change in grazed sites. Given that all three factors operate simultaneously, the authors expect SOC to decrease under grazing in sites located with a mean annual precipitation range of 400-850mm, as all factors in this range would decrease SOC stocks- root biomass deceases, soil compaction increases, and N limitation increases

Long-term studies in North America have found generally similar results, with grazing generally having a positive effect on SOC in most rangeland dominant geographies and a precipitation gradient affecting SOC sequestration rates under various grazing treatments (Derner & Schuman, 2007).

6.1.4.1 Grazing-Management Effects on Soil Carbon Stocks

The response of rangeland SOC to grazing intensity and stocking rate is variable (Derner & Jin, 2012). Grazing intensity, over long time periods, does not appear to have an effect on SOC stocks in the Great Plains. Changes in biological carbon sequestration observed following changes in stocking rates is likely the result of grazing-induced changes in the plant community composition and not directly driven by the stocking rate (Derner & Schuman, 2007). For example, moderate and heavy grazing over 81 years in a northern mixed-prairie near Mandan, North Dakota, increased SOC by 19% and 34% in the surface 5cm relative to a non-grazed exclosure (Wienhold, et al., 2001). At a nearby study site, Liebig et al (2010) also found moderate and heavily grazed native pastures to be a net SOC sink, increasing 16.0% and 15.4% over the surface 60cm in a 44-year period. For the moderately grazed pasture (0.16 animals per acre), an annual accrual of 0.58 ± 0.07 tCO_2e/ac/yr, was detected. Similarly, grazing strategy did not affect SOC sequestration over an 11-year period in a northern mixed-grass prairie near Cheyenne, Wyoming, as no differences were evident among short-duration rotational grazing, rotationally deferred grazing, and continuous season-long grazing at heavy stocking rates (Manley, et al., 1995). In direct comparisons to adjacent non-grazed exclosures, both moderate and heavy stocking rates in shortgrass steppe and light-to heavy stocking rates in mixed-grass prairie increased SOC in the surface 30cm (Derner & Schuman, 2007). Field CO_2 flux data has likewise observed a net sequestration effect for moderate grazing in northern mixed-grass prairie (Frank, 2004). Across ecosystems, grazing has been found to have a positive- to no effect on SOC sequestration at study sites in Colorado, Wyoming, North Dakota, and Oklahoma, including the Shortgrass prairie, Northern mixed-grass prairie, and Southern mixed-grass prairie ecoregions (Derner & Schuman, 2007). Outside of the plains, grazing has been found to have no adverse effect on California annual grasslands, although there has been limited research to date (Silver, et al., 2010).

6.1.4.2 Grazing and Climate Interactions Affect Soil Carbon Sequestration

Specific to North American locations, Derner et al. (2006) found a precipitation effect similar to that of Pineiro et al (2010), with grazing increasing SOC in North American dry shortgrass ecosystems, but decreasing it in more humid mid- and tallgrass prairie ecosystems. Although grazing sites had lower overall system carbon storage than ungrazed sites at the mid- and tall- grass locations (midgrass: 118 vs. 129 tCO_2e/ac; tallgrass: 123 vs. 134 tCO_2e/ac), both locations had nearly twice as much system carbon as the short-grass location (60 vs. 48 tCO_2e/ac) (Derner, et al., 2006). A precipitation threshold for the Great Plains, where grazing has been observed to change from having a positive SOC effect to a negative one, has been estimated at precipitation levels greater than 440mm for the 0 to 10 cm of the soil surface, and at 600mm for the 0 to 30 cm depth (Derner & Schuman, 2007). Grazing effects during drought conditions, regardless of regional precipitation averages, may also cause a loss of SOC (Ingram, et al., 2008).

6.2 Soil Carbon Stock Measurement and Stratification

6.2.1 Stratification

For larger and aggregated projects, SOC stocks are likely to be highly variable across the landscape. \ To reduce the sampling intensity required to achieve desired precision targets, large or heterogeneous project areas should be stratified into smaller units or strata that can be considered homogeneous for carbon accounting. The protocol must include procedures to conduct stratification and parameters by which the area is stratified. Typical parameters that may be used to stratify the project area are:

- Soil unit and components as available from the SSURGO database
- Common historical rangeland or cropland management practices, for AGC and CCG projects, respectively
- Common future rangeland management practices or expected cropland practices after the start of the project or under the baseline conditions, for CCG and AGC projects, respectively
- Presence of special status soils (e.g., serpentine soils, histosols, etc.)
- Ecological site (soil texture class, aspect, slope, hydrology, climate, or plant communities)
- Degradation status
- Differences in legally binding requirements affecting management of the project area (e.g., easement status of land, ownership)

A stratum is then defined as the largest possible area in which each of the variables above is identical. The location and size of strata can be calculated using a GIS analysis based on the SSURGO database, a Digital Elevation Model (DEM), and some expert knowledge of site productivity and conditions. It is known that the SSURGO database can be unreliable. Therefore, some cross-checking of the boundaries of the soil units acquired from SSURGO through a field visit may be appropriate. Other protocols have relied on NRCS staff or certified rangeland experts to conduct cross-checking and stratification, given that a fully standardized approach to stratification is often challenging. The Reserve may consider a hybrid approach where potential stratification variables are included in the protocol, but could be overridden by local experts according to well-specified rules to ensure consistency.

It may make sense to set the soil sampling density (i.e., number of samples per project or area) as a fixed number of samples per stratum instead as a fixed number of samples per area or per project.

6.2.2 Carbon Accounting Procedures and Soil Carbon Stock Measurements

Soil carbon stock measurements are expensive and time consuming. Therefore, soil carbon stock sampling requirements should always be evaluated from a cost-benefit standpoint.

Carbon accounting that is solely based on field measurements will not be cost effective for AGC and CCG projects. Empirical or biogeochemical models are appropriate tools to quantify carbon stocks and carbon stock changes and their use should be considered by the LUC protocol(s). Examples of such tools include IPCC factors, regional factors, or biogeochemical process models such as CENTURY, or DNDC. A model like CENTURY allows specifying the soil type and the historical management and will predict what the current-day SOC stocks will be. Likewise, the IPCC has a procedure to estimate standing SOC stocks based on a set of factors that represent climate, soils and management. However, we do recommend that the protocol requires some SOC sampling to, at least, confirm the accuracy of values of empirical or biogeochemical models used in determining the baseline SOC stocks.

In addition, the protocol should include some flexibility regarding soil sampling to allow project participants to receive a greater volume of credits if a more intense sampling strategy is followed. For example, the protocol could include a minimum required soil sampling depth but incentivize sampling to greater depths or greater spatial density of sampling points. We will limit the discussion of soil sampling requirements to select issues that differ across established soil sampling protocols and that the LUC protocol(s) should specify. These issues include:

- The minimal depth to which one should sample. Whereas sampling to greater depths will yield greater carbon stocks, the difference in carbon stocks between grassland and cropland soils decreases with depth. Therefore, sampling to more shallow depths is conservative and sampling to greater depths leads to diminished marginal returns. However, the exact depth that maximizes returns will be dependent on the soil type and land use, and should, therefore, be left to the project developer to decide. Often, 20 or 30-cm is a standard minimal depth prescribed by sampling protocols, while some procedures specify to sample up to 100 cm.
- All soil carbon measurements should be accompanied with a robust soil bulk density measurement to be able to calculate carbon stocks. Often, the variability in SOC stocks is more

impacted by the variability in the bulk density than there is in the carbon concentration of a soil. Therefore, sufficient attention must go to bulk density measurements.

- Procedures should indicate what to do in case of rocks, stones, or visible plant debris. There are several ways of correcting for these, but it is important that the way this is treated is fixed to keep samples standardized.
- Procedures should also specify whether soil samples should be stored for a certain time

6.3 Potential for Standardized Additionality Testing

6.3.1 Project Eligibility Criteria

Developing a standardized additionality test would require the Reserve to examine several factors across economic, social, and political boundaries. First and foremost, the Reserve will need to establish that eligible land is indeed legally convertible to grassland or has the biophysical capacity to be cropped. The Reserve will also need to establish standard acceptable practices that are eligible for this protocol. In particular, determining whether "cultural management" as defined by the USDA qualifies land for participation will be relevant.

In general, the project area should be demonstrably "suitable for cropping". Areas that have been classified as "not suitable for croplands" by a designated authority should be made ineligible to ensure that "avoided conversion" projects are not being planned in areas that would not otherwise be converted under the baseline scenario. In the United States, in absence of a classification system adopted by a state or other designated authority, the USDA Land Capability Classification (LCC) system could be used. The current LCC includes eight classes of land designated by Roman numerals I thru VIII. The first four classes are arable land suitable for cropland in which the limitations on their use and the necessity for conservation measures and careful management increase from I thru IV. Additional prescriptive criteria may be added. Such additional criteria may be either direct (e.g., maximal slope grade or minimal availability of water), or indirect (a grassland with similar characteristics but at a different location has been effectively converted).

6.3.1.1 Potential Exclusion of Soil Types to Simplify Accounting

The Reserve may also consider limiting the eligibility of AGC and CCG project activities to particular soil types and related environmental conditions to ensure emissions accounting is consistent and accurate. For example, saturated soils and soils high in organic matter found in peatlands and wetlands can have significantly different GHG emissions profiles compared to other soil types. Anaerobic conditions associated with saturated soils with high soil water content are known to have a positive effect on methane (CH_4) production (Neff, et al., 1994). These soils when coupled with fertilization can also show significant increases in nitrous oxide (N_2O) emissions (Neff, et al., 1994). Organic matter is also a good predictor of CH_4 production in peatlands (also often saturated soils). High CH_4 emissions have been associated with high levels of organic matter in comparison to low-organic matter soils (Schimel, 1995) and where organic matter is labile (Yavitt & Lang, 1990). Peat grasslands characterized by wet fields had the highest CH_4 production rates within a grassland system where organic matter contents > 40% (Best & Jacobs, 1997). Plant biomass has also been a predictor for CH_4 emissions in grassland ecosystems with above ground biomass being an indicator for higher levels of CH_4 as organic matter from plants may serve as a substrate for methanogens (Chanton, et al., 1995; Van den Pol-van Dasselaar, et al., 1999).

There are several grassland biomes that have soils that can create significant GHG emissions that may be challenging to quantify. The Reserve may consider excluding certain systems including peatlands, wet meadows (or vernal pools), and tundra and permafrost systems to simplify carbon accounting. If such systems, which are typically known for having saturated soils that may increase GHG emissions, remain included in the protocol, it will be necessary to account for the potentially excessive CH_4 emissions coming from these systems and will have to be decided by the grassland workgroup.

The presence of vernal pools may prove to be more difficult to consider, since they are seasonal in nature and can be quite small. In the Northeast, many vernal pools are present in woodland and forest areas (Morgan & Calhoun, 2012); however, grasslands, particularly in California, are also commonly known to have vernal pools (US EPA, 2012a). Temporally, vernal pools can last only one week or often for several months before they dry up in the summer (Keeler-Wolf, et al., 1998), further complicating emissions accounting. Several databases across the United States, usually at a watershed or state level exist to trace vernal pools (Western Pennsylvana Conservancy, 2012).

6.3.1.2 Limiting CCG Project Activities to Degraded and Marginal Land

In the RFP for this Issue Paper, the Reserve indicated their consideration to potentially limit CCG activities only to marginal or degraded croplands within the United States. This limitation would ideally minimize the potential adverse effects associated with removing productive cropland from cultivation (leakage, food security, etc.). In general, degradation is primarily used to refer to a reduction in productivity of land or soil. The challenge with this definition is the subjective and relative nature of the reduction in productivity. Specifically, temporal and spatial scales, location, and methodologies must be set to make a judgment that the productivity of land is reduced. Several options exist for defining marginal or degraded cropland, including the USDA Erodibility Index, CDM policies for Afforestation and Reforestation Projects, and the National Resources Inventory among others that are reviewed here.

6.3.1.2.1 NRCS Land Capability Classes

In its national soil survey for the US, the NRCS utilizes a land capability classification system to categorize the capacity for soil units to support the production of cultivated crops over time. The *National Soil Survey Handbook*(NRCS, 2012c) describes eight land capability classes:

- Class I (1) soils have slight limitations that restrict their use.
- Class II (2) soils have moderate limitations that reduce the choice of plants or require moderate conservation practices.
- Class III (3) soils have severe limitations that reduce the choice of plants or require special conservation practices, or both.
- Class IV (4) soils have very severe limitations that restrict the choice of plants or require very careful management, or both.
- Class V (5) soils have little or no hazard of erosion but have other limitations, impractical to remove, that limit their use mainly to pasture, range, forestland, or wildlife food and cover.

- Class VI (6) soils have severe limitations that make them generally unsuited to cultivation and that limit their use mainly to pasture, range, forestland, or wildlife food and cover.
- Class VII (7) soils have very severe limitations that make them unsuited to cultivation and that restrict their use mainly to grazing, forestland, or wildlife.
- Class VIII (8) soils and miscellaneous areas have limitations that preclude their use for commercial plant production and limit their use to recreation, wildlife, or water supply or for esthetic purposes.

A national map of soils data, including land capability classes, called the Digital General Soil Map is offered by the NRCS (NRCS, 2006).

6.3.1.2.2 USDA Erodibility Index

The USDA Erodibility Index combines a variety of factors to determine the overall vulnerability of a soil to erosion and degradation. Potential erodibility for sheet and rill erosion and wind erosion are calculated separately. The USDA determines highly erodible land based on definitions set forth in the National Food Security Act Manual in part 511.

Potential Erosion is calculated as following:

$$PE_{SR} = R \times K \times LS$$

where:

PE_{SR}	Potential sheet and rill erosion
R	Rainfall and runoff
K	Susceptibility of the soil to water erosion
LS	Combined effects of slope length and steepness

$$PE_W = C \times I$$

where:

PE_W	Potential wind erosion
C	Climatic characterization of wind speed and surface soil moisture expressed as a percentage
I	Susceptibility of the soil to wind erosion

The complete Erodibility Index (EI) is calculated by dividing the potential erodibility (for sheet and rill and wind erosion separately) by the soil loss tolerance value (NRCS, 2004).

Though the USDA Erodibility Index provides strong biophysical grounding for determining the potential erodibility of a land, it may not be a complete measure to understand marginal and degraded lands. Since the index does include measures of socio-economic or political factors that can drive degradation it may miss components of a more holistic approach to defining marginal lands. For example, some human or ecological influences can degrade cropland irrespective of their potential for erosion (i.e., increases in salinity). These croplands may still be considered marginal or degraded, but for reasons beyond erosion potential.

In 2012, the USDA Conservation Reserve Program (CRP) announced a new program for highly erodible land for landowners to "plant long-term, resource conserving covers to improve the quality of water, control soil erosion, and enhance wildlife habitat". The CRP defines highly erodible land as having an Erodibility Index of 20 or greater (USDA FSA, 2012b). The CRP process for determining whether land is highly erodible seems to be a straightforward process whereby farmers meet with local Farm Service Agency officers who use the NRCS soil survey data along with GIS to determine what lands are eligible. If the Reserve is able to use similar data, the Erodibility Index could serve as one component of a more comprehensive baseline.

6.3.1.2.3 CDM Tool for the Identification of Marginal or Degraded Lands

The Clean Development Mechanism (CDM) provides a tool to distinguish "degraded" and "degrading" lands. The CDM tool is a helpful guideline measure to consider when determining degradation across landscapes. The method proposed by the CDM includes a two-step approach. The first stage involves an initial screening to determine whether land within a given place has been described as degraded under various classification systems. A second investigative stage is necessary for project lands that have no documented classification of degradation. In this case, a direct visual inspection or a comparison of potential project lands to other degraded lands with similar ecological and social influences shall be conducted.

The CDM tool suggests that the presence of one of the following is substantial to demonstrate that a land is degraded and/or degrading:

1. *Documented evidence that the area is classified as degraded under legitimate land classification systems or peer-reviewed studies from the last ten years. If the study is older than ten years then evidence shall be provided that details that the degradation drivers and pressures that caused the degradation are still present.*
2. *Demonstrate that the proposed land have similar conditions and socio-economic pressures as degraded lands elsewhere.*
3. *Direct evidence from the site visually gathered to assess the condition of the land. Degraded and/or degrading lands will have at least one of the following:*
    a. *The severity and extent of soil compaction and soil erosion, as determined by the presence of: reductions in topsoil depth (as shown by root exposure, presence of pedestals; exposed sub-soil horizons or armour layers); gully, sheet or rill erosion, landslides, or other forms of mass-movement erosion;*
    b. *Decline in organic matter content and/or recession of vegetation cover as shown by reduction in plant cover or productivity due to overgrazing or other land management practices, thinning of topsoil organic layer, scarcity of topsoil litter and debris (GPS and photo evidence should be provided);*
    c. *Presence of plant species locally known to be related to the condition of degradation of the land or field/lab tests showing nutrient depletion (e.g., reduced growth, leaf*

loss, desiccation, leaf chlorosis), salinity or alkalinity, toxic compounds and heavy metals;

d. *A reduction in plant cover or productivity due to overgrazing or other land management practices* (CDM EB, 2008).

The use of this CDM tool for CCG project activities may reasonably ensure projects are taking place on degraded land. However, it will be important to match up the tool's descriptive characteristics with existing databases such as the National Resources Inventory or the National Land Cover Database (both described in Appendix A: Land Type Definitions). Without quantitative data to determine how these characteristics are distributed throughout the United States or through target regions, a much greater investment in verification may be necessary. If existing databases are insufficient to assist in determining degradation, it may be possible to include a degraded lands assessment via site visit by verification agents.

6.3.1.2.4 National Resources Inventory

The National Resources Inventory dataset could also be utilized to help determine degraded and marginal land status for CCG project activities. NRI measures total state land cover by class and subclass and provides specific data for cropland and erosion by state and land use type (See Table 22 in Appendix A: Land Type Definitions). According to the NRI, an Erodibility Index of 8 or above is considered highly erodible land, in contrast to the recent CRP classification that highly erodible land is 20 or higher. In addition the NRI can determine national land capability classifications, which group soils based on their potential productivity as cropland or pastureland. Capability classes range from 1 to 8 and can indicate systems that have limited potential for agriculture based on issues related to erosion, wetness, root zone limitations and climatic limitations (NRCS, 2009b). This set of measures may reasonably be used to confirm the status of marginal and/or degraded lands with limited agricultural options.

6.3.1.2.5 Additional Definitions of Marginal or Degraded Land

The California Air Resources Board's Expert Working Group stated the following with regards to marginal land:

> *"Currently, no real accepted definition of 'marginal land' exists via USDA, FAO, or other agriculturally-based organizations or credible entities, but in general it may be something of the order of: "Land, such as upland, or desert border, which is difficult to cultivate, and which yields little profit or return and may have been the first land to have been abandoned." Another definition may be: "Lands which cannot adequately sustain required levels of production to at least maintain necessary soil health"* (Gibbs, et al., 2010)

Several other sources have defined marginal or degraded lands in ways that can be more easily quantified and provide the potential for the Reserve to verify degraded lands. The Tennessee Farm Service Agency defined marginal pasturelands as:

> *"Areas with environmental limitations with the primary cover of native and/or introduced grasses and/or legumes that are used or are suitable for the grazing of livestock. Based on the above criteria, marginal pastureland may be considered any grassland, fenced or not, occurring*

along a stream or permanent water body suitable for establishment to a forest riparian buffer and capable of filtering sediment and meeting one of the following two conditions:

1. *Predominantly land capability classes IIs, III, or above.*
2. *Currently experiencing a significant environmental problem. Grasslands in land capability classes I, IIe, and IIw may be eligible if they meet the criteria below.*

Significant environmental problems include:

1. *Streambank erosion.*
2. *Overgrazing.*
3. *Scour erosion*
4. *Sand blows/depositional problems.*
5. *Other applicable problems associated with erosion or sedimentation as documented* (Tennessee State Farm Service Agency, 2007)[11]

The detail provided by the definition above aligns with the Reserve's intent to standardize and provide objective criteria to confirm the status of marginal land. The type of detail provided here in relation to degradation may help the Reserve to establish a specific and unambiguous definition for degradation.

6.3.2 Legal Surplus Test

To guarantee that projects certified to avoid the conversion of grasslands or to convert marginal croplands to grassland are additional, project developers are generally required to demonstrate that there are no laws, statutes, regulations, court orders, environmental mitigation agreements, permitting conditions or other legally binding mandates prohibiting the conversion of grassland or mandating the conversion of marginal croplands to grasslands, respectively. Usually, protocols developed by the Reserve include a non-exhaustive list of laws or regulations that have been pre-identified as potentially conflicting with the legal requirement test. Currently, several government programs provide payments to producers to implement practices similar to AGC and CCG activities. These include:

Incentive Programs to Establish or Maintain Grasslands:

- Conservation Reserve Program
- Farmable Wetlands Program
- Grasslands Reserve Program
- Wildlife Habitat Incentive Program
- Environmental Quality Incentives Program
- Conservation Stewardship Program

Laws that Potentially Affect Land Use:

[11] Additional research did not yield such detailed descriptions of marginal pastureland at the state Farm Service Agency level. Instead, most states defer to Conservation Reserve Program/USDA definitions. According to federal conservation law, marginal pastureland is eligible for participation in several practices related to wetlands. Marginal pastureland included in the CRP meets at least one of the following: 1) marginal pasture land converted to wetland or established as wildlife habitat prior to November 28, 1990; 2) marginal pasture land to be devoted to appropriate vegetation, including trees, in or near riparian areas, or devoted to similar water quality purposes (including marginal pastureland converted to wetland or established as wildlife habitat" 16 U.S.C. § 3831 (2011).

- Endangered Species Act
- Clean Water Act
- California Endangered Species Act
- California Williamson Act
- Rangeland, Grazing Land and Grassland Protection Program
- County-Level Planning

Enrollment opportunities for the incentive programs established under the conservation title of the Farm Bill are likely to be eliminated or significantly reduced in the future with the passage of a new Farm Bill in 2012 or 2013. The 2008 Farm Bill, the conservation title of which establishes and sets the appropriation levels for these programs, has expired meaning there are no further enrollments until a new Farm Bill is signed into law. As of the writing of this report, the U.S. Senate has passed its version of the Farm Bill and the House Agriculture Committee has passed a version out of committee that has not been voted on by the floor. Both versions call for the consolidation of conservation programs, reductions in enrollment caps, and funding cuts of over $6 billion for Fiscal Years 2013 to 2022 (Congressional Research Service, 2012).

Please see Appendix C: Summary of Selected Policies Relevant for Determining Legal Surplus for a short description of each of these programs.

Once the decision to convert grassland to cropland has been made, a regulatory framework within the USDA is followed to enroll the newly-converted land into USDA Farm Bill programs, e.g., subsidized crop insurance, direct payments, etc. This process is initiated by the producer's submission of a "new breakings request" to the local FSA and NRCS offices. Although this process does not mandate the conversion of grassland, it does provide documentation of approval for conversion by a federal agency. "New breakings" refers to native grassland that is converted to cropland and then incorporated into the landowners agricultural land base (Ducks Unlimited and The Eco Products Fund, 2009). As part of the new breakings request, the producer works with local NRCS officials to perform an environmental assessment and development a management plan to confirm that the newly broken land will comply with federal environmental requirements. The FSA makes a final determination on whether the newly, or intended to be broken rangelands will be eligible for FSA administered Farm Bill Programs and payments. The USDA Farm Service Agency (FSA) maintains a "breaking database" across three states-North Dakota, South Dakota, and Montana- to measure cropland acres broken; however, data are inconsistent across the regions[12]. Performance Standard Test

In earlier offset protocols (e.g., the Forest Project Protocol, Nitrogen Management Project Protocol) the Reserve has prepared technical analyses to justify the establishment of a performance standard metric that can be used to determine the additionality and/or set limitations on baseline activities. In the context of AGC and CCG, the examination of land use conversion trends offered the most logical framework for setting benchmarks regarding the additionality of AGC and CCG at local or regional levels.

[12] The FSA reports that 55,405 acres in South Dakota and 10,373 acres in Montana were broken in 2005. In 2006, 47,167 acres were broken in South Dakota; 20,592 acres in North Dakota and 6,245 acres in Montana were broken.

As demonstrated in the earlier discussion of land use trends for both grassland conversion to cropland and vice-versa, however, setting benchmarks based on regional or local conversion rates is likely to be fraught with challenges owing to numerous factors such as variable land class definitions, land cover identification approaches, and trend quantification among available data sources. In light of these highly variable trends and datasets, it is unlikely that the Reserve could determine quantitative benchmarks classifying CCG or AGC as additional and non-additional based directly on localized conversion rates.

6.3.2.1 Potential for a Standardized Financial Performance Test

A fair market valuation type test could potentially be used to construct an additionality test for avoided grassland conversion. The primary challenge for such a test is confidently determining and valuing the best alternative use for a given grassland parcel. To function well, the test must deliver a conservative and reasonable baseline land use scenario as well as an accurate property valuation. Using publicly available information is not likely to resolve these issues, as grassland conversion rates (and the ensuing land use types) are highly variable across space and data available for determining the “fair market value” of current and alternative uses are generally not sufficient for valuing specific properties. These considerations suggest that a performance standard considering financial aspects of the project area would necessarily require project-specific valuation through something like a formal land appraisal. Through a formal appraisal, the project proponent may reasonably demonstrate the “highest and best” land use and be value this land use in contrast to the with-project land cover type. This approach, for example, has been incorporated into the Avoided Grassland Conversion methodology currently under consideration at the Verified Carbon Standard (VCS) and American Carbon Registry (ACR) (Dell, et al., 2012a; 2012b)[13].

6.3.2.1.1 Grassland Conversion Discount Factor

Discount factors have been used in previous protocols (e.g., Climate Action Reserve Forest Project Protocol) to account for uncertainty regarding whether the project land would have been converted in the absence of the project. In the case of AGC, conversion uncertainty arises from several sources, including: 1) measurement error or uncertainty in land values, 2) uncertainty regarding the intent of the landowner in terms of land use decisions, or 3) the return to the landowner provided by carbon revenue may be so small in comparison to other cash flows that carbon revenue is unlikely to be the decisive factor behind land management choices.

Following previous protocols, a grassland conversion discount factor (GCDF) could be similarly constructed by comparing the value of the project area in its highest valued alternative use (VA) to its value in grassland use (VG). Acknowledging measurement error in both VG and VA, the greater the differential (VA-VG) or ratio (VA/VG) the higher the likelihood of conversion.

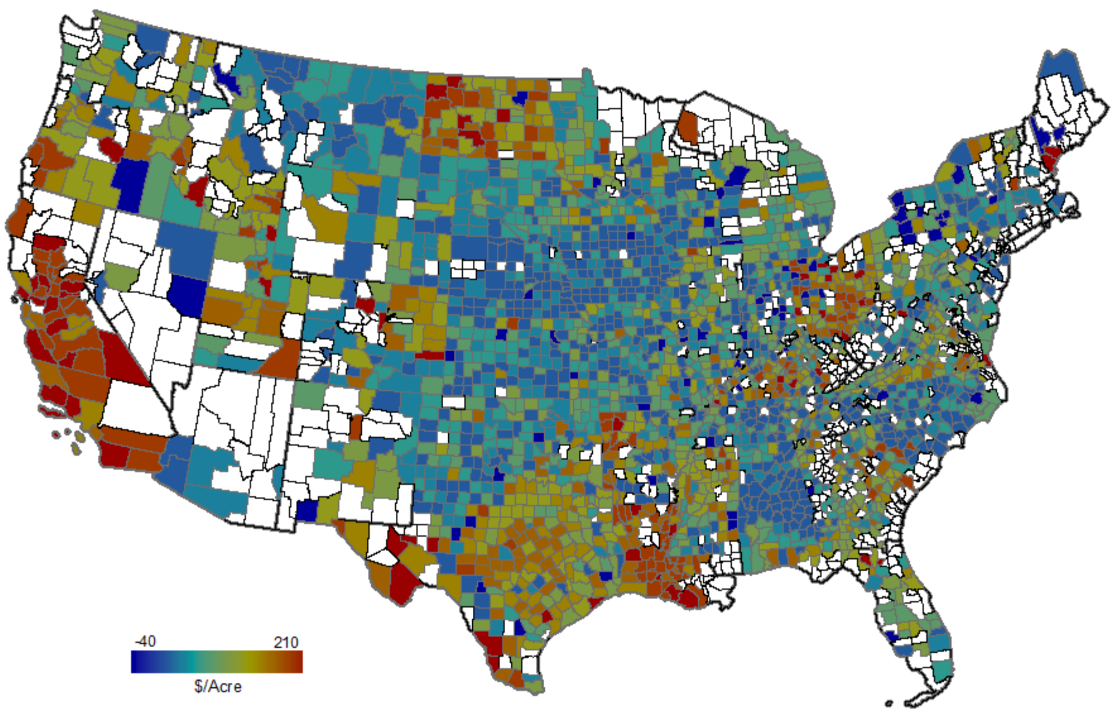
Establishing additionality using economic considerations may be considerably more challenging, as the Reserve needs to determine that the absence of a market would drive conversion. Several protocols (e.g., Ducks Unlimited VCS and ACR methods, Climate Action Reserve Forest Project Protocol) have

[13] Several authors of this Issue Paper are also authors of these proposed avoided grassland conversion methodologies under review at VCS and ACR.

proposed that eligible land must be valued at least 40% higher as a cropland than grasslands (Dell, et al., 2012a; 2012b). An appraisal can be provided to the Reserve to determine whether the land class is relevant for the protocol and the land value. Other economic considerations may include crop prices, government program participation and relevant payouts.

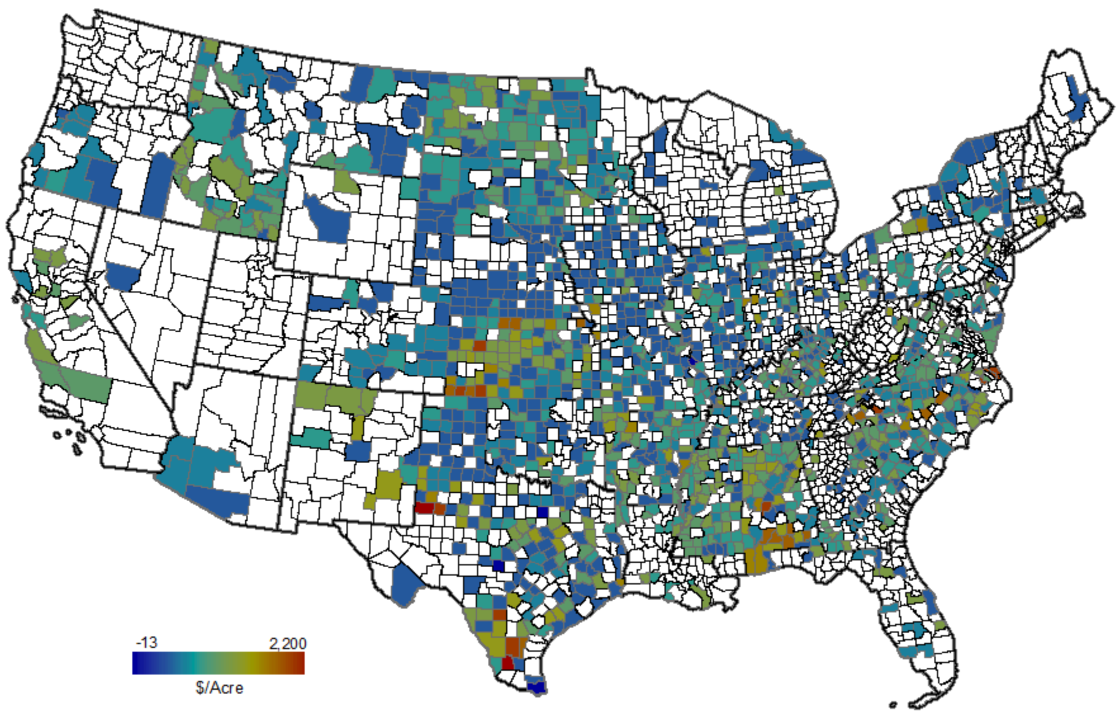
Using the NASS rental rates, value differentials are highly variable across space. In general (for counties that have rental rate data), irrigated and non-irrigated cropland rental rates are significantly higher than pastureland rental rates. On average, irrigated cropland rental rates are 8.6 times greater than pastureland rental rates (i.e., VA/VG = 8.6, on average across counties), and non-irrigated cropland rental rates are 3.6 times higher. The differentials, however, range from less than one to 156 – demonstrating that there is significant variation across counties, without regard for within-county variation. Nonetheless, comparisons of rental rate or other value estimate differences could be used to generate a GCDF similar to the approach adopted for Avoided Conversion projects in the Reserve's Forest Project Protocol.

Figure 32: Rental rate differentials between non-irrigated cropland and pasture



Note: Blank counties have no data. Negative values imply average pasture rental rate is greater than average cropland rental rate.

Figure 33: Rental rate differential between irrigated cropland and pasture



Note: Blank counties have no data. Negative values imply average pasture rental rate is greater than average cropland rental rate.

There are several additional challenges, however, to developing a GCDF. Most importantly, what cut-off values (i.e., the 0.4 and 0.8 in the Reserve's Forest Project Protocol) should be used to initiate the discounting? The rental rate data indicates that relatively few counties have value ratios less than one (four using irrigated cropland values, and 66 using non-irrigated cropland values), while the vast majority of counties have ratios between one and ten. Moreover, there is relatively little correlation between the estimated rental rate ratios and the grassland to cropland conversion rates estimated above (i.e., counties with high cropland to grassland rental rate ratios do not necessarily also have high historical conversion rates).

The authors of this Issue Paper have not found any literature that specifically provides a ratio of cropland and grassland values as a determinant of grassland conversion. Previous research on grassland conversion rates, however, has illuminated additional factors beyond land value alone that affect grassland conversion to cropland. In the Great Plains and Prairie regions, for example, previous research indicates that higher soil quality grassland plots are much more likely to be converted to cropland than low quality plots (Rashford, et al., 2012b; Stephens, et al., 2008). Rashford et al. (2012b) estimated that grasslands with land capability class 1 or 2 are approximately 1.5 times more likely to convert to cropland than plots with land capability class 3 or 4, and six-times more likely than the lowest quality plots. This suggests that project-specific soil quality could be another factor used to assess additionality and to potentially construct a GCDF.

6.3.3 Common Practice Considerations

There are a number of factors that that have been observed to drive innovation and agricultural practice change. Besides farm and producer characteristics, i.e., demographics, farm size (Caviglia & Kahn, 2001; Feder, et al., 1985; Fuglie & Kascak, 2001; Prokopy, et al., 2008), social, economic, and environmental factors can drive conversion to croplands. Economic considerations including crop prices (Claassen, et al., 2011b), farm programs (Claassen, et al., 2011a), and perceived profit (Cary & Wilkinson, 1997) can strongly impact conversion to croplands.

Social drivers of behavior change are also important. It has been demonstrated that producers' social networks and proximity to other producers can drive adoption of conservation practices (Breetz, et al., 2005; Risgaard, et al., 2007). Participation of a producer in a carbon project may encourage the participation of other nearby producers; however, if other producers in a region are converting grassland or marginal lands this may equally influence other producers.

Historical records can help to establish common practice at different scales to determine this. Environmental attitudes and values can also drive adoption of conservation practices (Luzar & Diagne, 1999) such as conversion of marginal lands or avoided conversion of grasslands. To the extent that these values would have resulted in the adoption of AGC activities regardless of the market, they would be considered "business as usual" and therefore not additional.

Establishing common practice among regions may also be necessary to confirm whether a project's proposed baseline scenario is realistic for its region. A number of measures can help establish common practice and the likelihood of producers to implement grassland preservation or conversion of marginal cropland. Quantifying the potential for conversion using the appraisal approach above is one option. Another option may be to use historical records to determine whether similar types of land in the project region have been converted recently. As previously mentioned there is baseline data to determine land use change from cropping systems to grassland and vice versa. Other regions have also already been identified as those highly sensitive to grassland conversion (e.g., the Northern Prairie, see earlier data on breaking requests). For conversion of marginal croplands, erodibility classifications can be used to determine marginal lands, though there is some inconsistency among definitions about what constitutes highly erodible land.

6.3.3.1 Government Programs and Potentially Conflicting Incentives

Government programs can play a significant role in influencing producer decision-making and conversion of grasslands and marginal lands. For example, crops grown in converted grasslands can receive crop insurance and disaster payments and marketing loan benefits, potentially motivating producers to plant areas of marginal cropland after two years of crop production (Claassen, et al., 2011a). Across 77 counties studied in North and South Dakota, Claassen et al. (2011a) estimated that government programs including crop insurance, marketing loans, and disaster payments increased producer incentives to plant areas in cultivated crop by 686,000 acres between 1998 and 2007 (roughly 2.9% of cropland total in the area). Disaster assistance had the largest effect causing a 1.2% rise in cultivated cropland in the area (292,000 acres), followed by crop insurance (235,000 acres, 1% rise) and marketing loan benefits (161,000 acres, 0.7% rise). Other studies have found that the average crop insurance payments per acre for the 16 counties in South Dakota with the highest rates of conversion were nearly twice as high as the average payment for all other counties in the state (GAO, 2007). Therefore, baseline measurements of farm program payment rates may be an appropriate statistic to use to determine where grassland conversions are most likely. Data also suggests that there is a role for the market to play in incentivizing avoided conversion because not enough government funding is available. In North and South Dakota it is estimated that 400,000 acres of native grasslands owned by more than 900 landowners were offered to be put into land easements but government funds were only available to protect 40,000 acres annually (Stephens, et al., 2008).

6.3.3.2 Private Conservation Initiatives

Land trusts are now well established throughout the country by private groups like The Nature Conservancy and often catalyzed through government initiatives. These land trusts provide financial incentives to landowners to protect their grasslands through conservation easements (Jantz, et al., 2007). Land trusts already provide the incentive for landowners to protect their land, though they may not stipulate that they have to be kept in grasslands. If land trusts simply require the land to be agricultural land use or open-space there may be the opportunity for properties with conservation easements or land trust agreements to be eligible as an AGC project. Common practice analysis can inform whether conservation agreements are commonplace for a particular region.

6.4 Baseline Data Requirements

Both CCG and AGC projects need to be able to account for baseline GHG emissions associated with cropland management activities. For AGC projects, GHG quantification for baseline scenario should account for any material GHG emissions associated with the act of land conversion.

The principal sources, sinks, and reservoirs (SSRs) related to cropland management in the baselines of CCG and AGC projects that should be considered include:

- soil carbon changes due to tillage
- direct and indirect N_2O emissions from the land application of fertilizer (organic and inorganic)
- above and below-ground plant biomass
- fossil fuel emissions from operation of farm equipment, including irrigation

For AGC projects, the GHG impacts of the baseline conversion of grassland to cropland will generally be expected to affect these same SSRs. However, relatively larger changes in SOC stocks may be concentrated closer to the time of conversion for AGC projects, and fossil fuel emissions from operation of farm equipment may also be higher during conversion in AGC baselines than would be observed in the ongoing management of the area once it has been converted to cropland.

If the Reserve considers including AGC baseline scenarios for other land uses beside cropland (e.g., (sub)urban development, golf courses, mineral and gas extraction, etc.), the GHG emissions during and following conversion are likely to be substantially different from cropland. In general, comparatively less research has been devoted to quantifying changes in SOC in conversion to these other land use types, so the Reserve may need to use significantly different approaches to quantifying baseline emissions if additional land use types can be accounted for in the baseline scenario. Considering the primary source of avoided emissions for AGC derives from protection of SOC stocks, the scale of soil disturbance may be a useful proxy for determining the most conservative baseline scenario if multiple land use scenarios are possible. Alternative baseline land use scenarios with relatively lower levels of soil disturbance would be expected to produce fewer emissions than those with higher levels of soil disturbance.

In addition to these SSRs described above, the Reserve may elect to account for upstream emissions from the production or transport of fertilizers and manure, or irrigation and fertilizer application equipment. In the two other agricultural land management methodologies currently approved by the Reserve (i.e., the Rice Cultivation Project Protocol and the Nitrogen Management Project Protocol), these upstream emissions have been excluded. Baseline activities for CCG projects are likely to be defined with the continuation of current input levels, and thus no change in upstream emissions should be expected to occur in the baseline for this project type. For AGC projects, however, baseline activities should be expected to increase the use of these inputs and their corresponding upstream emissions. As emission factors are available in the scientific literature to estimate the upstream emissions for these agricultural inputs, the choice whether to account for them is primarily a policy, rather than scientific, question. The relevant arguments that have been used regarding these upstream emissions in other protocols are:

- upstream emissions associated with any increase in the use of these inputs are expected to be very small in relation to net project GHG emissions
- changes in upstream emissions occur offsite
- upstream emissions may be regulated through other programs or policies

As in the other protocols, the scale of changes in upstream emissions for AGC baseline activities is also likely to be relatively small in comparison to other SSRs such as SOC changes and N_2O emissions from fertilizer application, and exclusion of these emissions from the baseline scenario would be conservative. The other points above involve the policy stance of the Reserve regarding accounting for indirect emissions that are not directly related to specific project types and thus would not be expected to change between protocols.

6.4.1 Quantifying Baseline Sources, Sinks, and Reservoirs

6.4.1.1 Baseline emissions from farm inputs

The process for estimating baseline emissions from cropland inputs will necessarily involve assumptions by the Reserve regarding the average or most likely levels of inputs in terms of type and amount of fertilizer applied, type of equipment used and its duration of use. These input levels would then be multiplied by emissions factors that could be derived from scientific literature or other offset protocols.

For AGC projects, these input levels should ideally be based upon values reported in farm surveys or scientific literature that are representative of the conversion and management of cropland in the region or locality where the project is implemented. Potential sources for such data include the USDA Natural

Resources Inventory, the USDA Economic Research Service's Cropping Practices Survey and National Agricultural Statistics Service.

It is also important to consider that the choice of inputs may vary significantly between specific crops or crop rotations. For example, nitrogen fertilizer application would be expected to be substantially lower when nitrogen-fixing crops are grown. The specific choice of crops considered in the baseline scenario could be based on that which would generate the highest expected economic returns or the prevalence of acreages allocated to these crops in the project region, for example. Land capability classes and subclasses—as defined by the USDA Natural Resource Conservation Service—for the soils in the project area may also provide a useful source of information regarding the scope and extent of tillage, water management, and nutrient inputs that might be expected.

For CCG projects, baseline emissions from these inputs could be more reliably based upon historical records from the project area, similar to the 5-year lookback period required for projects under the Nitrogen Management Project Protocol.

6.4.1.2 Baseline soil carbon stocks

As the pool most likely to dominate the GHG accounting for AGC and CCG projects, a process to provide reasonable estimates for SOC stocks or changes is critical to maintain the integrity of any offset credits derived from these project types. Changes in SOC stocks in the baseline are expected to be larger for AGC projects as opposed to CCG projects since the land use conversion is the primary driver of these changes, but the major considerations outlined below apply to baseline quantification for both project types.

6.4.1.2.1 Soil carbon stocks at project initiation

If the Reserve adopts the stock-change accounting approach (as opposed to the flux approach) for estimating baseline soil carbon emissions, SOC stocks at project initiation must be quantified[14]. The two general options for quantifying starting carbon stocks are field sampling or the use of a look-up table with SOC averages estimated across regions or climate zones, which can be defined at various scales.

6.4.1.2.2 Look-up table approaches

There are three general approaches to generating look-up values for initial SOC stocks: direct averaging of SOC values using soil survey data; using the IPCC approach with regional SOC reference levels and multiplication factors reflecting the effects of land use, management, and inputs; and a hybrid approach where IPCC-style land management factors could be multiplied by SOC reference levels calculated from soil survey data within smaller geographic regions than the IPCC climate zones.

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[14] The difference between stock change and flux approaches is described in Section 3.1.4 "General Methods" of IPCC's Good Practice Guidance for Land Use, Land-Use Change and Forestry (2003, pp. 3.15-3.16). Most offset protocols require periodic quantification of carbon stocks and use the stock change approach. However, the flux-based approach may be used without ever quantifying soil carbon stocks. For example, the flux-based approach is adopted in the CCX soil carbon protocols where pre-determined sequestration rates are assigned to project activities based on their location (derived from climate and soil characteristics) without absolute soil carbon stocks ever being quantified. The primary concern regarding a flux approach is the reliability of the sequestration rate that is assigned to projects without regard for their actual soil carbon stocks.

There are three primary sources of data for soil carbon stocks in the US. The Soil Survey Geographic Database (SSURGO) and the USDA State Soil Geographic Database (STATSGO) provide data by soil map units with national coverage. The other common source for US soil carbon data is National Cooperative Soil Survey (NCSS) Soil Characterization Database[15], which provides measurements collected from individual pedons across the US. Most efforts that have been published to date involving estimation of SOC storage in the US utilize these data sources (e.g., Buell & Markewich, 2004; Guo, et al., 2006; Sundquist, et al., 2009; US EPA, 2012a).

Averages from NCSS soil carbon data have been reported in Ogle et al. (2003) as SOC reference values by climate zones and soil types that can be multiplied by US-specific land-use and tillage factors provided in Ogle et al.(2006) to estimate equilibrium SOC stocks for the chosen management strategy and location[16].

The application of IPCC-style factors like those described in Ogle et al. (2003; 2006), rely upon an assumption that SOC stocks move between equilibrium values under specified configurations of land use, tillage, and inputs over 20 years. For lands that have recently changed from one land use, tillage type, or input level to another, SOC levels may still be changing and could therefore introduce error into the estimates of SOC stocks at project initiation. Repeated historical measurements of SOC stocks will generally not be available for offset projects to estimate baseline trends in SOC. If the Reserve desires to evaluate the trajectory of SOC stocks in the absence of the project, the only option is thus to extrapolate these SOC stocks using the look-up approach above or through the use of biogeochemical modeling.

For AGC projects, the primary concern in terms of conservative baseline accounting regards assigning an SOC stock level at project initiation that is too high. This may occur, for example, if the potential AGC project area was recently converted to grassland from cropland. Assuming SOC stocks at project initiation were at a higher grassland equilibrium level could then overestimate baseline SOC emissions during conversion to cropland. For projects with grasslands that were recently converted from cropland, the actual emissions benefits would more likely arise from increased SOC sequestration by maintenance of grassland cover as opposed to avoided emissions due to conversion back to cropland. It may thus be more appropriate to consider accounting for project lands that were recently converted from cropland to grassland that are vulnerable to re-initiation of cultivation using a GHG accounting approach more comparable to CCG projects. Precluding eligibility to these types of lands is also an option for dealing with this situation, but such a policy would effectively remove incentives for GHG benefits that could be accrued by securing a commitment from the landowner to maintain the current grassland cover.

For CCG projects, the primary concern regards assigning an SOC stock level at project initiation that is too low. This may occur, for example, if the land manager only recently reduced levels of inputs or

[15] Available at http://ncsslabdatamart.sc.egov.usda.gov/default.htm

[16] This is the approach used to calculate technical emission reduction potential for AGC and CCG activities in this issue paper. This general approach was originally described by in the IPCC's *Good Practice Guidance for Land Use, Land-Use Change, and Forestry* (IPCC, 2006).

adopted tillage practices with greater soil disturbance. Assuming that SOC stocks are at equilibrium under these recently adopted practices could lead to an artificially low estimate of baseline SOC stocks. To avoid this type of "gaming" for CCG projects, the Reserve could consider assigning an intermediate SOC stock value between equilibrium values, limiting eligibility to exclude these types of lands, or assigning an SOC stock value that is based on the highest equilibrium value of management practices that had been utilized in the project area over recent history.

6.4.1.2.3 Defining when conversion occurs across the project area

In some circumstances, the baseline conversion of grassland areas in AGC projects may occur over more than one year. In preparing a baseline scenario, potential AGC projects may be asked to justify an expected timeframe for the successive conversion of the project area in the baseline, or the Reserve may make simplifying assumptions regarding the default rate of conversion (e.g., compete conversion in year 1, proportional conversion over several years, etc.).

6.4.1.2.4 Change in SOC stocks associated with baseline activities

Following the determination of initial SOC stocks and the timing of conversion, the next step in estimating baseline SOC changes involves quantifying the expected change in SOC corresponding to cropland conversion for AGC projects, and ongoing cropland management for CCG projects.

The two primary methods for estimating these SOC changes are IPCC Tier 2 (regional factor-based) and Tier 3 (biogeochemical model-based) approaches. Both of these approaches, for example, are utilized to derive estimates of SOC impacts due to land use change in the US national GHG inventory (US EPA, 2012a).

For the factor-based approach, this could involve defining the expected SOC stock change over a defined period of time and then amortizing those SOC changes for each year. This Tier 2 factor-based approach recommended by the IPCC is generally applied with a linear amortization of SOC stock changes over a period of 20 years, with the ultimate equilibrium SOC stock determined through multiplication of land use, tillage, and input factors by a reference SOC level (Ogle, et al., 2003; Ogle, et al., 2006).

Biogeochemical process models such as CENTURY or DNDC may also be suitable for simulating a dynamic SOC baseline for both CCG and AGC project types. At present, the expertise required to operate these models suggests requiring their use by project proponents is likely to be a significant hurdle to project development. However, a suitable alternative that could be the simulation of several land management scenarios and the derivation of generalized SOC change curves for a variety of soil types and common baseline management practices. For example, the Reserve could prepare model runs simulating grassland to cropland conversion in each ecoregion or county reflecting the five-to-ten most common soil types and crop management practices, producing a matrix of SOC changes expected for each geographic area. The successive classification of soils from soil orders down to soil series is likely to correspond to reduced levels of variation in SOC stocks (Guo, et al., 2006). It is also likely, however, that many CCG or AGC projects would be developed across of broader range of soil series than could be feasibly individually modeled in each ecoregion or county. A compromise approach could involve model runs prepared for the most common soil classes and averaged across a higher-level of classification (e.g., soil family, subgroup, or great group).

Both the factor-based approach and the biogeochemical modeling approach would likely be feasible (assuming the Reserve was responsible for the modeling) and suitable for standardized baseline accounting. The accuracy of both of these methods, however, will be subject to significant uncertainty if field sampling of soil in the project area is not conducted. At a minimum, it is recommended that projects undergo field sampling to confirm the distribution of soil types across the project area. Field sampling for SOC could introduce significant cost concerns for project feasibility, and the authors thus urge the Reserve to consider sampling costs when determining the desirable scope of soil sampling for each project.

Aggregation has been suggested for many project types to help reduce transaction costs for smaller projects, and the authors encourage the Reserve to prioritize the inclusion of aggregation rules in protocol development for all land use project types. Although the SOC sequestration rates identified for AGC and CCG may be substantially higher than other land use project types (e.g., nitrogen management, Rice Cultivation), the authors nevertheless expect aggregation will be a necessity for reaching a meaningful scale of AGC and CCG activities for most projects. One particularly valuable aspect of aggregation is that structural uncertainties that are inherent in the use of biogeochemical models can be progressively reduced as a greater number of sites are modeled within an aggregate.

6.5 Leakage Risks and Policy Options

As described in greater detail in Appendix E: Leakage Lit Review, changes in SSRs outside an offset project's boundary can be of such a scale that the conservativeness of offset project accounting may be materially affected. Although these changes in SSRs may positive or negative, most offset protocols focus in particular on increases in emissions or reductions in carbon stocks or sequestration outside the project's primary emissions scope that are due to the project's implementation. Estimating and accounting for these secondary emissions, becomes a principal concern for offset protocols accounting for these "leaked" emissions.

Of primary relevance for the AGC and CCG project types considered throughout this issue paper, any offset project that affects the supply of agricultural commodities may be expected to indirectly effect a change in land use outside the project boundary to compensate for the increased or decreased supply. A great deal of research has emerged attempting to quantify the extent and effects of these indirect land use changes (ILUC) and their associated emissions, which is summarized in Appendix E: Leakage Lit Review. This section focuses on the policy options to address ILUC as a source of potential leakage.

As the Reserve highlights in the RFP for this Issue Paper, there are three fundamental questions an offset project protocol leakage policy must consider to estimate and address emissions from ILUC:

i) *How much land will be brought into production (or converted to other land uses) to compensate for land taken out or precluded from production/conversion*

ii) *Where will the compensating conversions occur in relation to the project area (i.e. will leakage be localized, or will it primarily occur in other regions/ecosystems, and*

iii) *What are the biomass and soil carbon stock values for the ecosystem type where the compensating conversions occur?*

6.5.1 Potential Magnitude of Leakage

Most of the scientific literature characterizing ILUC has emerged within the scope of regulatory biofuels mandates. Although interest in offset project accounting has generated some research regarding ILUC (e.g., Murray, et al., 2007), to date there remains relatively few studies published focusing directly on this topic. A similar topic has received modest attention regarding "slippage" in government conservation programs, particularly the CRP (Wu, 2000; Roberts & Bucholtz, 2005; Fleming, 2010). These research studies have suggested a range of potential leakage from cropland set-aside[17]. Quantifying "slippage" as the proportion of land area enrolled in the conservation program that is offset by increased cropland cultivation elsewhere, these CRP-focused studies have successively produced lower and lower estimates of slippage over time.

Wu (2000) estimated slippage from CRP varied from 15% to 30% across regions in the US, with a program-wide average of 20%. Disputing the methods used by Wu (2000), Roberts and Bucholtz (2005) estimated lower slippage rates ranging from 2% to 19% across the country. Utilizing satellite imagery across the same region (earlier studies utilized NRI land cover data), Fleming (2010) estimated a much lower slippage rate of 3.7%-3.8%.

This "slippage" effect has also been characterized as a "Net Displacement Factor" in a review of biofuels ILUC literature by Plevin et al. (2010). Among those studies reviewed by Plevin et al. (2010) and others that have provided comparable figures (summarized in Table 27 in Appendix E: Leakage Lit Review), estimates for ILUC slippage, again in terms of land area, range from 9% to 89%. Murray et al. (2007) suggests a leakage rate, in terms of emissions rather than land area, for cropland retirement projects of 0% to 20%. There are several important assumptions and factors that affect the calculation of this Net Displacement Factor, which are described in greater detail in Appendix E: Leakage Lit Review.

Because ILUC is fundamentally an unobservable phenomenon, attempts to "map" the locations of ILUC and quantify the corresponding changes in carbon stocks fundamentally rely upon simulations. The biofuels literature contains several examples of broad global economic and land use models designed to estimate these factors from underlying economic effects and various biological and technological constraints. Reviews of the modeling approaches across the biofuels ILUC literature by Plevin et al. (2010) and the European Commission's Directorate General for Energy (2010) indicate that there is no clear "winner" among the various models that have been used to simulate ILUC. The sensitivity analysis prepared by Plevin et al. (2010) further complicates the matter by highlighting that the values these models produce are also highly variable and subject to several major uncertainties.

6.5.2 Policy Alternatives for Leakage Accounting

One approach anticipated by the Reserve in the RFP for this Issue Paper regards the potential for commissioning a global ILUC simulation using one or more existing models. This approach has been taken, for example, by the California Air Resources Board (ARB) in preparation for its Low Carbon Fuel

[17]Reductions of cropland or commodity supplies are not considered to be materially different between cropland set asides and avoided conversion of grassland to cropland. We expect market-effects leakage to respond equivalently to a reduction in cropland or commodity supplies, regardless of whether the underlying source of the supply reduction was AGC or CCG activities.

Standard, and the US Environmental Protection Agency (EPA) for the national Renewable Fuels Standard. A general review of these and other modeling approaches is described in Appendix E: Leakage Lit Review. Given the extent of variability and uncertainty involved in these modeling exercises, as well as the fact that many of the influential assumptions regarding these simulations are not transparent, the authors of this Issue Paper suggest a more transparent policy alternative for developing offset leakage accounting for a potential AGC and/or CCG methodology would involve the use of a simplified model for estimating expected leakage effects, such as that proposed by Plevin et al. (2010):

$$ILUC = \frac{NDF \times EF}{T \times Y}$$

Where:

$ILUC$	emissions due to indirect land-use change; tCO_2e per unit of yield reduction (e.g., tCO_2e/bushel)
NDF	Net Displacement Factor, the ratio of land area brought into crop production to the area subject to reduced yields due to project activities; dimensionless
EF	average GHG Emission Factor for the land area brought into crop production; tCO_2e/ac
T	timeframe over which project-induced yield reductions are considered (e.g., project life); years
Y	the annual yield reduction induced by project activities over the timeframe T; unit of yield reduction per acre per year (e.g., bushel/ac/yr)

This reduced-form model simplifies the multitude of drivers and responses simulated in global equilibrium models to comparatively intuitive factors more akin to the three-step process originally envisioned by the Reserve[18]. The sensitivity analysis by Plevin et al. (2010) suggests the single most important factor in this equation across global economic modeling, in terms of control exerted over the ultimate estimate of leakage due to ILUC, resides in the Net Displacement Factor (NDF).

The NDF variable can be approximated using available literature, such as consideration of published elasticity values for cropland supply and demand. Greater detail on the available data and potential avenues for estimating NDF and other factors for this reduced-form model can be found in Appendix E: Leakage Lit Review. In addition, Table 26 provides an overview of the various approaches to leakage that have been utilized to date in offset standards and methodologies.

6.5.3 Policy Options to Mitigate Leakage Risk

The primary driver of ILUC is generally considered to be market-mediated responses to changes in the supply and demand for agricultural products introduced by the project activity (in the case of biofuels literature, this is often discussed as a biofuel “shock”). It is important to recognize that the scope of market-mediated responses will encompass “activity-shifting” leakage, as well as the more obvious “market effects” leakage.

In the RFP for this Issue Paper, the Reserve indicated an interest in identifying the potential for variations in the potential for leakage effects related to specific crops or geographic regions. For potential offset projects in the US, most, if not all, of the baseline crop production activities that will be foregone by AGC and CCG project activities involve commodity crops that are traded and priced on international markets, and thus ILUC effects should be expected to occur globally rather than in localized areas surrounding potential offset projects. The responsiveness of these commodity markets to changes in supply for specific crops is quite complex, but can also be roughly estimated using the supply and demand elasticities discussed in Section 12.2.1.1.1 to estimate the proportional amount of land that may be brought into production to make up for lost supply due to offset project activities.

The Reserve has also indicated an interest in limiting the eligibility of projects to those on marginal or degraded lands with relatively lesser value for crop production. This strategy reflects the recognition that marginal agricultural lands would be less productive and the removal of these areas from cultivation through AGC or CCG activities would thus involve a smaller yield shock. Murray et al. (2007) also argue that most cropland retirement projects incentivized by carbon offset funding would be likely to occur on marginal lands and therefore justify a lower leakage discount that projects on more productive lands. As a general principle, the authors of this Issue Paper believe—as do Murray et al. (2007)—that it would be reasonable to assume that the areas likely to be protected through AGC or CCG activities are likely to have below-average productivity as cropland.

Unfortunately, the extent of leakage mitigation that could occur through this type of applicability criteria is not readily quantifiable, and the literature review on leakage for this Issue Paper did not uncover any quantitative estimates for this type of policy. While it is reasonable to assume that limiting project eligibility to marginal or degraded lands would reduce the absolute scale of leakage, there is no corresponding justification that leakage would be completely eliminated. Since the primary driver of market-mediated leakage is assumed to be supply and demand of the agricultural commodity affected by project activities, the foregone crop yield is the direct driver of leakage, regardless of the quality of land producing it. Murray and Baker (2010) provide additional steps to account for project-induced yield reductions, which would be multiplied by the ILUC emissions factor[19]. By incorporating the yield reduction induced by project activities, the reduced-form leakage model discussed above should fairly account for leakage due to yield reductions from projects on marginal or highly productive cropland.

The Reserve may consider alternative implementations for the ILUC emission factor by changing the units used to estimate yield reductions. For example, the ILUC emission factor could simply be emissions per acre of land taken out of production, or it could be used to address specific crop yields.

6.6 Reversal Risks and Policy Options

The primary GHG benefit for both AGC and CCG projects is the gain or avoided emission of SOC. Although SOC levels can fluctuate in response to climate, environmental or management factors, the

[19] Although no published offset project methodologies currently employ this approach to calculate project-specific leakage discounts based on yield reductions and elasticities of supply and demand, the American Carbon Registry is currently developing a methodology for rangeland and livestock management projects that does employ this approach. Specifically, yield reductions in milk or meat due to project activities are used to calculate discounts for market effects leakage using the elasticities of supply and demand for these commodities. This methodology is expected to be published in early 2013 (personal communication, Nick Martin, Nov 7, 2012).

SOC pool is relatively stable for grassland systems barring changes in land-use or long-term mismanagement. This section assesses the two primary forms of potential carbon reversal: natural disturbances and anthropogenic disturbances. Natural disturbances are those that occur through natural events, even if their occurrence is influenced by anthropogenic factors such as management decisions. Anthropogenic disturbances are those that occur as the direct result of a human decision, even if their occurrence may be outside the control of the Project Participant. A review of potential reversal events, the likely impact of their occurrence on project SSRs, and potential policy options to deter and mitigate their occurrence are presented in this section.

6.6.1 Natural Disturbances

The majority of natural disturbances (often considered unavoidable reversals) primarily affect aboveground biomass, indirectly affecting SOC and belowground biomass. The aboveground carbon pool in rangeland systems is a minor proportion of total ecosystem carbon storage, with mean residence time of this pool only a few years and yearly variations in aboveground biomass minimally affecting carbon storage (Derner & Jin, 2012). Most SOC in rangeland systems is recalcitrant and immune from minor natural disturbances. However, major disturbance such as tillage, severe over grazing, or wind or water erosion, can lead to significant declines in the SOC pool whether they occur naturally or as the result of poor management (Follett, et al., 2001). The major types of natural disturbances are discussed below.

- Fires, both naturally-occurring and prescribed, can have mixed effects on rangeland SOC by enhancing nutrient availability for the subsequent growing season, which can exceed fire combustion losses. Fires can also indirectly affect net system carbon storage by altering plant species composition, soil and canopy respiration and photosynthesis. Carbon emissions from the aboveground biomass are a small component of total annual ecosystem carbon losses, but may be more substantial for rangelands with more woody dominant herbaceous matter (Derner & Jin, 2012).
- Floods or other long-term inundations could change soil aerobic conditions, altering the flux of other GHGs and SOC levels.
- Changes in vegetative communities, through disease, pests or the spread of invasive species, can indirectly influence Soil Organic Matter and Organic Carbon levels. The interaction of these variables with management practices and climate will likely have regionally unique effects on SOC levels, positive or negative. Cheatgrass, for example, a common invasive annual grass in much of the west, can decrease SOC levels relative to native systems at deep levels (Rau, et al., 2011), but may increase OC at more near surface levels (Hooker, et al., 2008).
- Erosion from wind or water is a possible transfer mechanism for carbon from grassland systems. Soil erosion risks on U.S. grazing lands are less than for cropland, and are influenced by numerous factors (Lal, 2001). As erosion is mostly a transfer of soil carbon from one location to another, typically from an upland site to a basin, there is little net loss to the atmosphere. For example, the SOC content of wind and water derived sediment is estimated to be 2.0% and 3.0%, respectively, of which 20% of this SOC is estimated to be emitted to the atmosphere as CO_2 through either mineralization or oxidative processes (Lal, 2001).

6.6.2 Anthropogenic Disturbances

In the context of AGC and CCG, anthropogenic disturbances (often considered avoidable reversals) may include deliberate management decisions to alter grasslands to a different land cover or land use, primarily through physical disruption of the land cover and soils. Anthropogenic disturbances can be further categorized based on the agent of the disturbance: reversal by landowner, and reversal by a third party. The circumstances and legal repercussions for these events are described below.

6.6.2.1 Reversal by Landowner

In response to changing economic conditions, or a change in land ownership where the new landowner has different land use preferences than the original landowner, a landowner could knowingly undertake an activity that is in violation of the Project Agreement encumbered to the property that would result in the reversal of project soil carbon benefits. This scenario could be expected where the perceived economic or other benefits to the landowner of implementing the reversal activity exceed the costs of violating the Project Agreement. Based on the example of conservation easements, easement violations are more likely to occur by a second-generation landowner than by the original landowner that entered into an easement (vanDoren, 2004; Rissman & Bustic, 2011). Although it is possible that these violations are intentional, ignorance of contract terms by the new landowner is also possible.

Persistent and visible monitoring of easement or agreement terms, as well as credible enforcement and costly penalties for violations could minimize this potential risk. Penalties for reversal sufficient to cover the purchase of replacement CRTs could also be deployed. Clear communication of land encumbrances on Project Properties at the time of a transfer in land ownership would also alert potential landowners to their responsibilities and deter activities that would lead to soil carbon reversals.

6.6.2.1.1 Wind, Solar Energy Development

The right to develop wind and solar energy sources are maintained and granted by the surface owner, unless otherwise prohibited by a local, state or federal entity. Landowners therefore have direct control, or the leverage to negotiate, how these activities would impact an AGC or CCG project activity. Direct land impacts from wind development, including the tower footings and service roads, are typically less than 5% of the wind project area (Fargione, et al., 2012). The impacts of such activities on SOC stocks have not been well researched. It would be expected that the initial soil disturbance would cause a decrease in SOC, but the long-term effects are uncertain. Land use impacts by solar projects will vary by technology deployed (Denholm & Margolis, 2008), but would be expected to impact aboveground vegetation and the biological soil crust where installed. The net effect of these activities on SOC stocks is not well understood and additional research is needed.

6.6.2.2 Reversal by a Third Party

Third parties may intentionally or unintentionally enter Project Properties and undertake an activity that reverses soil carbon benefits of project activities. Trespass scenarios could include operators from an adjacent property accidentally crossing a property boundary and tilling grasslands, builders or adjacent landowners developing roads or buildings illegally on Project Properties, All Terrain Vehicle users illegally trespassing and disrupting project sites, or extractive resource developers illegally impacting Project Properties. In all cases, the landowner or the Agreement holding entity would have legal recourse to pursue remediation and financial compensation for illegal takings. Legal forms of reversal by third parties could include government agencies exercising Eminent Domain to develop or otherwise impair SOC levels of Project Properties, or where mineral development, e.g., oil and gas, allows the legal use and disruption of the surface property.

6.6.2.2.1 Eminent Domain

A conservation easement can be rescinded through the Eminent Domain process (Jones, et al., 2009). The activity implemented through Eminent Domain, e.g., road or other infrastructure development, would impact the magnitude of reversal. Even though the reversal of the activity may be minor or indeterminate, land ownership would no longer remain with the Project Participant, and offsets associated with the Project Property would need to be terminated. Limited data is available on the use of eminent domain in the United States, but common cases include infrastructure development and maintenance, building or maintenance of government buildings, blight and environmental remediation (GAO, 2006). A buffer pool could be used to account for the occurrence of these events.

6.6.2.2.2 Mineral Exploration and Development

Minerals generally include oil and gas, coal, metals and metal-bearing ores, and non-metallic minerals. Sand, gravel and peat, and other surface resources are typically classified as belonging to the surface estate but may be considered as part of the mineral estate in some states. In general, exploration and extraction of mineral resources in this document refers to oil and gas. Oil and gas operations typically include geophysical exploration, building drill pads, road building, drilling, potentially laying pipelines and waste water disposal pits. The direct land footprint of these activities is typically small, accounting for 5% of a development area, or approximately 12 to 79 acres (5 to 32 hectares) per well, depending on the mineral and extraction practice (McDonald, et al., 2009). It is possible for additional indirect effects to impact a larger geography, such as wastewater or oil or gas spills. The effects of these events on SOC stocks are not well understood and need additional research.

The ability of the landowner, or more specifically the surface owner, is typically limited in controlling how mineral exploration or development occurs. The United States is unique among nations in that it is one of the few where mineral rights can be held by private entities. In Canada, for example, the mineral rights of over 90% of lands are owned by the government (Natural Resources Canada, 2011). Laws governing the exploration and development of mineral resources vary by state, but typically favor those of the mineral estate over the surface estate. In Texas, and several other states, the granting of the mineral right permits the mineral interest the implied right to use as much of the surface area as is “reasonably necessary” for the exploration and development of the minerals (Fambrough, 2009). The dominant estate doctrine maintains that the mineral interest has no legal duty to pay for damages as long as they occur from reasonable operations, and has no duty to restore damages unless specified in an express contractual provision (Smith, 2008). For example, in Texas, the mineral owner’s rights extend to interfering with the surface owner’s use and significant damage to the surface without a legal obligation for compensation. In contrast, in Oklahoma and other states with surface damage legislation, a mineral interest is not prohibited from using disturbing the surface in such a way that causes long-term damage to vegetation and topography as long it is reasonably necessary to its operations, although the mineral interest must pay the surface owner the value of the land it uses and for any damages to the surface (Smith, 2008).

Violation Policies employed by Land Trusts may be preventative (e.g., communicating with landowner, frequent monitoring, educating local real estate professionals and town officials) and/or responsive (e.g., inspecting/documenting violations, contacting landowner to determine their awareness of violations, and legal consultation to remediate any violations)(vanDoren, 2004). A combination of these strategies could be used to reduce the likelihood of violations that would affect carbon pools of Project Activities. Funding requirements necessary to address the growing number and expense of legal challenges are a challenge facing the land trust movement nationwide, independent of the size and financial capacity of the land trust (Rissman & Bustic, 2011).

6.6.3 Extent of Reversals

Intermediate to long term changes in SOC are influenced by the availability of inputs and outputs, but are constrained by context factors and indirect controls. Context factors such as climate, potential biota, time, topography, and parent material, constrain SOC accumulation thereby establishing their potential range (Chapin, et al., 2002). The majority of reversal events are unlikely to change these factors in a significant manner. If avoided emissions from baseline activities, e.g., fertilizer emissions from crop agriculture, are included in a protocol, these avoided emissions would not be at threat of reversal. Carbon losses from the natural disturbances considered in Section 6.6.1 above could be regained through additional management practices that ameliorate the impact or outputs from the reversal event. Total system carbon losses from these events are not estimated to be large, however, at least relative to the land-use change possible as Avoidable Reversals.

6.6.3.1 Natural Disturbances

Impacts from unavoidable reversals on AGC and CCG projects would be generally similar. The extent of reversals are generally expected to be minor, although it is possible for more significant losses to occur with the interaction of certain management practices, environmental conditions and/or climate conditions. These interactive effects are highly regionally and temporally dependent, but in general, intermediate- to long-term changes in SOC are a result of cumulative inputs and outputs, as influenced by factors such as temperature, precipitation, nitrogen availability, and soil resources (Pineiro, et al., 2010). Even in these scenarios, ecosystem carbon losses are generally not substantial, decreasing SOC levels by 10-30%. Context and regional specific management will be necessary to minimize losses and to maintain or increase system carbon levels. For the most part, gains lost through these events can be regained through management practices that address system needs. An adaptive management plan incorporated in a Project Implementation Agreement could address the impacts from most of these events, and also prevent those influenced by management decisions.

6.6.3.2 Anthropogenic Disturbances

Avoidable reversals that involve land use change represent a greater magnitude of reversal, although the extent of the reversal will depend on the form of reversal and whether and how the system is restored. The extent of reversal, and the ability and rate at which losses can be regained are presented for AGC and CCG, as the reversal dynamics vary by project category.

In AGC projects, the extent of the reversal for avoidable reversals would be identical to the crediting baseline if the event that the project was implemented to avoid occurs. For native grassland systems, the loss of SOC from tillage has been estimated to range from 20-60%, with most losses occurring in the first 5 to 30 years at a non-linear rate (Davidson & Ackerman, 1993; Ogle, et al., 2005; Liebig, et al., 2005; DeLuca & Zabinski, 2011). Davidson and Ackerman (1993), in their initial survey of the literature, concluded that 24% to 43% of SOC is lost following conversion, with most losses occurring immediately after cultivation. Total ecosystem carbon losses, and the temporal dynamics of these losses are dependent on environmental, climate and management factors.

Where restoration is required and possible for AGC projects, such as being mandated and enforced by an easement or Project Implementation Agreement, carbon reversals could be partially regained, mitigating the total loss of the reversal event. A meta-analysis found that restoring perennial vegetation on former cropland returned SOC levels to within 82% and 93% of native sites for temperate moist and temperate dry climates, respectively (Ogle, et al., 2005). Most studies that have looked at SOC potential on restored sites have observed sites with a long history of cultivation, and it is possible that SOC levels could be restored to higher levels and at a quicker rate if the cultivation or other Reversal event is short-lived and the restoration performed in a timely manner.

CCG projects involve previously cultivated or degraded sites, and the extent of their reversal is different than a reversal for a native system such as under an AGC project. The guidance of the IPCC for land conversion from croplands-to perennial grasslands-to croplands, uses a 100% loss of SOC gains at the time of conversion back to cropland (IPCC, 2006). Additional SOC sequestration is accounted for at the initial rate of the project activity (IPCC, 2006).

CCG projects could be very similar to the USDA's Conservation Reserve Program, which established grassland and other perennial covers on highly erodible soils. Sequestration rates for the Cropland Reserve Program in the US have been variable (Gleason, et al., 2008; Kucharik, 2007). Paustian et al. (1995) observed a SOC gain of 0.9-3.6 tCO_2e/ac/yr for CRP. In estimating the total mitigation potential for US grazing lands, Follett et al. (2001) used an annual sequestration rate for converting cropland to pasture at 0.59 to 1.8 tCO_2e/ac/yr (400 to 1200 kg C/ha/yr) and enrolling land to CRP to sequester 0.9 to 1.3 tCO_2e/ac/yr (600 to 900 kg C/ha/yr). A reversal event, if mitigated through restoration of the project site, could be regained although a temporal loss would occur.

6.6.4 Summary and Policy Options for Reversals

6.6.4.1 Permanence and the Role of Easements

It is envisioned that easements or other forms of deed or other encumbrances (e.g., a Project Implementation Agreement) will be used to maintain grass land cover for project activities. As of July 30, 2012, there were over 95,000 conservation easements in the United States, encumbering over 18 million acres (The Conservation Registry, 2012). Easements are used for various reasons, including preservation of open space, historical preservation, recreation, educational or environmental system purposes, and are held by array of parties including various branches of government, non-governmental organizations, private parties, and others. Assessments of their performance in maintaining conservation objectives can be a useful indicator of the occurrence of activities that would lead to an avoidable reversal.

Two surveys conducted by the Land Trust Alliance, in 1999 and 2003, reported conservation easement violation rates of 5% per year (vanDoren, 2004). The most common major violation reported was construction of a prohibited structure, primarily residential. The most common violator was a subsequent owner, followed by third parties who include neighbors or other trespassers. The most frequent violation reported in the 2004 survey was dumping of waste or debris. Surface alterations were reported including violations resulting from all terrain vehicle use. Properties adjacent to public land where ATV use is permitted were more likely to report violations related to all terrain vehicle use. An update to the 2004 survey in 2008 found nearly half of surveyed land trusts had reported at least one legal challenge or violation of any size or significance, including both easements and lands owned by land trusts (Rissman & Bustic, 2011). Major violations or legal challenges (greater than $5,000 in legal or staff expenses) were found to be increasing over time. Similar to the earlier reports, the majority of legal challenges for conservation easements (16 of 25) involved second-generation landowners.

Over half of all easements in the United States are held by the federal or state governments, with the NRCS holding 2.28 million acres of easements (The Conservation Registry, 2012). The NRCS administers numerous easements through the Wetland Reserve Program, Grassland Reserve Program. A national monitoring framework is being developed to monitor biological outcomes and easement compliance, but no statistics have yet been reported (University of Tennessee, 2011). The U.S. Fish and Wildlife Service (US FWS), which actively purchases and accepts donated easements into the National Wildlife Refuge System, held easements on 3.43 million acres of land as of September 30, 2011(US FWS, 2011). Over 76% of total refuge easements held by the US FWS are located in North and South Dakota, part of the Mountain Prairie Region, Region 6. Historically, North Dakota has had a high rate of landowner compliance according to the US FWS.

6.6.4.2 Crediting and Reversals

The relatively short time period where crediting would be expected (i.e., 20 to 30 years) for AGC and CCG project activities, contrasts markedly with the 100-year timeframe established by the Reserve as a crediting period for forestry projects. This 20- to 30-year duration would be expected based on the timeframe during which SOC stocks are assumed to reach equilibrium under the new land management system. A Project Implementation Agreement that required remediation could restore some of the carbon lost, but there would be a temporal difference between when the additional carbon would be re-sequestered relative to the loss and the initial crediting.   The authors of this Issue Paper discourage the adoption of 100-year contractual monitoring obligations and liability for AGC and CCG project activities. Although the Reserve's desire to ensure the permanence of sequestered carbon, it is our expectation that most, if not all, landowners would be unwilling to commit to 100-year carbon monitoring and liabilities when crediting could only reasonably be expected for a small portion of that timeframe and since agricultural management decisions are generally not based on the same multi-decadal outlook as forestry activities are.

A buffer pool based on the likelihood of a reversal and the extent of net system loss that would be realized following restoration could account for both the likelihood and extent of a reversal to account for project non-permanence. A temporal discount could also be used, if desired. The likelihood of reversal could be determined through a chart or table, taking consideration of entity holding the easement/agreement, potential economic returns to project and baseline activities, strength of property laws, ease of monitoring, etc. The extent of a reversal for AGC projects could be calculated as the difference between measured SOC and the maximum potential as a CCG project. For CCG projects, it would be assumed that the maximum potential, or steady state condition, could be re-established.

The potential for reversal of project soil carbon benefits where the surface and mineral rights are held by separate parties is a possibility, especially in states with limited ability and recourse of the surface owner to control mineral development activities. A simple requirement to circumvent these scenarios would be a requirement that the surface owner also have a share in the mineral rights of the property. As these rights are typically severed in most grassland dominant geographies, such a restriction could greatly impact project uptake.

An alternative approach employed by the Internal Revenue Service (IRS) Tax Code in dealing with mineral development as it affects the value of donated conservation easements could be insightful for an AGC or CCG protocol. In the US, landowners may be eligible for a tax credit if they donate a perpetual conservation easement that maintains a conservation benefit on their property, e.g., open space or wildlife habitat. IRS Tax Code stipulates surface mining cannot occur on a donated easement, but mining methods that have a limited and localized impact and that do not irreversibly damage the conservation benefits of the property may be allowed (Jones et al. 2009). Alternatively, the surface owner in a split estate is eligible for a tax deduction where they can demonstrate that the probability of surface mining is essentially zero. This can be demonstrated through a mineral remoteness assessment, sometimes referred to as mineral report, performed by a certified geologist that indicates there are either no commercially important minerals present on the property or that their extraction is not commercially viable (Jones, et al., 2009).

It could also be considered conservative to use a deduction of surface area impacted from the most likely mineral development activity. As the footprint of these activities that would impact SOC stocks is relatively small, this approach may strike the appropriate balance considering the rights of most surface owners. Similarly, landowner participation in an AGC or CCG project would be expected to be severely limited if the Project Implementation Agreement prohibited the siting of wind towers on project lands. The economic returns to leasing rural land for wind energy development greatly exceed those of traditional economics activities and would be greater than anticipated returns to AGC or CCG activities. A wind development assessment similar to the mineral report could be employed to account for reversals associated with these activities.

6.7 Potential Co-Benefits and Negative Effects

Project activities, both AGC and CCG, can improve various environmental metrics relative to baselines of cultivation or other development. The extent of the environmental benefits will depend on additional management practices and the location of the Project Activity, but positive wildlife, water quality, and pollination services produced as co-benefits to carbon storage should be possible in most geographies. Wildlife, particularly grassland birds, could derive multiple benefits for various life-history needs from AGC and CCG Project Activities. Perennial grassland land covers can also significantly reduce surface erosion, sedimentation and nutrient loss to regional waterways and help sustain pollinator populations, such as native bees. Grassland management practices have the potential to generate negative environmental impacts and these are reviewed in greater detail in this chapter as well.

6.7.1 Potential Co-Benefits

6.7.1.1 Wildlife

The Prairies of the Great Plains, Montane grasslands, the Palouse, California's annual grasslands and other grassland are unique ecosystems that historically supported an array of wildlife. Historic conversion to cropland and suburban development and the resulting fragmentation of remaining habitat have negatively impacted wildlife. In crop dominant landscapes grass cover provides important refuge for a host of species. The avian community is the most widely studied wildlife community, and there is much research demonstrating the benefit of native and restored grasses on grassland bird species, particularly in the Midwest and Great Plains. The avian community of the Great Plains is particularly important as 330 of the 435 bird species that breed in the United States have been observed to nest in the Great Plains (Samson & Knopf, 1994), and breeding success is the greatest factor on avian population levels. Research on the effects of grasslands , both range and those established under the Conservation Reserve Program, on other forms of wildlife, including the status of grassland –supported wildlife, remains sparse (Jones-Farrand, et al., 2007; Krausman, et al., 2011). Mayeux (2000) estimated of US grazing animal populations of 20 million deer, 500,000 pronghorn antelope, 400,000 elk and 55,000 wild horses and burros. Wildlife benefits could be maximized with selective targeting of reestablished grasslands when located near existing blocks of grasslands, creating buffers or migration corridors (McLachlan et al. 2007).

6.7.1.2 Pollination

Bees provide important pollination services to both wild plants and the species that depend upon them, and also commercially grown crops. In North America, there are over 4000 species of native bees (Spivak, et al., 2011). In addition to our native bees, globally managed honey bee hives have increased 45% while bee-pollinated crop production has increased 300% (Spivak et al. 2011). The economic benefit of these cumulative pollination services provided by bees has been valued at $20 billion a year in North America alone (Spivak, et al., 2011). Despite the importance of bees, commercial honey bee populations have faced unexpected declines from Colony Collapse Disorder while native bee diversity has significantly decreased (Spivak, et al., 2011). Although research has been unable to identify a single factor driving these losses, it is likely that multiple interactive factors, including habitat fragmentation and loss, are responsible (Spivak, et al., 2011).

The loss and fragmentation of habitat is frequently cited as a contributing factor for declines in bee diversity (Spivak, et al., 2011; Kevan, 1999; Hatfield, et al., 2012). Bees require a heterogeneous mosaic of plants, such as rangelands and other diverse grasslands that contain an array of grasses and forbs for foraging and also other lifecycle needs (Black, et al., 2011). Monotypic landscapes that offer foraging opportunities for only brief periods and are deserts for most of the year, a common landscape as agriculture has become more intensive, is not sufficient to maintain native or honeybee populations. Pollinator surveys in California have shown that the chaparral community has the largest diversity of bees per unit area of any ecosystem type and plant community. (Black, et al., 2011). In addition to foraging, grassland can also provide important nesting habitat as 70% of bee species nest in the ground (Black, et al., 2011). Tillage, mowing, grazing and fire can are most common threats to pollinator nests.

6.7.1.3 Water

Agriculture is a common non-point source of pollution in many U.S. waterways(US EPA, 2009). Surface runoff, subsurface lateral flow, leaching, and volatilization and atmospheric deposition are common transport pathways for agricultural pollution to enter water resources (Blanco-Canqui & Lal, 2008). Grass cover from either AGC or CCG projects, managed appropriately, can contribute to the prevention of non-point source pollution addressing each of the three prevention strategies: source reduction, retarding transport, and interception of pollutants.

6.7.2 Potential Negative Impacts

As demonstrated in the previous sections, grass cover from either AGC or CCG can provide numerous environmental benefits relative to a baseline of cultivation or other development. Negative environmental impacts from AGC or CCG project activities are possible, but would be attributed to management practices implemented on the grass cover, such as over-grazing, haying at inopportune times, or improper fire management, rather than the change in land cover. The magnitude and opportunity for negative effects is not equivocal across practices, locations, or even time as the literature demonstrates conflicting impacts from the same practice at the same location but conducted in different years (Krausman, et al., 2011).

Introduction of exotic plant species under CCG project activities, and their spread to adjacent pastures or rangeland if not controlled, could adversely affect wildlife and soil qualities.

Grassland management practices can be detrimental to pollinators, such as mowing when flowers are in bloom or overgrazing in environments. Grazing can negatively affect foraging opportunities and habitat needs, especially in areas where the native taxa did not evolve with grazing pressure such as the American Southwest (Debano, 2006).

6.8 Monitoring and Verification Considerations

6.8.1 AGC

For AGC projects, with-project SOC stocks are not expected to substantially change from initial stock levels over time. Field sampling to determine SOC stocks for AGC projects on an ongoing basis would therefore be unnecessary. However, unsustainable management and on-going degradation may cause a continuous decline in carbon stocks under the with-project scenario. Therefore, the Reserve may consider including monitoring requirements to confirm the sustainability of rangeland management practices in order to avoid significant declines in carbon stocks on the grassland. An example of such monitoring could include documentation that the area of bare ground is not increasing from year to year.

Optionally, the Reserve may also consider allowing projects that not only avoid conversion but also change land management in such a way that carbon sequestration is increased compared to pre-project conditions. These projects could be allowed to claim carbon credits for the additional carbon sequestration. Under this scenario, on-going SOC quantification should be required. The carbon accounting for this scenario is likely to be substantially more complex than accounting for avoided conversion only.

6.8.2 CCG

For projects that convert croplands (back) into grasslands, the SOC stocks are expected to increase with project implementation. This increase in SOC stocks must be quantified. It must be considered that SOC stocks change very gradually. It does not make sense, therefore, to require SOC sampling more frequently than every 5-10 years. Since project proponents will want to generate carbon offsets more frequently than this period, empirical or process-based models are the suitable choice. A limited number of field measurements at key locations and time points could assist in verifying the estimated SOC stock changes.

6.8.3 Aggregation

Both AGC and CCG project types are expected to rely heavily on aggregation for achieving large scale emission reductions.   The authors believe the Ducks Unlimited AGC project in the Prairie Pothole region offers an excellent example of the type of project activities likely to be pursued. In contrast to other conservation programs that landowners may have experience with, carbon projects are likely to involve significantly higher transaction costs related to project development, documentation, measurement, reporting, and verification, particularly if the commitment for pursuing a project extends beyond the timeframe of typical conservation program contracts.

Recognizing that these transaction costs may critically determine the viability of AGC and CCG project activities in general, the authors strongly encourage the Reserve to evaluate hypothetical projects during the development of monitoring and verification rules to ensure they would introduce costs that would render most projects unviable.

7 Conclusion

Both AGC and CCG project types provide real opportunities for achieving a meaningful level of emission reductions in terms of national GHG mitigation and in terms of currently approved offset project types.

The technical potential for emission reductions through AGC are relatively clearly defined by the amount of land currently being converted from grassland to cropland each year. This yields an estimate of 0.2 to 2.0 million tCO_2e that could be achieved through AGC activities each year. The additional potential for setting aside cropland through CCG activities is comparatively much harder to quantify. Depending on the definition used for degraded land, as much as 45% of the US cropland area could qualify for CCG activities. On this landbase, approximately 130 million to 1.2 billion tCO2e could be generated each year.

Both AGC and CCG activities are currently incentivized by federal conservation programs such as the Grassland Reserve Program and Cropland Reserve Program. It is currently unclear how much additional AGC and CCG activities could reasonably be incentivized by carbon offset funding above and beyond these federal programs. Given the relatively massive scale of the CRP (>28 million acres enrolled, enrollment capped at 32 million acres) in comparison to the GRP (0.1 million acres enrolled, enrollment capped at 2 million acres), it is expected that a greater scope may exist for conserving grasslands acres through AGC carbon projects since government funding does not dominate this conservation activity so strongly. In general, we expect that the viability of an AGC and/or CCG offset project protocol would be strongly related to the comparability of carbon project requirements to CRP and GRP contracts which may offer higher value at less risk to landowners.

CCG and AGC activities would impact a variety of carbon pools and GHG sources and sinks, but the primary source of emissions reductions is clearly defined by the conservation and enhancement of soil carbon stocks. Quantifying baseline and with-project SOC stocks is achievable with current methods, but caution must be taken in the development of offset project protocols to minimize the cost to projects that would be induced by field sampling of soil carbon. Approaches such as IPCC land use factors, regional soil surveys, and the use of soil carbon models are encouraged to limit the need for soil carbon quantification via field measurement.

These biological carbon pools are also subject to potential reversal through natural and anthropogenic disturbances, but a variety of policy solutions, such as a buffer pool approach, should be well-equipped to manage these risks. Both CCG and AGC activities also carry a substantial leakage risk, particularly due to indirect land use change. Building from a review of literature evaluating land use effects from biofuel policies, it is expected that anywhere from 20-60% of emission reductions achieved by AGC and CCG activities may be leaked through indirect land use change and market effects. This is a significant discount for these project types, and the full effect of such a large adjustment to a project's emission reductions in terms of the potential to inhibit project development is not immediately clear.

In general, the authors of this Issue Paper believe aggregation will be critical to successfully implementing and scaling up AGC and CCG carbon projects. Although many landowners may exist with sufficient land to justify the costs of project development and verification, these landowners are expected to be a small minority of the potential area that could be conserved through a successful AGC or CCG aggregation program.

Having reviewed the land use trends, carbon accounting, and policy considerations, the authors encourage the Reserve to consider the development of an offset project protocol covering these two project types. AGC is likely to be the most intuitive and suitable to scaling up due to lower anticipated costs and less competition or potential conflicts with federal conservation programs, and may reasonably be positioned as the primary activity targeted by a new protocol. Although the technical potential for CCG is less clear and several accounting practices are fundamentally distinct from AGC, the authors believe a protocol could reasonably be developed to cover both of these activity types. It is unlikely that localized conversion rates could be used to establish a uniform threshold for common practice, but project-based land appraisals are likely to be an indispensable tool for evaluating project additionality and baseline scenarios.

8 Appendix A: Land Type Definitions

8.1.1 Grasslands

While most sources accept that rangelands are a broad land use category that encompass other systems across different climates, soils, and species (Table 21), grasslands are most clearly defined by the presence and usual domination of grasses with little or no tree cover (United Nations Food and Agriculture Organization, 2012). However, though grasses are the dominant vegetation, grasslands can also include legumes, forbs, and other vegetation (Claassen, et al., 2011b). Grasslands typically have minimal tree and shrub cover, though wooded grasslands may have up to 40% tree and shrub cover (United Nations Food and Agriculture Organization, 2012). Notably, existing protocols such as the Verified Carbon Standard define grasslands to also include managed rangeland and pastureland that are not cropland and where grazing dominates (VCSA, 2012). Grasslands may also include a variety of other ecological systems including savannas, woodlands, shrublands, tundra, prairies, and steppes (WRI, 2012). The interchangeability often encountered in these terms demonstrates the lack of standard definitions for grasslands (Keeler-Wolf, et al., 2007). As such, it is necessary to determine the appropriate boundaries and definitions for the project.

8.1.2 Rangelands

Though grasslands are often used interchangeably with rangelands, rangelands necessarily involve some type of livestock or grazing animal. Though rangeland definitions vary, most state that rangeland species composition is similar to grasslands- composed primarily of native grasses, forbs, or shrubs. By such definitions, many rangelands can be found in varying systems including savannas, wetlands, deserts, tundra, chaparral (NRCS, 2009b), meadows, and marshes (NRCS, 2011d). Furthermore, many existing rangeland definitions acknowledge that while native grasses are the dominant species, introduced species (i.e., crested wheatgrass) are within accepted rangeland definitions if they are managed like native species (CCX, 2009; Mitchell, 2010). The presence of livestock indicates that rangeland definitions often contain a human dimension implied by management associated with grazing regimes. The Chicago Climate Exchange suggests that rangelands are managed through livestock rather than agronomic inputs such as fertilization, mowing, or irrigation (CCX, 2009). The management associated with rangelands is important to define in order to distinguish rangelands appropriately from croplands or pasture as appropriate. The reference to other land-use types focuses the attention on the inherent diversity of rangelands. The broad land-use types associated with rangelands including grasslands, savanna, shrubland, desert, tundra, alpine, marsh, and meadow ecosystems may be challenging for the quantification of emission reductions and will require additional definitions if they are to be included within the project.

8.1.3 Pasture and Native Pasture

Though pasture is used interchangeably with grazing land, it is generally recognized as different because it involves more direct human management referred to by the NRCS as “cultural” treatments (NRCS, 2011d).

Key to this definition is the use of cultural elements such as "*human inputs of labor, material, and skill to raise a crop*" in the definition of a pasture.[20] Notably, grazing intensity, duration, and distribution are not considered cultural treatments by the NRCS. Given these inputs, pasture may be considered to more closely resemble a cropping system than a native grazing system.[21] The US Forest Service defines pastureland as non-cropland and woodland pasture that may have had lime fertilizer or seed applied or that have been improved through irrigation, drainage, or control of weeds and brush (USFS, 2004). The NRCS also acknowledges that many rotational pasture systems are part of cropland rotations (NRCS, 2011d). The United Nations Food and Agriculture Organization (2012) confirms the similarity of pasture to cropping systems by noting that pasture, "in well -watered areas it may replace natural grassland, often in association with crop production." Given that pastures may have more in common with cropping systems than with more extensive grassland systems, and may be more GHG intensive than rangeland systems (e.g., Bhandral, et al., 2007) the Reserve may want to consider to exclude the most intensively managed pasture systems from the applicability conditions of AGC projects or exclude the conversion to such pasture systems from CCG projects.

This is particularly true since there are not universal definitions for pasture. Though the NRCS draws clear lines between management associated with rangelands and pasture, the USDA Crop Insurance Program (2012) calls it "*means a community of plants composed primarily of native plants grown for grazing*" while the USDA National Organic Program's Pasture Rule defines pasture as "*land used for livestock grazing that is managed to provide feed value and maintain or improve soil, water, and vegetative resources*" (USDA Agricultural Marketing Service, 2010).

One option for the Reserve is to distinguish between native pasture and traditional pasture systems. Per the NRCS, they clarify that native pasture systems do not receive the cultural management typical of traditional pasture systems; instead grazing management principles shape native pastures (NRCS, 2011d). This distinction in many ways aligns native pasture more closely to rangelands and grasslands than to pasture as defined by NRCS. As a result, native pasture systems can simplify emission reduction calculations by avoiding complications associated with the use of fertilizers, irrigation and other potential inputs.

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[20] Note that the NRCS (2011d) defines cropland as "land used for the production of cultivated crops, including forage crops, and harvested primarily by human labor and equipment. As a secondary use, cropland can be grazed by livestock."

[21] The NRCS defines forage croplands and pasturelands together as "agricultural lands devoted, entirely or partially, to the production of introduced or native forage crops for livestock feeding. They receive cultural treatment to enhance forage quality and yields. The livestock raised on these lands may be pastured, be confined and fed stored forages, or be fed by both methods… On forage producing lands, they include at least one of the following practices: clipping, crop residue management, crop rotation, drainage, fertilization, irrigation, land clearing, mechanical harvest, pest control (e.g., brush, diseases, insects, and weeds), planting, rock picking, selection of new species and/or cultivars, soil amendment applications (e.g., compost, gypsum, lime, and manure), and tillage.

8.1.4 Operationalizing Grazing and Rangeland Definitions

The diversity of definitions to encompass grassland and rangeland systems makes it challenging to operationalize what grazing is for the purposes of this protocol. However, many similar components among definitions exist. Some of the most common elements used in the description of grasslands, rangelands, and native pasture are:

- Presence of grasses, grass-like species, shrubs, forbs (herbaceous plants) and trees in varying amounts. The introduction of species is possible as long as the system is managed through same techniques as for native grassland systems
- Presence of minimal management to differentiate grasslands from croplands and pasture lands.
- Primarily used for grazing
- Potential alternative land uses including conservation or recreation
- Recognition that rangeland may encompass many ecosystem types including savannah, prairie, deserts, tundra, meadows, and alpine regions

Importantly, most definitions define grassland, rangeland, and native pasture systems by the presence of particular vegetation including grasses, shrubs, forbs, and trees. This vegetation can be present in various amounts depending on the ecosystem in which it exists. A prairie or grassland will likely have few trees while a savannah may have many.

The second most important aspect of a definition is the extensive nature of the management. Minimal strategies related to species or grazing animal management are commonly accepted on rangelands, while more intensive management is not. As mentioned earlier, the presence of "cultural management" typical of traditional pasture is one defining aspect. Most definitions mention that rangeland systems are predominantly for grazing purposes but that recreation or conservation practices may also occur. Finally, there was agreement that introduced species are often possible in rangelands but that they are managed using strategies common in native systems.

The Reserve will need to strongly consider the way they define grassland and related terms, as it will necessarily draw the boundaries for the land use types and practices relevant for the protocol. Merely defining grazing will likely not suffice, as there are several pasture definitions that use the term grazing to describe such systems. Livestock graze on both pasture and rangeland; however, it is the management practices that define these systems and will ultimately influence whether they can participate in a Reserve protocol. Pasture-based systems that use intensive management such as fertilization, irrigation, and particularly "pasture renewal" (replacing grass every few years) and forage cropping for livestock grazing may have significantly different GHG emissions impacts than native pasture or rangeland systems that are continuously covered. For verification purposes that are consistent with the Reserve's strategy to use standardized tests, it may be prudent for the Reserve to prohibit practices like those often associated with "cultural management" as the USDA defines it. However, it is also likely that this may eliminate a significant number of landowners, particularly in the Northeast or Midwest that may be pasture-based systems. To ensure that additional and real emissions are prevented through avoided conversion of grasslands it may be necessary for the Reserve to consider whether tillage associated with forage crops or any type of grass system be allowed.

Table 21: Definitions for rangelands, grasslands, and related terms

Source	Definition	Key Components
<u>Offset Program Standards</u>		
VCS Standard Definitions	**Grasslands**: Areas dominated by grasses with a density of trees too low to meet an internationally accepted definition of forest, including savannas (i.e., grasslands with scattered trees). Grasslands also include managed rangeland and pastureland that is not considered as cropland where the primary land use is grazing, and which may also include grass-dominated systems managed for conservation or recreational purposes. **Shrublands**: Areas dominated by shrubs, with a density of trees too low to meet an internationally accepted definition of forest, including chaparral, scrubland, heathland and thickets.	Presence of grasses, shrubs, trees, "managed rangeland and pastureland". Not cropland. Primary land use grazing but also may be conservation or recreation.
CCX Sustainably Managed Rangeland Soil Carbon	**Rangeland:** (Per NRCS) "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, Rangeland supports native vegetation that is extensively managed through the control of livestock rather than by agronomic practices, such as fertilization, mowing, or irrigation. Rangeland also includes areas that have been seeded to introduced species (e.g., crested wheatgrass) but are managed with the same methods as native Rangeland. In most cases, Rangeland refers to lands in the western part of the U.S., while the more general term "Grazing Lands" is used in regions east of the Mississippi River. The use of the term Rangeland in this Protocol is a land use designation and not a geographic designation. Land that fits the above NCRS definition of Rangeland may be eligible for CCX Rangeland Soil Offsets whether it is nominally referred to as Rangeland or Grazing Land provided that appropriate crediting rates can be established by CCX.	Presence of native grasses, forbs, shrubs. Suitable for grazing. "Extensively managed" through livestock control. Introduced species possible but use same management as native rangeland.

Source	Definition	Key Components
Ducks Unlimited	**Native grasslands:** Dominated by diverse, endemic flora and fauna that have evolved since the glaciers retreated some 10,000 years ago. Prairie plants are adapted to grazing. Native grazers such as bison helped maintain diverse prairie habitats by altering the vegetation height and density. These animals grazed at different intensities and frequencies, creating patches of heavily to lightly grazed prairie. This patchiness provided different habitats for various plant and animal species. Currently, most native grasslands are used for annual grazing with only modest improvements for fencing and livestock water.	Native flora and fauna. Adapted to grazing. Presence of grazing. Habitat provider. Annual grazing with modest improvements (water, fencing). Recreation opportunities.
<u>Government Programs</u>		
Natural Resources Conservation Service (NRCS)	**Rangeland:** 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, rangeland supports native vegetation that is extensively managed through the control of livestock rather than by agronomic practices, such as fertilization, mowing, or irrigation. Rangeland also includes areas that have been seeded to introduced species (e.g., crested wheatgrass) but are managed with the same methods as native rangeland.	Presence of native grasses, forbs, shrubs. Suitable for grazing. "Extensively managed" through livestock control. Introduced species possible but use same management as native rangeland.
NRCS National Resources Inventory (NRI)	**Rangeland:** A Land cover/use category on which the climax or potential plant cover is composed principally of native grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species that are managed like rangeland. This would include areas where introduced hardy and persistent grasses, such as crested wheatgrass, are planted and such practices as deferred grazing, burning, chaining, and rotational grazing are used, with little or no chemicals or fertilizer being applied. Grasslands, savannas, many wetlands, some deserts, and tundra are considered to be rangeland. Certain communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also included as rangeland. **Pastureland:** A Land cover/use category of land managed primarily for the production of introduced forage plants for livestock grazing. Pastureland cover may consist of a single species in a pure stand, a grass mixture, or a grass-legume mixture. Management usually consists of cultural treatments: fertilization, weed control, reseeding or renovation, and control of grazing. For	Rangeland: grasses, grass-like plants, forbs, shrubs. Introduced species managed like rangeland. Practices include grazing, burning, chaining, rotational grazing. Little or no chemicals, fertilizer. Multiple ecosystems can be rangeland. Pasture: introduced plants for grazing. Management that includes cultural treatments.

Source	Definition	Key Components
	the NRI, includes land that has a vegetative cover of grasses, legumes, and/or forbs, regardless of whether or not it is being grazed by livestock.	
NRCS Range and Pasture Handbook	**Rangeland:** Rangeland is a kind of land on which the historic climax vegetation was predominantly grasses, grass-like plants, forbs or shrubs. Rangeland includes land revegetated naturally or artificially to provide a plant cover that is managed like native vegetation. Rangelands include natural grasslands, savannas, most deserts, tundra, alpine plant communities, coast and freshwater marshes, and wet meadows. **Pasture:** Grazing land permanently producing introduced or domesticated native forage species receiving varying degrees of periodic cultural treatment to enhance forage quality and yields. It is primarily harvested by grazing animals…pastureland does not include native or naturalized pasture that is permanent pastureland receiving no recent cultural management. Pastureland also does not include rotational pasture that is part of a cropland rotation. **Native Pasture:** Native and naturalized pasture are defined as forest land and naturalized open areas other than rangeland that are used primarily for the production of forage for grazing by livestock and wildlife. Overstory trees, if present, are managed to promote naturally occurring native and introduced understory forage species occurring on the site. These lands are managed for their forage value through the use of grazing management principles. These lands do not receive the cultural management received by pastureland…Native and naturalized pasture may be virtually free of tree growth or may have a partial, or rarely, a full stand of trees.	Rangeland: presence of grasses, forbs, shrubs. Managed like native vegetation (whether or not it is). Includes a variety of different regions and areas that can be grasslands. Pasture: grazing present. "cultural treatment". Harvested by grazing animals. Native Pasture: Forest land, Open areas. Used for forage and grazing. Management to maintain forage species. No cultural management. Can have presence or absence of trees.
USDA Crop Insurance Program	**Grazing Land:** Established acreage of forage on land suitable and intended for grazing by livestock. Acreage that is so steeply sloped, too far from water sources, etc. such that livestock would not normally physically graze such acreage, is not considered grazing land. **Rangeland:** Native pasture on which livestock graze. **Pasture:** land that is used for haying or grazing as a source of forage for livestock”	Grazing, haying, forage present. Grazing intended.

Source	Definition	Key Components
USDA National Organic Program	**Pasture:** Land used for livestock grazing that is managed to provide feed value and maintain or improve soil, water, and vegetative resources.	Livestock present. "Managed".
US Forest Service	**Pasture:** (paraphrased) non cropland and woodland pasture that may have had lime fertilizer or seed applied or that have been improved through irrigation, drainage, or control of weeds and brush.	Presence of "cultural management".
UNESCO	**Grassland:** land covered with herbaceous plants with less than 10% tree and shrub cover.	Presence of herbaceous plants. Little tree and shrub cover.
<u>Non- Profits, Academic</u>		
Society for Rangeland Management	**Rangeland:** Rangeland is a 'natural' vegetation complex dominated by grasses, grass-like plants, forbs, and/or shrubs. Thus by definition, rangelands include indigenous grasslands, savanna, shrubland, desert, tundra, alpine, marsh, and meadow ecosystems as well as introduced pasture systems, such as crested wheatgrass, that are managed as natural ecosystems.	Presence of grasses, forbs, shrubs, wide variety of systems can be grasslands including systems managed naturally.
World Resources Institute	**Grasslands:** Terrestrial ecosystems dominated by herbaceous and shrub vegetation and maintained by fire, grazing, drought and/or freezing temperatures. According to this definition, grasslands encompass not only non-woody grasslands but also savannas, woodlands, shrublands, and tundra."	Presence of herbaceous vegetation, shrubs. Maintained through climate conditions and events. Wide variety of systems can be grasslands.
Keller-Wolf et al. 2007	**Grasslands:** composed primarily of annual and perennial grasses and broadleaf herbaceous plants often called "forbs" or wildflowers. Many grasslands occur as openings or as large islands called meadows within forested areas. **Steppes:** Grasslands with varying amounts (20-80%) of woody shrubs present. **Savannah:** Grasslands with trees scattered evenly throughout.	Presence of annual, perennial grasses, broadleaf herbaceous plants (forbs), wildflowers. May be meadows. Other types of similar systems have trees.
Lincoln et al. 1998	**Prairie/Steppe:** Tree-less plain dominated by grasses and forbs found in moderately dry temperate regions.	Presence of grasses and forbs.
Immel et al. 2012	**Prairie:** Usually dominated by grasses but can also be populated with a number of species including forbs, mosses, lichens, sedges, rushes, and shrubs.	Presence of grasses, forbs, mosses, lichens, sedges, rushes, shrubs.

The National Resources Inventory (NRI) provides data on how non-federal U.S. rural land is utilized and the environmental condition of the land. NRI measures land cover in total area by state and in total acres by class and subclass. It also provides more specific data for cultivated and non-cultivated cropland, rill and wind erosion by state and land use type (NRCS, 2009b). The National Land Cover Database is coordinated through the Multi-Resolution Land Characteristics Consortium made up of federal agencies. The consortium has mapped the entire United States and

Puerto Rico, with most recent data from 2006 including 16 land cover classifications applied at a spatial resolution of 30 meters (Fry, et al., 2011; Multi-Resolution Land Characteristics Consortium, 2012).

Table 22: Land use and land cover database definitions

Database	Term	Definition
National Resources Inventory	Cropland	A Land cover/use category that includes areas used for the production of adapted crops for harvest. Two subcategories of cropland are recognized: cultivated and non-cultivated. Cultivated cropland comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hayland or pastureland that is in a rotation with row or close-grown crops. Non-cultivated cropland includes permanent hayland and horticultural cropland.
	Erodibility Index	A numerical expression of the potential of a soil to erode, considering the physical and chemical properties of the soil and climatic conditions where it is located. The higher the index, the greater the investment needed to maintain the sustainability of the soil resource base if intensively cropped. EI scores of 8 or above are equated to highly erodible land.[1]
	Hayland	A subcategory of Cropland managed for the production of forage crops that are machine harvested. The crop may be grasses, legumes, or a combination of both. Hayland also includes land in set aside or other short-term agricultural programs.
	Land Capability Classification	Land capability classification is a system of grouping soils primarily on the basis of their capability to produce common cultivated crops and pasture plants without deteriorating over a long period. Land capability classification is subdivided into capability class and capability subclass nationally.
	Capability Class	The broadest category in the system. Class codes I to VIII indicate progressively greater limitations and narrower choices for agriculture. The numbers are used to represent both irrigated and non-irrigated land capability.
	Capability Subclass	The second category in the system. Class codes **e** (erosion problems), **w** (wetness problems), **s** (root zone limitations), and **c** (climatic limitations) are used for land capability subclasses.
	Marshland	A subcategory of the Land cover/use category Other rural land, described as a non-forested area of land partly or intermittently covered with water and usually characterized by the presence of such monocotyledons as sedges and rushes. These areas are usually in a wetland class and are not placed in another NRI land cover/use category, such as rangeland or pastureland.

Database	Term	Definition
	Palustrine Wetland	Wetlands occurring in the Palustrine System, one of five systems in the classification of wetlands and deepwater habitats (see Wetlands, Cowardin et al. 1979). Palustrine wetlands include all non-tidal wetlands dominated by trees, shrubs, persistent emergent plants, or emergent mosses or lichens, as well as small, shallow open water ponds or potholes. Palustrine wetlands are often called swamps, marshes, potholes, bogs, or fens.”
	Pastureland	A Land cover/use category of land managed primarily for the production of introduced forage plants for livestock grazing. Pastureland cover may consist of a single species in a pure stand, a grass mixture, or a grass-legume mixture. Management usually consists of cultural treatments: fertilization, weed control, reseeding or renovation, and control of grazing. For the NRI, includes land that has a vegetative cover of grasses, legumes, and/or forbs, regardless of whether or not it is being grazed by livestock.
	Rangeland	A Land cover/use category on which the climax or potential plant cover is composed principally of native grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species that are managed like rangeland. This would include areas where introduced hardy and persistent grasses, such as crested wheatgrass, are planted and such practices as deferred grazing, burning, chaining, and rotational grazing are used, with little or no chemicals or fertilizer being applied. Grasslands, savannas, many wetlands, some deserts, and tundra are considered to be rangeland. Certain communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also included as rangeland.”
	Riverine System	All wetland and deepwater habitats contained within a channel, with two exceptions (1) wetlands dominated by trees, shrubs, persistent emergents, emergent mosses, or lichens; and (2) habitats with water containing ocean derived salts. One of the five systems in the classification of wetlands and deepwater habitats
	Row Crops	A subset of the Land cover/use category Cropland (subcategory, Cultivated) comprising land in row crops, such as corn, soybeans, peanuts, potatoes, sorghum, sugar beets, sunflowers, tobacco, vegetables, and cotton.”
	Wetlands	Lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For purposes of this classification wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is non-soil and is saturated with water or covered by shallow

Database	Term	Definition
		water at some time during the growing season of each year.
National Land Cover Database	**Herbaceous**	Areas characterized by natural or semi-natural herbaceous vegetation; herbaceous vegetation accounts for 75% to 100% of the cover. Includes the following:
	Grassland/Herbaceous	Areas dominated by graminoid or herbaceous vegetation, generally greater than 80% of total vegetation. These areas are not subject to intensive management such as tilling, but can be utilized for grazing.
	Sedge/Herbaceous	Alaska only areas dominated by sedges and forbs, generally greater than 80% of total vegetation. This type can occur with significant other grasses or other grass like plants, and includes sedge tundra, and sedge tussock tundra.
	Lichen	Alaska only areas dominated by fruticose or foliose lichens generally greater than 80% of total vegetation.
	Moss	Alaska only areas dominated by mosses, generally greater than 80% of total vegetation.
	Planted/Cultivated	Areas characterized by herbaceous vegetation that have been planted or are intensively managed for the production of food, feed, or fiber; or are maintained in developed settings for specific purposes. Herbaceous vegetation accounts for 75% to 100% of the cover. Includes the following:
	Pasture/Hay	Areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Pasture/hay vegetation accounts for greater than 20% of total vegetation.
	Cultivated Crops	Areas used for the production of annual crops, such as corn, soybeans, vegetables, tobacco, and cotton, and also perennial woody crops such as orchards and vineyards. Crop vegetation accounts for greater than 20% of total vegetation. This class also includes all land being actively tilled.
	Shrubland	Areas characterized by natural or semi-natural woody vegetation with aerial stems, generally less than 6 meters tall, with individuals or clumps not touching to interlocking. Both evergreen and deciduous species of true shrubs, young trees, and trees or shrubs that are small or stunted because of environmental conditions are included. Includes the following:
	Dwarf Scrub	Alaska only areas dominated by shrubs less than 20 centimeters tall with shrub canopy typically greater than 20% of total vegetation. This type is often co-associated with grasses, sedges, herbs, and non-vascular vegetation.

Database	Term	Definition
	Shrub/Scrub	Areas dominated by shrubs; less than 5 meters tall with shrub canopy typically greater than 20% of total vegetation. This class includes true shrubs, young trees in an early successional stage or trees stunted from environmental conditions.
	Wetlands	Areas where the soil or substrate is periodically saturated with or covered with water as defined by Cowardin et al. (1979). Includes the following:
	Woody Wetlands	Areas where forest or shrubland vegetation accounts for greater than 20% of vegetative cover and the soil or substrate is periodically saturated with or covered with water.
	Emergent Herbaceous Wetlands	Areas where perennial herbaceous vegetation accounts for greater than 80% of vegetative cover and the soil or substrate is periodically saturated with or covered with water[2]

1 The new CRP program for highly erodible land requires an EI of 20 or higher.

2 The areas that are periodically saturated with or covered with water would be similar to vernal pool systems.

9 Appendix B: Conversion Rate and Land Value Tables

Table 23: Net state-level annual grassland conversion rates, ac/yr

State	Grassland to Cropland	Shrub/Scrub to Cropland	Grassland to Development	Shrub/Scrub to Development	Grassland to Forest	Shrub/Scrub to Forest
AL	-2,585.47	-6,629.05	830.64	1,425.77	-103,711.05	4,159.98
AR	-76.24	-514.75	122.45	197.22	-39,896.02	16,306.01
AZ	321.94	829.84	312.38	12,115.74	-2,597.43	-30,540.64
CA	1,999.90	1,111.79	13,768.35	3,903.33	-6,836.27	-41,319.92
CO	4,641.82	-304.32	6,932.08	1,664.44	-5,288.59	-33,862.19
CT	-3.34	-0.53	32.56	307.71	-88.16	-21.71
DC	0.00	0.00	0.00	0.00	-0.49	0.00
DE	-4.09	-431.71	4.18	75.08	-737.82	501.81
FL	-2,190.05	169.69	2,082.77	2,298.36	-38,072.83	17,388.00
GA	-1,091.33	-6,578.25	4,287.14	1,174.15	-31,399.70	31,795.12
IA	808.36	184.90	1,250.04	9.16	-195.48	-8.14
ID	-1,785.21	-6,377.43	348.89	390.35	-6,367.91	-38,116.10
IL	46.88	-266.25	1,770.53	66.23	-311.09	-59.25
IN	-141.35	-6.72	584.81	169.86	-233.83	-159.59
KS	5,921.74	79.26	1,891.29	14.28	-1,412.92	0.18
KY	-201.04	-42.43	243.26	14.01	-10,232.86	-1,072.92
LA	-2,984.62	-2,357.65	224.89	566.88	-34,946.63	43,249.77
MA	-15.52	-4.14	238.10	473.88	-289.47	-178.63
MD	-123.21	-428.47	27.75	164.71	-938.77	1,114.82
ME	-155.41	-20.42	35.89	125.39	-25,312.32	-14,448.84
MI	-131.57	-23.71	580.32	68.36	-14,149.63	-5,564.84
MN	272.34	-808.80	736.30	259.98	-5,814.28	-3,283.52
MO	580.94	156.39	165.73	105.01	-7,496.96	-903.19
MS	-1,541.06	-5,088.43	0.93	1,306.26	-89,724.30	5,104.31
MT	835.76	2,242.54	524.09	172.98	-45,518.95	-99,329.43
NC	126.54	1,809.36	2,414.36	1,420.79	-27,495.88	4,975.05
ND	5,490.52	400.04	918.67	12.41	-14.63	-33.49
NE	28,429.40	64.76	1,809.22	2.71	-441.90	-66.01
NH	-19.57	-6.49	41.32	115.96	-3,993.27	-3,128.51
NJ	-6.18	15.12	67.25	404.89	-212.30	1,022.39
NM	4,645.42	1,227.66	812.23	4,925.64	-857.15	-9,456.08
NV	-67.61	628.98	289.20	5,835.54	-3,872.51	-18,544.37
NY	36.83	93.98	168.31	491.18	-544.07	306.59
OH	-12.41	-17.92	693.78	93.58	-335.99	-1,032.00
OK	1,374.49	-91.80	3,785.73	115.91	-19,347.17	-3,416.91

State	Grassland to Cropland	Shrub/Scrub to Cropland	Grassland to Development	Shrub/Scrub to Development	Grassland to Forest	Shrub/Scrub to Forest
OR	611.10	1,025.15	292.00	760.01	-80,199.77	-80,140.84
PA	-95.23	42.30	340.44	544.24	-1,145.55	842.79
RI	68.05	-1.07	37.05	66.36	-18.37	-1.96
SC	-2,416.76	-2,085.70	2,303.96	1,372.35	455.11	191.17
SD	-4,367.34	-82.46	558.39	63.29	-789.90	-3,187.58
TN	-470.76	-489.53	869.52	696.98	-33,890.79	-2,743.73
TX	17,579.13	12,797.47	18,825.87	12,795.47	-43,091.74	-30,156.96
UT	144.65	-733.46	519.82	2,696.13	142.87	-20,722.76
VA	-741.55	346.89	251.75	704.95	-9,955.49	-8,427.02
VT	-201.27	0.49	4.36	10.94	-878.73	-895.89
WA	-15.52	-2,388.56	689.42	1,406.24	-55,603.74	-60,291.64
WI	140.95	-377.45	789.59	213.32	-6,434.81	-3,892.13
WV	-24.82	-12.94	148.25	0.27	-2,637.33	-1,566.01
WY	2,319.13	444.43	455.64	1,626.77	-4,378.55	-14,917.78

Note: Net conversion rates greater (less) than zero imply a loss (gain). For example, a "grassland to cropland" net conversion rate of -38.59 implies that 38.59 acres of cropland converted to grassland.

Table 24: State-level gross annual conversion rates, ac/yr

State	Grassland to Cropland	Shrub/Scrub to Cropland	Cropland to Grassland	Cropland to Shrub/Scrub
AL	166.13	519.96	2,751.60	7,149.00
AR	16.72	61.03	92.96	575.78
AZ	414.59	4,276.96	92.65	3,447.11
CA	8,590.61	3,041.69	6,590.71	1,929.89
CO	4,702.13	888.96	60.31	1,193.28
CT	7.29	6.54	10.63	7.07
DC	0.00	0.00	0.00	0.00
DE	4.00	10.76	8.10	442.48
FL	836.96	2,086.37	3,027.01	1,916.68
GA	2,173.68	1,581.98	3,265.02	8,160.23
IA	895.76	216.92	87.40	32.02
ID	2,497.31	1,318.09	4,282.52	7,695.52
IL	322.29	21.39	275.41	287.65
IN	41.90	38.39	183.25	45.10
KS	5,984.81	140.82	63.07	61.56
KY	61.07	8.72	262.11	51.15
LA	529.83	613.19	3,514.46	2,970.83
MA	0.00	0.89	15.52	5.03
MD	8.54	51.15	131.75	479.62
ME	81.22	52.75	236.63	73.17
MI	564.70	96.74	696.27	120.45
MN	344.36	219.50	72.01	1,028.31
MO	615.72	180.81	34.78	24.42
MS	64.81	161.32	1,605.87	5,249.76
MT	9,182.58	3,689.35	8,346.82	1,446.81
NC	1,501.07	3,144.04	1,374.53	1,334.68
ND	5,833.14	424.37	342.62	24.33
NE	28,505.10	69.65	75.70	4.89
NH	1.07	0.67	20.64	7.16
NJ	20.19	79.75	26.38	64.63
NM	4,765.96	6,052.96	120.54	4,825.29
NV	179.34	2,394.17	246.95	1,765.19
NY	127.08	198.64	90.25	104.66
OH	35.00	37.23	47.41	55.15
OK	1,430.22	138.06	55.73	229.87
OR	1,110.86	2,390.52	499.76	1,365.37
PA	63.78	116.58	159.01	74.28

State	Grassland to Cropland	Shrub/Scrub to Cropland	Cropland to Grassland	Cropland to Shrub/Scrub
RI	69.52	0.00	1.47	1.07
SC	1,008.51	2,295.29	3,425.28	4,380.99
SD	12,829.85	616.30	17,197.19	698.76
TN	79.39	80.46	550.16	570.00
TX	19,094.04	18,147.61	1,514.91	5,350.14
UT	303.08	551.27	158.43	1,284.73
VA	283.73	816.99	1,025.28	470.10
VT	6.00	9.52	207.27	9.03
WA	835.31	2,929.16	850.84	5,317.72
WI	219.55	198.95	78.59	576.40
WV	1.11	0.00	25.93	12.94
WY	3,182.60	1,572.64	863.47	1,128.21

Notes: Gross conversion rates do not include any acreage that converted in the opposite direction

Table 25: State-level average rental rates and Grassland Reserve Program payment rates, $/ac

State	Rental Rate ($/acre) Irrigated Cropland	Non-irrigated Cropland	Pasture	Grassland Reserve Program ($/acre)
AL	246.86	35.70	15.01	8.98
AZ	97.00	25.60	4.02	5.00
AR	144.31	78.48	22.06	11.40
CA	187.15	157.43	41.09	10.16
CO	106.85	74.66	23.51	6.58
CT	275.00	50.83	26.31	15.13
DE		62.38	23.94	13.83
FL	128.03	72.69	21.77	14.26
GA	125.73	71.74	21.87	8.52
ID	154.08	91.15	20.34	8.45
IL	98.27	58.63	20.27	16.50
IN	115.78	47.68	20.67	16.50
IO	76.85	28.37	10.93	16.50
KS	218.73	44.12	17.08	11.49
KY	136.68	87.42	26.14	13.70
LA	177.73	150.46	41.61	10.20
ME	138.50	88.29	38.17	8.78
MD	111.02	50.85	19.08	13.24
MA	100.63	37.22	14.17	14.07
MI	90.64	62.46	19.09	10.47
MN	126.52	66.28	22.08	12.80
MS	137.59	74.89	19.59	9.59
MO	107.78	49.48	18.77	13.59
MT	93.33	47.10	22.48	7.54
NE	71.05	30.23	11.79	12.23
NV	60.00	41.06	25.90	5.00
NH		30.38	15.50	10.70
NJ	124.44	29.91	16.02	14.21
NM	183.75	59.28	24.21	5.42
NY	100.04	48.92	21.26	10.53
NC	230.99	40.40	18.47	10.27
ND	151.87	122.43	36.10	8.23
OH	118.52	112.81	31.72	14.84
OK	110.30	61.42	24.85	8.31
OR	111.27	88.82	22.48	15.07
PA	125.83	67.40	23.04	10.15

State	Rental Rate ($/acre) Irrigated Cropland	Non-irrigated Cropland	Pasture	Grassland Reserve Program ($/acre)
RI				15.00
SC	122.36	60.49	16.31	8.20
SD	104.57	47.41	19.13	10.90
TN	106.94	50.38	22.83	10.69
TX	164.10	73.87	17.64	7.30
UT		87.99	18.86	5.59
VT	71.00	84.60	28.18	12.48
VA	149.01	78.21	24.32	10.59
WA		63.24	17.04	13.21
WV	188.25	46.72	12.23	7.77
WI	72.72	42.92	15.65	11.77
WY	127.17	64.14	19.81	6.82

10 Appendix C: Summary of Selected Policies Relevant for Determining Legal Surplus

10.1.1.1 Conservation Reserve Program (CRP)

The largest federal government conservation program for private lands, CRP provides incentives and assistance for producers to adopt conservation practices. CRP contracts are for 10 to 15 years. Significant data are available from the Farm Service Agency on their website (USDA FSA, 2012a), which can assist the Reserve in determining project types and their implementation across regions. The CRP uses the Environmental Benefits Index (EBI) to rank potential CRP lands to capture their multiple ecological benefits. The index is based on several categories including wildlife habitat benefits, water quality benefits, reduced erosion benefits, benefits beyond the contract period, air quality benefits, and cost (USDA FSA, 2011).

One recent development is that as of February 2012, the CRP has a new initiative for highly erodible cropland. This project aims to protect up to 750,000 acres across the United States. Highly erodible lands (those with an erodibility index of 20 or greater) can receive incentives to plant this land in long-term cover through the CRP (USDA FSA, 2012c). Cover options include 1) native grasses, 2) wildlife cover mixes, and 3) trees, if suitable (USDA FSA, 2012b).

The program currently has an acreage cap of 32 million acres. During the next two years contracts covering 10 million acres are set to expire. Under the 2012 Senate version of the Farm Bill (S. 3240, Agriculture Reform, Food, and Jobs Act of 2012) the acreage cap will be lowered through a multi-year step-down program to 25 million acres by 2016 with a priority for the most highly erodible lands. No more than 1,500,000 acres of grassland would be able to be enrolled in the CRP program at any one time between 2013 and 2017. Should these provisions be passed in the 2012 Farm Bill, these acres would be candidates for an avoided conversion program.

The CRP likely has several policies that would impact additionality and therefore be relevant for a protocol on avoided conversion of grasslands and conversion of marginal croplands to grassland. The new 2012 CRP program for landowners with highly erodible land (an "Erodibility Index" of 20 or greater) (USDA FSA, 2012b) would prevent land enrolled in this program from participating in any Reserve offset program.

10.1.1.2 Farmable Wetlands Program

This voluntary program aims to restore up to one million acres of farmable wetlands and buffers in all states. Producers plant long-term, resource- conserving covers to improve water quality, soil erosion and wildlife habitat on land already enrolled in the CRP. Contracts range between 10 and 15 years. Croplands are eligible for this program, including those that are subject to a natural overflow of a prairie wetland, which potentially could include vernal pools. Conservation practices included in this program are flood prairie wetlands (USDA FSA, 2009).

Land enrolled in the CRP would not be considered additional as this government program would have driven this behavior change. Similarly, CRP contracts and programs that maintain grasslands would also be ineligible. With the Farmable Wetlands Program it may be possible given some of the language and practices that producers with marginal cropland could convert their land to flooded prairie wetlands, which could include grasslands with vernal pools. If this were the case, participation in this program would indicate that the conversion of marginal cropland to grassland was undertaken as a result of the CRP and not the carbon market.

10.1.1.3 Grassland Reserve Program

The Grassland Reserve Program "provides assistance to landowners and operators to protect grazing uses and related conservation values on eligible private range and pasture lands. The program emphasizes support of grazing operations, maintaining and improving plant and animal biodiversity, and protecting grasslands and shrublands under threat of conversion to cropping, urban development, and other non-grazing uses" (USDA, 2010). Producers may enroll under a variety of options including 10-, 15-, or 20-year contracts or permanent easements in perpetuity or the maximum allowed by state law. Eligible Conservation Reserve Program lands whose contracts are within one year of their expiration are given priority for enrollment, with no more than 10%of the total GRP acres each year allocated to expiring CRP lands. Participants must sign form AD-1026 related to planting in highly erodible lands and wetlands. There are no minimum or maximum area requirements to enter into the program and eligible lands include privately owned land including tribal lands. Eligible land types are:

- Grassland, land that contains forbs, or shrubs (including rangeland and improved pastureland) for which grazing is the predominant use; or
- Land located in an area that has been historically dominated by grassland, forbs, and shrubs, is compatible with grazing uses; and
- Land that has potential to provide habitat for animal or plant populations of significant ecological value if the land is retained in the current use and condition of the land or restored to a natural condition or contains historical or archeological properties (including, but not limited to sites, buildings, structures, objects, and landscapes, including properties of traditional religious and cultural importance to an Indian tribe or Native Hawaiian organization (see Section 101(c)(6) of the National Historic Preservation Act (NHPA)) listed in or eligible for listing in the National Register of Historic Places or addresses issues raised by state, regional and national conservation priorities.

Ineligible land includes amongst other things, publicly-owned lands and their subsidiaries, land under active Environmental Quality Incentive Program contracts or land under an existing conservation easement, contract or deed that already provides grassland resource protection. The program specifically prevents the production of crops other than hay (USDA, 2010).

Since the GRP directly provides contracts and easements to protect grasslands and shrublands from cropping, lands currently enrolled in the GRP with these explicit types of contracts would not pass a test for additionality. It would be assumed that individuals participating in the GRP under these types of practices would have undertaken practices as a result of this program and not a carbon market.

However, it may be possible that lands enrolled in the GRP for practices that do not include avoided conversion (i.e., restoration or conservation practices related to biodiversity), or have contracts that are set to expire, could be a Regulatory Surplus. Under these circumstances it could be possible that additional grassland conservation practices beyond those covered during the terms of the GRP contract would be additional.

10.1.1.4 Wildlife Habitat Incentive Program (WHIP)

A voluntary government program reauthorized under the 2008 Farm Bill, WHIP provides technical assistance and up to 75% cost share assistance for producers to establish and improve wildlife and fish habitat. Contracts can last no more than ten years (NRCS, 2012e). Various grassland practices are eligible under WHIP, but eligibility varies by state. For example, in Vermont, the practices relevant to grasslands deal with forage removal, which would not be relevant to the protocol(s) under consideration by the Reserve protocols (NRCS, 2010). If and where WHIP eligible programs provide incentives for landowners to keep their land in grasslands and avoid conversion to cropping systems or other development, lands enrolled in these programs would not meet Reserve additionality requirements.

10.1.1.5 Environmental Quality Incentives Program (EQIP)

EQIP is a voluntary government program that provides funding and technical assistance to producers with contracts up to ten years. Conservation practices that improve soil, water, plant, animal, air and related resources on agricultural land are eligible. There are several relevant practices in EQIP programs that are applicable in certain states. Forage Biomass Plantings are an EQIP practice that allows for existing cropland to be converted to pasture or hay land or for the establishment of livestock grazing forages (NRCS, 2011b; NRCS, 2011c; NRCS, 2012d). In addition, Conservation Cover programs within EQIP can provide funding for producers to plant grasses, legumes, forbs and other species depending on the state.

10.1.1.6 Conservation Stewardship Program

The Conservation Stewardship Program (CSP), a sub-program of EQIP, has a variety of programs that incentivizes producers to manage their lands for native grasses, wildlife habitat, and general natural resource health. Among the CSP eligible practices is prairie restoration for grazing and wildlife habitat that involves restoring/renovating prairie habitats by establishing native vegetation. This project type may allow producers to convert cropland or other lands to native prairies and grasslands. The CSP also includes a conversion of cropped land to grass-based agriculture program that aims to establish mixtures of perennial grasses, forbs or legume species on cropland where annually seeded cash crops were previously grown in monocultures. Similar CSP programs also include pasture and hay planting (establishing native or introduced forage species), range planting (establishment of adapted perennial vegetation such as grasses, forbs, legumes, shrubs, and trees in order to establish a function range ecology), and tree/shrub establishment (establishing woody plants by planting seedlings or cuttings, direct seeding, or natural regeneration) (NRCS, 2011a).

Participation in any of these programs could prevent landowners from eligibility for a Reserve protocol. Participation in other Conservation Stewardship Programs may not preclude protocol participation as these programs would not provide incentives for the avoided conversion or conversion of marginal lands to grasslands.

10.1.1.7 Endangered Species Act (ESA)

Currently in California there are 74 grassland-dependent species (9 vertebrates, 14 invertebrates, and 49 plants) listed as threatened or endangered under the ESA. Section 10 of the ESA lists exemptions, permits, and exceptions to the ESA including the permitting of incidental takes. Under the ESA, the Secretary of Interior can issue incidental “take” permits in conjunction with habitat conservation plans, which can mitigate incidental take of grassland-dependent endangered species. Mitigation must occur to the maximum extent possible (Jantz, et al., 2007). A “take” is defined as, *“[…] to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct*.” “Harm” includes significant habitat modification that actually kills or injures a listed species through impairing essential behavior such as breeding, feeding, or sheltering”(US FWS, 2011b). Many ranchers with endangered species on their property may have to develop these plans, and to the extent that conversion of grassland or marginal croplands would impact upon an endangered species, this may be illegal. If endangered species have been identified on private property, and the ESA dictates that this land must remain in its current form, such properties would not pass additionality requirements as set forward by the Reserve.

10.1.1.8 Clean Water Act (CWA)

Section 404 of the CWA regulates the fill of jurisdictional wetlands. Following a Supreme Court 2001 Decision (SWANCC v Army Corps of Engineers) federal jurisdiction is no longer valid for isolated wetlands and ponds. However, to the extent that vernal pools or wetlands in grassland areas are connected to drainage systems linked with federal land they may still come under CWA jurisdiction. Otherwise these systems may be covered under state or local regulatory bodies (Jantz, et al., 2007). If a wetland in a rangeland system is on land covered under the Clean Water Act, there may be laws in place that would prevent this land from being converted to cropland and would make it non-additional. Current versions of other protocols prevent participation from grasslands on non-forested wetlands (e.g., Dell, et al., 2012a).

10.1.1.9 State Endangered Species Acts

According to the Michigan State University College of Law Animal Legal and Historical Center, only four states (Alabama, North Dakota, West Virginia, and Wyoming) do not have state level endangered species laws. These laws vary amongst states but can provide statues that may influence landowner decision making related to grasslands (Michigan State University College of Law, 2012). For example, the California Endangered Species Act (CESA) extends the protection of endangered plant species to privately owned lands in California. Under the CESA there are 58 grassland-associated species listed and Natural Community Conservation Plans must be obtained for “takes”. Unlike the federal ESA, mitigation of takes is required roughly to be proportional to the take (Jantz, et al., 2007).

Since the CESA particularly covers private lands and there are a significant number of grassland endangered species in California, there are likely many landowners that have take permits. These permits may not preclude a property from participating in the carbon market, but if it dictates that mitigation must happen or that land cannot be converted then this would prevent a landowner from participation, as the land would not pass an additionality test.

10.1.1.10 Williamson Act

Specific to California only, the California Land Conservation Act ("Williamson Act") allows county governments to provide landowners with contracts to preserve agricultural land and open space. Contracts are for ten years and provide landowners with tax incentives (Jantz, et al., 2007) that assess land value based on their actual use, rather than their potential market value. As of 2009 there were approximately 15 million acres of farmland enrolled in the Williamson Act, representing about half of all farmland. The California Department of Conservation also tracks the nonrenewal trends of farmland that has been taken out of Williamson Act contracts. For these landowners, a Reserve protocol may be particularly relevant to prevent conversion of grassland. In recent years, given the economic downturn, there has been a significant rise in nonrenewal acreage, which peaked in 2007 (CA DOC, 2010).

Producers participating in the Williamson Act are stipulated by contracts not to develop their land. As much, they would likely not pass the legal requirement test for the avoided conversion to grasslands. However, they may be able to participate in a protocol to convert marginal croplands to grasslands since this land would remain in farmland. Importantly, the Williamson Act has received significant cuts in recent state budgets and a large increase in non-renewals, indicating that there may be opportunity for producers who did not renew their Williamson Act contracts to participate in a Reserve program that would incentivize them to keep their grasslands in farmland.

10.1.1.11 Rangeland, Grazing Land and Grassland Protection Program

The Wildlife Conservation Board in California implements the Rangeland, Grazing Land and Grassland Protection Act of 2002 (California Public Resources Code § 10330-10344), which aims to protect California's rangeland, grazing land and grassland through conservation easements. The programs stated goals include:

1) To prevent the conservation of rangeland, grazing land and grassland to nonagricultural uses.
2) To protect the long-term sustainability of livestock grazing
3) To ensure continued wildlife, water quality, and watershed and open-space benefits to the State of California from livestock grazing.

Producers participating in this program and in possession of conservation easements will fail to meet additionality requirements and likely be ineligible for participation in AGC.

10.1.1.12 County-Level Planning

Many counties and regional jurisdictions have the capacity to create policies that dictate open space and regional planning. One report specific to California found that of 16 county plans assessed, only nine open space plans, ten conservation plans and four land use plans recognized the ecological significance of grasslands. An additional seven counties had agricultural elements, four of which discussed grasslands and/or rangelands. These plans would be implemented at the county level through zoning ordinances. Some zoning ordinances can specifically allocate land so that it cannot be developed or so that it must stay in open space or agriculture. Jantz et al. (2007) found that nearly 75% of California grasslands were presently zoned for agricultural use or open space. However, if a land owner who had the intention to sell their grassland found out that the land was not zoned appropriately this may influence additionality associated with avoided conversion since the zoning laws are driving their behavior rather than the market.

These county-level zoning level ordinances would only be relevant for conversion of grasslands to non-open space or agricultural use. If the Reserve sets the scope and boundaries of the protocol to avoid the conversion of grassland to other types of agricultural land use or cropping land only, then the county-level planning policies would not apply. The county-level planning policies would likely only be relevant if a private landowner was seeking to develop their land.

11 Appendix D: Synthesis of Existing Methodologies

11.1 Synthesis of Methodology Approaches

11.1.1.1 What are the project activities eligible for crediting?

The methodologies reviewed here contain a varied scope of eligible activities. Several methodologies have been designed to narrowly address a specific project activity or scenario, while others have been defined more broadly to accommodate significant variations in project design. The most common fundamental decision point in terms of constraining project activity eligibility is which land-use types/cover classes are permitted and whether land-use change (and what types) is accommodated in both the baseline and project scenarios.

11.1.1.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The most common limitation in the methodologies reviewed that is relevant to AGC and CCG project types is the prohibition of activities on wetlands or organic soils. This limitation is generally provided due to the recognition that the factors driving SOC dynamics in wetland or organic soils are significantly different from those affecting mineral soils, particularly in terms of hydrology, which affects numerous biological, chemical, and physical processes in soils that relate to carbon storage and turnover (such as non-CO_2 emissions).

Most other limitations on project activities and scenarios are aimed at simplifying accounting for baseline and project emissions and sequestration. One of the most common limitations of this type is the requirement that the current status of the project lands is degraded or degrading. Demonstrating that project lands are degraded and degrading enables a simplified accounting of baseline carbon stocks (typically conservatively assumed to be at steady state). Similar limitations include requirements that pre-project agricultural practices such as grazing will not be displaced by a project to avoid the need to account for market or activity-shifting leakage.

Where projects rely upon models for carbon estimation in the baseline and/or project scenarios, applicability conditions will often limit the scope of potential projects to the activities and environments the models are well suited for simulating. These limitations generally address whether specific models must be used, or, if none are specified, what types of models can be used. These limitations are also designed to confirm that the models application to the project is justified (e.g., to the soil and climate types) through peer reviewed publications or other relevant scientific findings.

11.1.1.3 What are the methods for determining additionality?

Most of the methodologies reviewed here apply a project-specific additionality testing approach, using a step-wise process exemplified by the “Combined tool to identify the baseline scenario and demonstrate additionality in A/R CDM project activities” and the “Tool for the Demonstration and Assessment of Additionality in VCS Agriculture, Forestry and Other Land Use (AFOLU) Project Activities.” All of the methodologies include requirements of regulatory surplus and some justification that approved activities are not common practice.

The only methodologies that currently take a standardized approach to additionality testing are the CCX Sustainably Managed Rangeland protocol and the Proposed VCS Methodology for Avoided Conversion of Grasslands and Shrublands from Planned Conversion. The CCX methodology essentially offers a blanket additionality determination to all project activities that meet the eligibility criteria based on the argument that the project activities are demonstrably beyond common practice as evidenced through current rates of land degradation and adoption of best management practices. The proposed VCS methodology utilizes a standardized financial analysis to demonstrate project additionality in the same manner as that adopted by the Climate Action Reserve for avoided conversion projects in the Forest Project Protocol. Through the use of a formal land appraisal which considers the project and alternative baseline land uses, project activities with a value below a defined threshold relative to the baseline land-use scenario are deemed additional.

11.1.1.4 How are baseline soil organic carbon levels determined?

To avoid the uncertainty introduced through modeling of SOC stocks, many methodologies simplify baseline carbon accounting by applying eligibility criteria that allow the project to assume baseline activities would reduce carbon stocks in soil organic matter and that they can be conservatively excluded from further accounting by being treated as if they were at steady state.

Because most of the methodologies have been designed with non-soil carbon pools as the primary carbon pools of interest, very few require direct field measurement of initial SOC stocks. Those methodologies that do require the estimation of an absolute value for SOC stocks (as opposed to solely the quantification of stock change) at project initiation often rely upon the application of the IPCC SOC equilibrium values which can include default factors for land-use, management activities, and inputs to the land.

Those few methodologies which currently require soil sampling to determine initial carbon stocks prescribe varying sampling protocols ranging from the tool for “Calculation of the number of sample plots for measurements within A/R CDM project activities” to a more prescriptive approach as shown in the Proposed “Module/Tool – Soil Carbon” developed by The Earth Partners and undergoing validation for use in VCS.

11.1.1.5 How is soil organic carbon sequestration measured and quantified?

In general, those methodologies that account for SOC sequestration in the project scenario or avoided emissions from the SOC pool in the baseline scenario follow the accounting guidance established by the IPCC. This approach involves the estimation of equilibrium levels for particular configurations of land-use types, management practices, and inputs to the land. Changes in land-use, management practices, or inputs are assumed to produce a linear transition in SOC stocks from one equilibrium level to another.

For quantifying changes in SOC stocks, several methodologies allow for the use of empirical or biogeochemical process models (e.g., Roth-C, DNDC, CENTURY) to replace the IPCC default factors, transition times, and transition trajectories (e.g., linear versus exponential decay) that are generally expected to be more conservative. These model estimates are typically ground-truthed at project initiation and/or over time with some level of field sampling required.

11.1.1.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

Most of the methodologies reviewed limited the applicability of the methodology to preclude the need to account for leakage due to market pressures or activity-shifting. In general, very few of the methodologies reviewed here permit activities that would be expected to produce indirect land use change (typically by virtue of restrictive applicability conditions regarding pre-project agricultural activities or land use change). Several address activity-shifting leakage, but only two address market leakage with substantial methodological guidance.

Activity-shifting leakage is typically considered as a transfer of pre-project emitting activities outside the project area to a specific and identified location by the same land manager or owner. This type of leakage is often precluded based on eligibility criteria or through other standardized justification such as the attestation required by landowners in the CCX methodology that grazing practices have not been altered outside the project area to compensate for expected decreases in productivity inside the project area. The CDM tool "Estimation of the increase in GHG emissions attributable to displacement of pre-project agricultural activities in A/R CDM project activity (Version 01)" is the current approach used to account for these activity-shifting emissions in several methodologies and relies upon a very simplified metric based on the proportion of the project country that is covered in forest.

Of the methodologies reviewed here, most were not designed to encourage project activities that may be expected to produce indirect land use change (e.g., taking agricultural land out of production, or preventing the conversion of grassland to cropland). The two methodologies most closely related to this type of project activity are the Proposed "Module/Tool – Soil Carbon" developed by The Earth Partners and the Proposed "Methodology for Avoided Conversion of Grasslands and Shrublands from Planned Conversion" developed by Ducks Unlimited, both of which are currently undergoing validation for use under VCS.

Both of these methodologies address market leakage through a consideration of the expected impact the reduced production of crops or other agricultural products within the project area will have on broader market dynamics. This is quantified through consideration of how the market has responded in the past to changes in supply and demand. Both approaches are described in more detail under their respective methodology reviews.

11.1.1.7 How is permanence of soil carbon addressed?

Apart from CDM A/R methodologies and tools, which generally presume the use of temporary offset credits, all other methodologies reviewed here utilize a buffer pool approach coupled with some default or risk-based quantification for the number of buffer credits to be held to mitigate the risk of reversals from biological carbon stocks.

11.2 Review of Approved Methodologies, Modules, and Tools

Methodologies, modules, and tools reviewed in this section have been formally adopted by a carbon offset standard. They include:

- Continuous Conservation Tillage and Conversion to Grassland Soil Carbon Sequestration Offset Project Protocol (CCX) – pg. 120
- Sustainably Managed Rangeland Soil Carbon Sequestration Offset Project Protocol (CCX) – pg. 121
- Combined tool to identify the baseline scenario and demonstrate additionality in A/R CDM project activities (Version 01) – pg. 123
- Estimation of the increase in GHG emissions attributable to displacement of pre-project agricultural activities in A/R CDM project activity (Version 01) – pg. 124
- Tool for estimation of GHG emissions related to displacement of grazing activities in A/R CDM project activity (Version 02) – pg. 125
- Tool for estimation of change in soil organic carbon stocks due to the implementation of A/R CDM project activities (Version 01.1.0) – pg. 127
- Tool for the Demonstration and Assessment of Additionality in VCS Agriculture, Forestry and Other Land Use (AFOLU) Project Activities, v3.0 – pg. 129
- AFOLU Non-Performance Risk Tool (VCS) – pg. 131

11.2.1 VM0017 – Adoption of Sustainable Agricultural Land Management (VCS)

11.2.1.1 What are the project activities eligible for crediting?

VCS Methodology VM0017 takes a broad approach to crediting agricultural land management activities, and allows accounting for any project activities that increase carbon stocks. Specifically:

> *...project activities that reduce emissions in agriculture through adoption of sustainable land management practices (SALM) in the agricultural landscape. In this methodology, SALM is defined as <u>any practice that increases the carbon stocks on the land</u>. Examples of SALM are (but are not limited to) manure management, use of cover corps, and returning composted crop residuals to the field and the introduction of trees into the landscape.* (p. 4, emphasis added)

This methodology is fundamentally tied to carbon stocks rather than sequestration or emission rates, and is thus most closely related to the project activity of converting marginal cropland to grassland (CCG) considered in this Issue Paper. This methodology does not accommodate project activities that only avoid the loss of carbon stocks or those that would reduce emissions from agricultural land management but that do not result in increased carbon stocks. However, it may be applied to projects that avoid SOC losses due to grassland conversion in the baseline scenario so long as the project also anticipates increases in the carbon stocks on the land beyond initial levels.

11.2.1.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The methodology does not limit eligible project activities by geographic location; projects may not be conducted on wetlands, but there are no further restrictions based specifically on soil types (beyond the fact that organic soils typically qualify an area as wetland); project lands must be classified as cropland or grassland. The full applicability conditions of this methodology are:

a) *Land is either cropland or grassland at the start of the project;*
b) *The project does not occur on wetlands;*
c) *The land is degraded and will continue to be degraded or continue to degrade;*
d) *The area of land under cultivation in the region is constant or increasing in absence of the project;*
e) *Forest land, as defined by the national CDM forest definition, in the region is constant or decreasing over time;*
f) *There must be studies (for example: scientific journals, university theses, local research studies or work carried out by the project proponent) that demonstrate that the use of the Roth-C model*[22] *is appropriate for: (a) the IPCC climatic regions of 2006 IPCC AFOLU Guidelines*[23]*, or (b) the agroecological zone (AEZ) in which the project is situated, using one of options presented below:*[24]

   Option 1: *The studies used in support of the project should meet the guidance on model applicability as outlined in IPCC AFOLU 2006 guidelines in order to show that the model is applicable for the relevant IPCC climatic region. The guidance notes that an appropriate model should be capable of representing the relevant management practices and that the model inputs (i.e., driving variables) are validated from country or region-specific locations that are representatives of the variability of climate, soil and management systems in the country.*

   Option 2: *Where available, the use of national, regional or global*[25] *level agroecological zone (AEZ) classification is appropriate to show that the model has been validated for similar AEZs. It is recognized that national level AEZ classifications are not readily available; therefore this methodology allows the use of the global and regional AEZ classification*[26]*.*

   Where a project area consists of multiple sites, it is recognized that studies demonstrating model validity using either Option 1 or Option 2 may not be available for each of the sites in the project area. In such cases the study used should be capable of demonstrating that the following two conditions are met:

[22] For ROTH-C see http://www.rothamsted.bbsrc.ac.uk/aen/carbon/rothc.htm.
[23] IPCC. 2006, 2006 IPCC Guidelines for National GHG Inventories, Volume 4: Agriculture, Forestry, and Other Land Use. Prepared by the National GHG Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan. http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html
[24] The IPCC climatic regions are shown in Figure 3A.5.1 page 3.38.
[25] The agro-ecological zone (AEZ) methodology is standardized framework for the characterization of climate, soil and terrain conditions relevant to agricultural production. Crop modeling and environmental matching procedures are used to identify crop-specific limitations of prevailing climate, soil and terrain resources, under assumed levels of inputs and management conditions.
[26] The details of global agroecological zones classification outlined by Food and Agricultural Organization of United Nations (FAO), Rome, Italy and International Institute for Applied Systems Analysis, Laxenburg, Austria are available at: http://www.iiasa.ac.at/Research/LUC/GAEZ/index.htm

(i) *The model is validated for at least 50% of the total project area where the project area covers up to 50,000 ha*[27]*; or at least 75% of the total project area where project area covers greater than 50,000 ha; and*

(ii) *The area for which the model is validated generates at least two-thirds of the total project emission reductions.*

11.2.1.3 What are the methods for determining additionality?

The methodology requires projects to apply the "Combined tool to identify the baseline scenario and demonstrate the additionality in A/R CDM project activities."

11.2.1.4 How are baseline soil organic carbon levels determined?

Baseline SOC levels for cropland and grassland areas are assumed to be at equilibrium, meaning they remain constant over time if the land use type and management practices do not change. This assumes no additional carbon sequestration in the SOC pool. The specific equilibrium value for baseline SOC levels is calculated using the Roth-C or other approved biogeochemical models. The specific equation applied for estimating baseline SOC levels is:

$$BS_{equil,t} = \sum_{m_C} BA_{C,m_C,t} \cdot SOC_{C,m_C} + \sum_{m_G} BA_{G,m_G,t} \cdot SOC_{G,m_G}$$

Where:

$BS_{equil,t}$	Baseline SOC in equilibrium year t, tC
$BA_{C,m_C,t}$	Baseline areas in cropland with management practice, m_C, year t, ha
SOC_{C,m_C}	Soil organic carbon density at equilibrium for cropland with management practice, m_C, tC/ha
m_C	An index for cropland management types, unitless
$BA_{G,m_G,t}$	Baseline areas in grassland with management practice, m_G, year t, ha
SOC_{G,m_G}	Soil organic carbon density to a depth of 30 cm, at equilibrium for grassland with management practice, m_G, tC/ha
m_G	An index for grassland management types, unitless

Baseline scenarios where grassland cover changes over time (either through conversion of cropland to grassland or vice versa) would thus show baseline SOC levels changing based on the proportion of project lands in each land use type and management practice and the corresponding equilibrium SOC value. However, the applicability condition that project lands are degraded and continuing to degrade requires that baseline SOC levels (i.e., $BS_{equil,t}$) may not increase over time and must either stay constant or decrease. Although this methodology was not specifically designed for avoided grassland

[27] The project area of 50,000 ha is reasonable taking into account the wide range of soil carbon sequestration rates, which depend on climate, soil and land use characteristics. The project area is also influenced by the rates of SALM adoption that are in turn influenced by factors such as farmer awareness to SALM, institutional support and extension systems. Assuming a conservative soil sequestration rate of 0.5 tC/ ha/ yr applied in CDM A/R methodologies, a project of 50, 000 ha is likely to generate 25,000 tC/ ha/yr, and is considered reasonable taking into account the implementation, monitoring and verification costs.

conversion projects, if applied in this context, the baseline SOC equation above would account for the SOC lost from grassland converted to cropland as instantaneously lost to the atmosphere in the year of land use change.

11.2.1.5 How is soil organic carbon sequestration measured and quantified?

This methodology follows the approach described in the 2006 IPCC Guidelines for National GHG Inventories[28]. Soil organic carbon levels at equilibrium under a specific land use type and management practice are estimated using a biogeochemical model (Roth-C). Changes in SOC levels due to a transition to new management practices or land use types are assumed to change linearly over a defined transition period. Field collection of soil samples for SOC analysis is not required. At equilibrium, project SOC levels ($PS_{equil,t}$) are calculated in the same manner as baseline SOC levels ($BS_{equil,t}$) described above.

For project activities where SOC levels are expected to transition from one equilibrium level to another due to a change in land use type or management practice, these transitions are accounted for using the following equation:

$$PS_t = \frac{1}{D} \sum_{t-D+1}^{t} PS_{equil,t} \cdot \Delta t$$

Where:

PS_t	Estimate of the project SOC in year t, tC
$PS_{equil,t}$	Estimate of the project SOC in equilibrium year t, tC
D	The transition period required for SOC to be at equilibrium after a change in land use or management practice, year
Δt	Time increment = 1 year

This equation assumes a linear transition from one equilibrium value to another over the time period D. The project receives credit each year for incremental increases (also referred to as "removals") in the value of PS_t over time.

11.2.1.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

This methodology assumes the only potential source of leakage is an increase in the use of fuel wood and/or fossil fuels from nonrenewable sources for cooking and heating purposes due to the decrease in the use of manure and/or residuals as an energy source. Although applicability conditions c) and d) described above may suggest the cultivation of new cropland could be unrelated to project activities, this methodology provides no apparent justification for excluding the possibility that project activities which transition land out of cropland to grassland (e.g., CCG project type as discussed in the Issue Paper)

[28] Available online at http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. The approach to soil carbon change estimation is in Section 2.3.3 of Volume 4: Agriculture, Forestry and Other Land Use.

could result in the cultivation of new land in response to the reduced availability of cropland acres due to project activities.

11.2.1.7 How is permanence of soil carbon addressed?

All Agriculture, Forestry and Other Land Use (AFOLU) projects developed under VCS methodologies utilize the most recent version of the VCS AFOLU Non-Performance Risk Tool (described separately below).

11.2.2 Continuous Conservation Tillage and Conversion to Grassland Soil Carbon Sequestration Offset Project Protocol (CCX)

11.2.2.1 What are the project activities eligible for crediting?

This protocol applies to two distinct project types: implementation of conservation tillage and cropland to grassland conversion. Grassland conversion projects are comparable to cropland retirement.

Grassland conversion projects are defined as:

> *For CCX purposes Grassland conversion is defined as the act of converting land previously used for crop production to grassland cover for the purpose of capturing (sequestering) atmospheric carbon through photosynthesis during growth, and allowing the vegetation to remain on and in the soil, wherein the vegetative matter will decompose to stable organic carbon.*

11.2.2.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

For grassland conversion projects, 47 states are listed as eligible, with 38 offered blanket eligibility approval and 9 additional states given county-by-county eligibility listings. Alaska, Hawaii, and Mississippi are not listed among eligible states for grassland conversion projects. It is unclear whether the absence of Mississippi from eligibility is intentional, as the state is included within the eligibility list for conservation tillage projects.

11.2.2.3 What are the methods for determining additionality?

The methodology requires regulatory surplus and provides justification that the eligible project activities are not common practice, thus practically offering a blanket determination of additionality to all eligible project activities.

11.2.2.4 How are baseline soil organic carbon levels determined?

CCX does not require projects to determine baseline SOC levels. Rather, credits are awarded based on standardized rates depending on the location of the project area.

11.2.2.5 How is soil organic carbon sequestration measured and quantified?

CCX applies standardized rates to these projects, greatly reducing the measurement, reporting, and verification costs for participation in the offset program. The offset issuance rates for grassland conversion projects are either 0.4 tCO_2e/ac/yr or 1.0 tCO_2e/ac/yr. CCX offers the following explanation for the derivation of these rates:

Offset issuance rates were established with the help of leading U.S. soil science experts who provided published sequestration rates and expert opinion. Rates were established by taking the average of the sequestration rates published in peer reviewed academic literature for specific regions. For conservativeness these rates were then discounted by twenty% from the average published rate for the region.

11.2.2.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

CCX maintains that no leakage counting is necessary for these project types:

CCX does not expect continuous conservation tillage or grassland conversion projects to result in new or changed activities that increase GHG emissions outside of the Project Boundary and, therefore, no project specific leakage assessment is required.

11.2.2.7 How is permanence of soil carbon addressed?

CCX manages a Soil Carbon Reserve Pool that holds 20% of a project's credits as a buffer that can be canceled in the event the project is found in non-compliance with the protocol. CCX also implemented a 20% discount to the standardized offset issuance rates to respond to risks of reversal following the completion of the five-year offset project term. No credits are technically held in Reserve and no monitoring of project areas following the five-year term is required; instead, CCX states this discount should be sufficient to account for reversal risks after a project is completed. CCX states:

In working with outside experts to establish crediting rates for eligible practices, CCX established a "permanence reserve" through the use of discounted crediting. Outside experts provided CCX with estimated average sequestration rates applied to specific geographic areas as found in Appendix B. The average rates were then discounted by 10-20% in order to account for the potential loss of carbon should project participants reverse practices by returning to conventional tillage. These discounted rates create a pool-wide permanence reserve of actual offsets that have occurred but have never been issued to the Project Owner. Provided that any reversal of carbon stored caused by producers who discontinue offset practices after their contracts expire is, in aggregate, less than the offsets in the permanence reserve, then, in practice, the reductions are considered permanent.

11.2.3 Sustainably Managed Rangeland Soil Carbon Sequestration Offset Project Protocol (CCX)

11.2.3.1 What are the project activities eligible for crediting?

The Chicago Climate Exchange's rangeland protocol gives a Project Definition of:

A Project consists of the adoption of a forward looking, documented plan which includes a minimum five year commitment to manage Rangeland for increased soil carbon storage through practices that identify and accommodate periods of grazing, ensure sustainable forage-animal balance such that forage produced meets demand of livestock *and/or wildlife and provides for a contingency plan for management under drought conditions.*

Projects must also adopt a formal grazing plan in order to be eligible.

11.2.3.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The CCX limits projects geographically to those occurring within approved counties. Selected counties were chosen based on the USDA Land Resource Regions (LRRs). The LRRs with eligible counties are:

- B: Northwestern Wheat and Range Region
- C: California Subtropical Fruit, Truck, and Specialty Crop Region
- E: Rocky Mountain Range and Forest Region
- F: Northern Great Plains Spring Wheat Region
- G: Western Great Plains Range and Irrigated Region
- H: Central Great Plains Winter Wheat and Range Region
- I: Southwest Plateaus and Plains Range and Cotton Region
- J: Southwestern Prairies Cotton and Forage Region

Furthermore, this methodology requires the project to occur on land classified as rangeland, utilizing the NRCS definition of rangeland:

> *"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, Rangeland supports native vegetation that is extensively managed through the control of livestock rather than by agronomic practices, such as fertilization, mowing, or irrigation. Rangeland also includes areas that have been seeded to introduced species (e.g., crested wheatgrass) but are managed with the same methods as native Rangeland."*

11.2.3.3 What are the methods for determining additionality?

The methodology requires regulatory surplus and provides justification that the eligible project activities are not common practice, thus practically offering a blanket determination of additionality to all eligible project activities.

11.2.3.4 How are baseline soil organic carbon levels determined?

The methodology does not require or provide guidance for determining absolute SOC levels in either the baseline or project scenarios. Baseline SOC levels are assumed to be steady or decreasing, and assumed SOC gains associated with project activities are awarded with pre-defined rates.

11.2.3.5 How is soil organic carbon sequestration measured and quantified?

Default SOC sequestration rates based on expert committee recommendations are provided for each LRR. The absolute values of SOC levels in the project and baseline scenarios are not considered, and therefore field measurement or other quantitative methods are not necessary.

11.2.3.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

All projects must provide a declaration that stocking rates on Rangeland owned and controlled outside the eligible LRRs have not increased as a result of the enrollment of the Project.

Participating rangeland owners are required to enroll all of their properties within the relevant LRR in the project.

11.2.3.7 How is permanence of soil carbon addressed?

Project participants must commit to at least 5 years of project activities and contribute 20% of issued offset credits into a collective buffer pool. Field verification is required on a 10% sample of project acres each year over the five-year project life.

11.2.4 Combined tool to identify the baseline scenario and demonstrate additionality in A/R CDM project activities (Version 01)

11.2.4.1 Context

This tool complements CDM A/R methodologies to provide a project-specific additionality and baseline evaluation.

11.2.4.2 What are the project activities eligible for crediting?

N/A

11.2.4.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

N/A

11.2.4.4 What are the methods for determining additionality?

This tool guides project developers through the preparation of alternative baseline scenarios and the subsequent consideration of regulatory surplus and implementation barriers for both the alternative baseline scenarios and the project scenario. Following the definition of alternative baseline scenarios and the application of regulatory surplus and implementation barriers steps, projects with more than one remaining alternative baseline scenario may select the most conservative scenario in terms of lowest baseline emissions or must justify a less conservative emissions scenario through a financial analysis of the net present value for the alternative scenarios. Finally, the project scenario must pass a common practice test indicating the project activities are not common practice in the relevant sector and applicable geographic area.

11.2.4.5 How are baseline soil organic carbon levels determined?

N/A

11.2.4.6 How is soil organic carbon sequestration measured and quantified?

N/A

11.2.4.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

N/A

11.2.4.8 How is permanence of soil carbon addressed?

Offset credits issued to CDM A/R projects are temporary and must be renewed over time.

11.2.5 Estimation of the increase in GHG emissions attributable to displacement of pre-project agricultural activities in A/R CDM project activity (Version 01)

11.2.5.1 Context

This tool superseded the "Tool for estimation of GHG emissions related to displacement of grazing activities in A/R CDM project activity" on June 3, 2011. It takes a similar approach, but has significantly reduced the assumptions and qualifications present in the preceding tool. This tool is the most recent (and currently active) version for calculating the leakage of CDM A/R project emissions reductions due to displacement of pre-project grazing to nearby areas.

11.2.5.2 What are the project activities eligible for crediting?

N/A

11.2.5.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The only condition limiting the application of this tool to CDM A/R projects is that it cannot be applied to projects where the displacement of agricultural activities results in any drainage of wetlands or peatlands.

11.2.5.4 What are the methods for determining additionality?

N/A

11.2.5.5 How are baseline soil organic carbon levels determined?

N/A

11.2.5.6 How is soil organic carbon sequestration measured and quantified?

N/A

11.2.5.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

This tool simplifies the calculation of leakage due to displaced agricultural activities compared to the earlier grazing displacement tool it superseded. It also broadened the scope for leakage activities beyond grazing alone and now address displacement of any agricultural activities that have been displaced and meet the threshold for significance in terms of project emissions reductions.

The first step in calculating leakage for this tool is quantifying the annual change in carbon stocks on the portion of the project area which will have displaced activities, and then determining a discount factor that is fundamentally based upon the amount of forest cover in the country. The calculation of leakage is:

$$LK_{Agric,t^*} = \frac{44}{12} * \frac{f}{T_{cred}} * \Delta Cd_{t^*}$$

Where:

LK_{Agric,t^*}	Leakage due to displacement of agricultural activities in year t*; t CO_2-e
f	Fraction of land covered by forest (according to the national definition of forest) in the region containing the A/R CDM project activity; dimensionless
T_{cred}	Number of years contained in the first crediting period; dimensionless
$* \Delta Cd_{t^*}$	Sum of annual changes in carbon stock in all selected carbon pools since the start of the A/R CDM project activity to the year of verification t_{ver} attributable to the area subject to pre-project agricultural activities that are displaced during year *t* since the start of the A/R project activity; t C
t	1, 2, 3, … t* years elapsed since the start of the A/R CDM project activity
$\frac{44}{12}$	Ratio of molecular weight of CO_2 to carbon; t CO_2-e/tC

The primary simplifying move made in this tool compared to the last one is the estimation of the discount factor based fundamentally on the proportion of forest cover in the country where the project occurs. This leakage discount no longer relies upon estimation of the grazing characteristics of displaced livestock or the identification of the specific lands to which the activities are displaced. Countries with higher proportion of forest cover produce higher discount factors. With crediting periods set at 20 or 30 years, the discount factor would be a maximum of 5% or 3%, respectively, assuming 100% forest cover.

11.2.5.8 How is permanence of soil carbon addressed?

N/A

11.2.6 Tool for estimation of GHG emissions related to displacement of grazing activities in A/R CDM project activity (Version 02)

11.2.6.1 Context

This tool is no longer active, having been superseded by the tool "Estimation of the increase in GHG emissions attributable to displacement of pre-project agricultural activities in A/R CDM project activity" on June 3, 2011. This tool includes several substantially different assumptions from its successor and the assumptions of this tool have been cited within other methodologies to calculate leakage.

11.2.6.2 What are the project activities eligible for crediting?

N/A

11.2.6.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

This tool is generally applicable to CDM A/R projects, but can only be utilized when the displacement of grazing occurs with the following circumstances:

1. *If the grazing animals are already in a zero-grazing system or are moved to a zero-grazing system then the grazing activity that is monitored is the production of fodder.*

2. *The tool can be used to estimate the emissions caused by displacement to:*
    - *Identified Forest land;*
    - *Identified Cropland;*
    - *Identified Grassland; and*
    - *Unidentified land.*
3. *The tool is not applicable for estimating GHG emissions due to implementation of an A/R CDM project activity that causes displacement to:*
    - *Settlements;*
    - *Wetlands; and*
    - *Other lands – as defined by the GPG LULUCF (i.e., bare soil, rock, ice, and all unmanaged land areas that do not fall into category of forest land, cropland, grassland, settlements or wetlands.*

11.2.6.4 What are the methods for determining additionality?
N/A

11.2.6.5 How are baseline soil organic carbon levels determined?
N/A

11.2.6.6 How is soil organic carbon sequestration measured and quantified?
N/A

11.2.6.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?
This tool quantifies the emissions due to displaced grazing activities based on the fodder requirements for the displaced livestock, including consideration of the number, and dry matter demand for grazing animals. A critical assumption of the tool is that the sale of livestock that previously grazed the project area to another entity not involved in the project (whether for grazing or slaughter) does not produce leakage. Thus, the tool is applicable where the pre-project livestock are displaced outside the project area, but are still owned or controlled by the project proponent, meaning this tool is technically addressing direct rather than indirect land use change.

The tool accounts for emissions due to displacement of grazing animals including the consumption of biomass on the lands (e.g., perennial crops, grasses, etc.). These calculations can be made primarily using lookup values in most circumstances. In addition, N_2O emissions due to increases in fertilizer use in the area where livestock are displaced are estimated using actual invoices or records of fertilizer use on these lands outside the project area.

11.2.6.8 How is permanence of soil carbon addressed?
N/A

11.2.7 Tool for estimation of change in soil organic carbon stocks due to the implementation of A/R CDM project activities (Version 01.1.0)

11.2.7.1 Context

As a methodological tool for CDM A/R projects, this document is not a stand-alone carbon project methodology, but nevertheless offers an example of certain policies that could be considered for development of accounting procedures for SOC in AGC and CCG project types discussed in the Issue Paper.

11.2.7.2 What are the project activities eligible for crediting?

N/A

11.2.7.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The tool may not be applied to organic soils or wetlands.

Soil disturbance as part of the project activity is only permitted in the case of site preparation activities before planting, cannot occur more than once in a 20-year period, and must be in accordance with appropriate soil conservation practices.

The tool also contains a table/matrix containing combinations of temperature/moisture regimes, cropland tillage intensity and nutrient inputs that are not eligible. In addition, a separate table for temperature/moisture regimes, grassland degradation status (degraded, improving, etc.), and nutrient inputs also describes management scenarios under which the tool would not be applicable.

11.2.7.4 What are the methods for determining additionality?

N/A

11.2.7.5 How are baseline soil organic carbon levels determined?

This tool generally follows the approach described in the 2006 IPCC Guidelines for National GHG Inventories[29]. SOC levels are estimated through the use of lookup values categorized by climate region and soil type and several stock change factors. Initial SOC levels are estimated as:

$$SOC_{INITIAL,i} = SOC_{REF,i} * f_{LU,i} * f_{MG,i} * f_{IN,i}$$

Where:

$SOC_{INITIAL,i}$	SOC stock at the beginning of the A/R CDM project activity in stratum *i* of the areas of land; t C ha^{-1}
$SOC_{REF,i}$	Reference SOC stock corresponding to the reference condition in native lands (i.e., non-degraded, unimproved lands under native vegetation - normally forest) by climate region and soil type applicable to stratum *i* of the areas of land; t C ha^{-1}

[29] Available online at http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. The approach to soil carbon change estimation is in Section 2.3.3 of Volume 4: Agriculture, Forestry and Other Land Use.

$f_{LU,i}$	Relative stock change factor for baseline land-use in stratum *i* of the areas of land; dimensionless
$f_{MG,i}$	Relative stock change factor for baseline management regime in stratum *i* of the areas of land; dimensionless
$f_{IN,i}$	Relative stock change factor for baseline input regime (e.g., crop residue returns, manure) in stratum *i* of the areas of land; dimensionless
i	1, 2, 3, … strata of areas of land; dimensionless

The tool provides default values for $SOC_{REF,i}$ and all three relative stock change factors.

SOC stocks in the baseline are assumed to be steady-state or decreasing (and conservatively treated as if they were at steady state) and are thus not based upon successive measurement.

11.2.7.6 How is soil organic carbon sequestration measured and quantified?

The change in SOC levels due to project activities is based upon reference values provided in lookup tables and is assumed to proceed linearly from one steady-state reference level to another over the course of 20 years following the change in land use or management practice.

Soil disturbance due to site preparation activities occurring on more than 10% of the area of a stratum is accounted as $SOC_{LOSS,i}$ for with an assumed 10% loss of initial carbon stocks ($SOC_{INITIAL,i}$) in the year of site preparation.

The change in SOC levels following site preparation is estimated as:

$$dSOC_{t,i} = \frac{SOC_{REF,i} - \left(SOC_{INITIAL,i} - SOC_{LOSS,i}\right)}{20\ years} \quad \text{for } t_{PREP,i} < t \leq t_{PREP,i} + 20$$

Where:

$dSOC_{t,i}$	The rate of change in SOC stock in stratum *i* of the areas of land, in year *t*; t C ha^{-1} yr^{-1}
$t_{PREP,i}$	The year in which first soil disturbance takes place in stratum *i* of the areas of land
$SOC_{LOSS,i}$	Loss of SOC caused by soil disturbance attributable the A/R CDM project activity, in stratum *i* of the areas of land; t C ha^{-1}
$SOC_{REF,i}$	Reference SOC stock corresponding to the reference condition in native lands (i.e., non-degraded, unimproved lands under native vegetation - normally forest) by climate region and soil type applicable to stratum *i* of the areas of land; t C ha^{-1}
$SOC_{INITIAL,i}$	SOC stock at the beginning of the A/R CDM project activity in stratum *i* of the areas of land; t C ha^{-1}
i	1, 2, 3, … strata of areas of land; dimensionless
t	1, 2, 3, … years elapsed since the start of the A/R CDM project activity

11.2.7.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

N/A

11.2.7.8 How is permanence of soil carbon addressed?

All offset credits issued to CDM A/R projects are temporary and must be renewed over time.

11.2.8 Tool for the Demonstration and Assessment of Additionality in VCS Agriculture, Forestry and Other Land Use (AFOLU) Project Activities, v3.0

11.2.8.1 Context

This tool complements offset project methodologies and may be applied for project-specific additionality determinations for AFOLU projects.

11.2.8.2 What are the project activities eligible for crediting?

This tool is generally applicable to AFOLU activities with an applicable VCS-approved methodology.

11.2.8.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

This tool is generally applicable to AFOLU activities with an applicable VCS-approved methodology. Two additional applicability criteria for this tool are:

1. *AFOLU activities the same or similar to the proposed project activity on the land within the proposed project boundary performed with or without being registered as the VCS AFOLU project shall not lead to violation of any applicable law even if the law is not enforced;*
2. *The use of this tool to determine additionality requires the baseline methodology to provide for a stepwise approach justifying the determination of the most plausible baseline scenario. Project proponent(s) proposing new baseline methodologies shall ensure consistency between the determination of a baseline scenario and the determination of additionality of a project activity.*

11.2.8.4 What are the methods for determining additionality?

The tool goes through four primary steps, which function in a manner similar to the commonly-invoked CDM "Combined tool to identify the baseline scenario and demonstrate additionality in A/R CDM project activities". The four steps applied in this tool are:

STEP 1. Identification of alternative land use scenarios to the AFOLU project activity;
STEP 2. Investment analysis to determine that the proposed project activity is not the most economically or financially attractive of the identified land use scenarios; or
STEP 3. Barriers analysis; and
STEP 4. Common practice analysis.

In Step 1, projects create a list of alternative scenario that are consistent with applicable laws and regulations, including consideration of systematic non-enforcement where appropriate. The methodology being applied to account for the project's emissions reductions should provide a step-wise process to select the baseline scenario from among the scenarios identified.

Projects then have the option of applying either Step 2 (financial analysis) OR Step 3 (barriers analysis). The financial analysis includes three test options, with option 1 available if a project does not expect financial benefits from the baseline activity other than carbon revenues:

OPTION I. Simple cost analysis

If a project demonstrates it will produce no financial benefits other than VCS-related income, it is deemed additional.

OPTION II. Investment comparison analysis

Using an economic indicator chosen by the project (e.g., internal rate of return, net present value, payback period, etc.), project activities without VCS-related income that are not the most financially attractive option compared to other potential baseline scenarios identified in Step 1 of this tool are deemed additional.

OPTION III. Benchmark analysis

Using an economic indicator chosen by the project (e.g., internal rate of return, net present value, payback period, etc.), project activities without VCS-related income must produce a value for this indicator that is below a benchmark value which represent common financial indicators that would be used in the sector for making the business decisions such as investing in the project or baseline activities. Project activities that produce an indicator value that is less favorable than the benchmark are deemed additional.

Note: Options II and III are also subject to a sensitivity analysis that reviews the outcome of the investment comparison and benchmark analyses if input factors or criteria are varied.

For projects applying the barriers analysis (Step 3), the proponent must demonstrate the existence of barriers that would prevent the implementation of project activities in the absence of the carbon project, and that these barriers would not prevent the implementation of at least one of the alternative land use scenarios identified in Step 1.

All projects must complete Step 4 to demonstrate that project activities are not common practice in the geographical area where the project is implemented. This includes consideration of similar (in terms of scale, regulatory context, and environmental conditions) activities that have been undertaken within 10 years of the project's start date that have not been developed as carbon projects.

11.2.8.5 How are baseline soil organic carbon levels determined?

N/A

11.2.8.6 How is soil organic carbon sequestration measured and quantified?

N/A

11.2.8.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

N/A

11.2.8.8 How is permanence of soil carbon addressed?

N/A

11.2.9 AFOLU Non-Performance Risk Tool (VCS)

11.2.9.1 Context

All projects applying VCS methodologies within the Agriculture, Forestry, and Other Land Use (AFOLU) sectors must utilize the Non-Performance Risk Tool to determine the anticipated risk of reversals and corresponding risk mitigation (buffer) requirement. This tool is not a standalone methodology, but rather complements the individual methodologies developed for use under VCS.

11.2.9.2 What are the project activities eligible for crediting?

All AFOLU project activities under approved VCS methodologies.

11.2.9.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

N/A

11.2.9.4 What are the methods for determining additionality?

N/A

11.2.9.5 How are baseline soil organic carbon levels determined?

N/A

11.2.9.6 How is soil organic carbon sequestration measured and quantified?

N/A

11.2.9.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

N/A

11.2.9.8 How is permanence of soil carbon addressed?

AFOLU projects developed under a VCS methodology define risks related to project management, financial viability, opportunity cost, project longevity, land ownership and access, community engagement, political risk, and natural disturbances using a common scoring system.

The resulting risk score is used to determine the proportion of a project's issued credits that must be held in a buffer pool for potential reversals. Projects with a risk rating >60% are prohibited from being issued any credits.

11.3 Review of Proposed Methodologies, Modules, and Tools

Methodologies, modules, and tools reviewed in this section were under review and pending approval by a carbon offset standard at the time this Issue Paper was prepared. They include:

- Methodology for Avoided Conversion of Grasslands and Shrublands from Planned Conversion (VCS & ACR) – pg. 132
- Adoption of Sustainable Grassland Management through Adjustment of Fire and Grazing – Version 1 (VCS) – pg. 136
- Agricultural Land Management: Improved Grassland Management (Version 2.4) (VCS) – pg. 139
- Methodology for Sustainable Grassland Management (SGM) (VCS) – pg. 141
- Proposed Module/Tool – Soil Carbon (VCS) – pg. 145

11.3.1 Methodology for Avoided Conversion of Grasslands and Shrublands from Planned Conversion (VCS & ACR)

11.3.1.1 Context

Two methodologies have originated from the Ducks Unlimited Avoided Grassland Conversion Project in the Prairie Pothole Region. Separate methodologies have been submitted for approval and posted for public comment by VCS (Dell, et al., 2012b) and ACR (Dell, et al., 2012a). There are several small differences between the two methodologies, but the accounting approaches considered in the following questions are generally equivalent across these two methodologies. The following discussion should thus be considered applicable to both the VCS and ACR public comment versions of these methodologies.

11.3.1.2 What are the project activities eligible for crediting?

These methodologies were designed for use by projects intending to avoid the planned conversion of grasslands or shrublands to cropland. A conservation agreement, such as a conservation easement, is required for the duration of the project.

11.3.1.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The applicability conditions for both methodologies are equivalent. Projects are limited geographically to the USA and Canada. Organic soils are excluded. Management practices in the with-project scenario that would lead to a sustained decrease in vegetative cover are prohibited. Livestock may only be present in projects where the animals are primarily forage fed and where their manure is not managed or dispersed in liquid form.

The applicability criteria in the ACR version are:

- All Participant Fields in the Project Area are currently grassland or shrubland, have qualified as grassland or shrubland for at least 10 years prior to the Start Date, will remain as grassland or shrubland throughout the Project Term, and are legally able to be converted and would be converted to cropland in the absence of the project activity.

- All Participant Fields enrolled in the Project Area must be subject to a Land Conservation Agreement entered into by the Project Participant prohibiting the conversion of the land from grassland or shrubland for the duration of the Project Term.
- All Participant Fields must have the 'highest and best use' identified as cropland through an independent appraisal, and the appraised value of each Field as cropland must be at least 40% greater than its value as grassland or shrubland in the project scenario.
- Land may remain in use for animal husbandry and be subject to prescribed burning or wildfires during the project scenario, so long as prescribed burning conforms to current best management practices in the Project Region and does not knowingly contribute to the succession of native grasslands or shrublands to an alternative vegetation type.
- This methodology is only applicable to projects avoiding the complete conversion of grasslands or shrublands to cropland and not the degradation of grasslands or shrublands.
- Project Proponents can demonstrate control over the Participant Fields and Project Area, and own rights to the greenhouse gas benefits of the project activity for the length of the Project Term.
- The Project Area can include either one continuous parcel, or multiple discrete parcels of land. If the Project Area consists of multiple discrete parcels, Project Proponents must demonstrate that each discrete parcel meets all applicability criteria of the methodology.
- Project Areas shall not include grasslands on organic soils or peatlands, or grasslands on non-forest wetlands.
- Where livestock are present in the project scenario, manure may not be managed, stored, or dispersed in liquid form. Livestock shall be primarily forage fed and not managed in a confined area, e.g., feedlot.
- In the project scenario, overgrazing, overstocking, or overuse of prescribed fires leading to the progressive loss of vegetative cover shall not occur, allowing carbon pools to remain at a steady state. Supplemental management practices that increase carbon stocks are allowable but the resultant emissions avoided or removed are not eligible for crediting unless quantified through a separate methodology.
- The Project Area is located in the United States or Canada.

11.3.1.4 What are the methods for determining additionality?

Each methodology requires satisfaction of additionality criteria defined by VCS and ACR. For VCS, this refers to the VCS Tool for the Demonstration and Assessment of Additionality in VCS AFOLU Project Activities, described in Section 11.2.8 above. For ACR, this refers to ACR's Three Prong Additionality Test, which includes consideration of Regulatory Surplus, Common Practice, and Implementation Barriers.

The methodology authors have proposed the use of a parcel-specific appraisal, market study report, or general narrative (collectively termed appraisal) prepared by a certified general appraiser to satisfy additionality requirements where cropland is identified as the highest and best land-use for the Project Area and the value of the Project Area as cropland is 40% or more greater than the value of the Project Area as grassland.

The ACR methodology provides further methods for demonstrating common practice:

Step 1 - Entity Acquiring Land Conservation Agreement

Is the Land Conservation Agreement held and purchased by a land trust, government agency, or other entity that holds similar Land Conservation Agreements in the Project Region? If no, project activity satisfies common practice analysis, otherwise proceed to Step 2.

Step 2 - Historic Availability of Easements in Project Region

If the answer to any of the following questions is yes, the project activity shall be deemed additional. If none of the below conditions apply, the project activity shall not be considered additional. Project Proponents shall provide sufficient evidence in the GHG Project Plan to prove additionality based on at least one of these criteria, and also to demonstrate the role of carbon finance in differentiating project activities.

- *Are the project's Land Conservation Agreements the first on grassland or shrubland in the Project Region?*
- *If easement programs or other programs implementing land use restrictions such as those in a Land Conservation Agreement have been in existence in the Project Region, has there been a decrease in funding available from historical funding sources for Agreements over the past five years?*
- *If easement or other Agreement programs have been in existence, regardless of funding status, has there been an essential distinction in the competitiveness of Agreement offers prior to the project activity due to funding sources or administrative restrictions that have hindered Agreements from remaining competitive with incentives for conversation to cropland?*
- *Are Agreements implemented on parcels that are at elevated risk of conversion relative to other Agreements (existing and candidate), which may have been targeted for objectives other than risk of conversion, e.g., biodiversity conservation?*
- *Does carbon finance provide funding for 100% of the Agreement, e.g. no additional financial sources are used to implement project activities?*

11.3.1.5 How are baseline soil organic carbon levels determined?

Both methodologies require stratification of the Project Area, and determination of initial SOC stocks is permitted through direct measurement or through the use of regional soil carbon inventories. Following initial SOC determination, baseline SOC levels for avoided grassland conversion must be estimated, as they cannot be directly measured.

Both methodologies quantify baseline SOC stocks using an emissions factor, which by default uses the IPCC approach to quantify SOC changes over a 20-year period between equilibrium SOC values. For example, the VCS version prescribes SOC baseline determination as follows:

…total soil organic carbon stocks in the baseline scenario for each Participant Field in the Project Area shall be calculated as:

$$C_{SOC,BL_{p,y}} = \sum_{i}^{I} \sum_{t=0}^{t \leq 20} C_{SOC_{i,y=0}} * EF_{t,i,y} * F_{i,y} * FC_{t,y}$$

Where:

$C_{SOC,BL_{p,y}}$	*Carbon stock of soil organic carbon for Participant Field p in the baseline scenario in year y; tCO$_2$e*
$C_{SOC_{i,y=0}}$	*Total initial (year y=0) soil organic carbon stock for stratum i, fixed for project duration; tCO$_2$e*
$EF_{t,i,y}$	*Emission factor for stratum i in year y, the fraction of soil organic carbon pool remaining t years since conversion to cropland*
$F_{i,y}$	*The proportion of Participant Field p included in stratum i in year y; hectares Participant Field p (hectares stratum i)$^{-1}$*
$FC_{t,y}$	*Proportion of Participant Field p that has been converted to cropland in the baseline scenario for t years as of year y; dimensionless*
I	*Total number of strata*
t	*Time since conversion of grassland to cropland in the baseline scenario, maximum value of 20; years*

By default, this method assumes the emissions from soil organic carbon following conversion proceed linearly for 20 years (i.e., D_i = 20), at which point a new equilibrium level of SOC is reached in the converted state. A linear EF function may be used per the IPCC AFOLU Guidelines 2006 (adapted from Eq. 2.25, Ch2, p2.30), in which case:

$$EF_{t,i,y} = \frac{1 - (FSOC_{LU_i} * FSOC_{MG_i} * FSOC_{IN_i})}{D_i} * t$$

Where:

$EF_{t,i,y}$	*Emission factor describing the fraction of soil organic carbon pool remaining t years since conversion to cropland for stratum i in year y; dimensionless*
$FSOC_{LU_i}$	*Fraction of soil organic carbon pool remaining after transition period, accounting for land use factors in stratum i; dimensionless*
$FSOC_{MG_i}$	*Fraction of soil organic carbon pool remaining after transition period, accounting for management factors for stratum i; dimensionless*
$FSOC_{IN_i}$	*Fraction of soil organic carbon pool remaining after transition period, accounting for input of organic matter factors for stratum i; dimensionless*
D_i	*Transition period for soil organic carbon for stratum i, time period for transition between equilibrium SOC values, default value of 20; years*

t *Time since conversion of grassland to cropland in the baseline scenario, maximum value of 20; years*

Alternatively, a non-linear function may be used to calculate $EF_{t,i,y}$ values for each soil organic carbon stratum if the function and derived values are:

- *Derived from a peer-reviewed study of soils and a region similar to the Project Area or Project Region, or*
- *An output from a biogeochemical model, e.g., DNDC, DAYCENT, or others, that requires input data for management practices, climatology, and/or other factors determined significant to the rate of soil carbon oxidation and resulting emission factor, or*
- *An empirical result from a pair-wise field measurement at a site materially similar to the Project Area, and soil samples are collected from the relevant soil layers that would be affected by the conversion process and baseline activity. A sample-based emission factor shall not be projected for a period of time longer than the collection period, and at a minimum shall be measured following the same management treatments for duration of 5 years. Use of pair-wise samples from similar lands shall be adjusted for uncertainty as described on page 22 of the VCS Standard version 3.1 or the equivalent section of the latest version of the VCS Standard, or section 5.2.35 of IPCC GL AFOLU 2006*.

11.3.1.6 How is soil organic carbon sequestration measured and quantified?

Both methodologies require stratification of the Project Area, and determination of initial SOC stocks is permitted through direct measurement or through the use of regional soil carbon inventories. Following initial SOC determination, with-project SOC levels are assumed to remain unchanged.

11.3.1.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

A default discount rate of 20% is applied. This rate was based on consideration of reported elasticities of supply and demand and the logic for accounting for market effects leakage as described in Section 12.2.1.1.1 of this Issue Paper.

11.3.1.8 How is permanence of soil carbon addressed?

Potential reversals for both methodologies are currently proposed to be addressed using the latest AFOLU Non-Performance Risk Tool approved by VCS (see above).

11.3.2 Adoption of Sustainable Grassland Management through Adjustment of Fire and Grazing – Version 1 (VCS)

11.3.2.1 What are the project activities eligible for crediting?

This methodology addresses projects that implement changes in grazing density or the frequency of prescribed fire on unfertilized and uncultivated grasslands.

11.3.2.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

There are no restrictions based on geographic location or soil types. In terms of cropping systems, the methodology scope is limited to uncultivated grasslands where there is no net import of manure or fertilizer. The full applicability conditions described in the methodology are:

a) *Land is uncultivated, unfertilized (no net import in manure or fertilizer) grassland at the start of the project;*
b) *There is constant or increasing agricultural pressure on lands in absence of the project;*
c) *The land will be used primarily for animal husbandry, wildlife conservation or both;*
d) *Land is potentially subject to prescribed burning or wildfires;*
e) *Forest land in the area is constant or decreasing over time;*
f) *Existing woody perennials are not removed during the first two years of project implementation and no net loss of woody perennials occurs during the project from either human removal, fire or animal-caused herbivory or mortality;*
g) *There will be no net displacement of livestock to areas outside that of the project;*
h) *There is no significant displacement of manure from outside the project boundary to within the project boundary or addition of fertilizer of any other kind inside the project boundary;*
i) *Anticipated increases in the use of fossil fuels for grazing or fire management (i.e., use of vehicles to herd livestock or control fire) or for ecotourism (vehicles, lodges, camps, etc.), or for cooking and heating will be considered as leakage. Such leakage will be monitored during the project and either deducted from calculated offsets or offset within the project*

11.3.2.3 What are the methods for determining additionality?

This methodology requires projects to apply the “Combined tool to identify the baseline scenario and demonstrate the additionality in A/R CDM project activities” (see discussion of this tool above).

11.3.2.4 How are baseline soil organic carbon levels determined?

This methodology is based on the approach described in the 2006 IPCC Guidelines for National GHG Inventories[30].

To calculated initial SOC levels, field measurement is required. The methodology recommends the use of the CDM A/R tool “Calculation of the number of sample plots for measurements within A/R CDM project activities” to guide stratification of the project area and sampling design. The project area is expected to be stratified according to climate and other edaphic properties affecting SOC dynamics. Soil samples are to be collected to the shallowest option between a depth of 40cm, the depth of a hardpan (soil layer), or the depth of bedrock.

Any transitions from the initial land use or management system in the baseline scenario are then assumed to shift SOC levels from one equilibrium level to another over a defined transition time (default

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[30] Available online at http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html. The approach to soil carbon change estimation is in Section 2.3.3 of Volume 4: Agriculture, Forestry and Other Land Use.

is 20 years). The resulting equilibrium SOC level in the new land use or management system must be estimated using a soil carbon model such as Roth-C or CENTURY.

11.3.2.5 How is soil organic carbon sequestration measured and quantified?

Initial carbon stocks are equal in both the baseline and project scenario and are quantified based on field sampling (described briefly above).

In the project scenario, project proponents must make predictive estimates of SOC levels using soil carbon models and their modeled equilibria for each stratum according to the land use and management practices simulated in the model. These estimates are then reconciled with field samples collected every 3-10 years.

Field measurements of changes in SOC levels are compared to model-estimated equilibrium values for each stratum. Field measurements are used to record the change in SOC density as follows:

$$MPRS_t = \sum_m \sum_{i=1}^{n} MSOC_{im,t} - BSOC_{im}$$

Where:

$MPRS_t$	measured SOC project removals for management area *m* in year *t*
$MSOC_{im,t}$	measured SOC density at station *i* in management area *m* in year *t*
$BSOC_{im}$	baseline initial SOC density at station *i*
n	total field samples of SOC density
t	1, 2, 3, … years since project initiation

These estimates for SOC density based on field sampling are then compared to modeled estimates that are calculated as:

$$PSQ_{,t} = \sum_m \left(PSOC_m^{equil} - BSOC_m\right) \cdot \frac{t}{D}$$

Where:

$PSQ_{,t}$	cumulative sequestration predicted by the soil carbon dynamic model by year *t* of the project
$PSOC_m^{equil}$	the model-estimated equilibrium SOC in management area *m*
$BSOC_m$	baseline SOC in management area *m*
D	the model-estimated time to project SOC equilibrium

To adjust modeled estimates based on field measurement, the methodology requires:

> *Project shortfalls result if $MPRS_t < PSQ_{,t}$ and the difference in estimated versus measured offsets ($PSQ_{,t} - MPRS_t$) must be rectified through the project buffer…. Also, model parameters should be adjusted in subsequent years such that the model(s) used predict soil sequestration for the measurement period t.*

11.3.2.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

The only indirect land use change considered by this methodology regards leakage due to the displacement of livestock outside the project area due to project activities. This is precluded by applicability condition g) described above.

11.3.2.7 How is permanence of soil carbon addressed?

Potential reversals are addressed using the latest AFOLU Non-Performance Risk Tool approved by VCS (see above). For this methodology, the buffer pool is also used to address errors introduced through the prediction of SOC levels using soil carbon models that are later compared to field measurements.

11.3.3 Agricultural Land Management: Improved Grassland Management (Version 2.4) (VCS)

11.3.3.1 What are the project activities eligible for crediting?

This methodology relies upon the latest definition of the "improved grassland management" category as defined in the VCS Agriculture, Forestry and Other Land Use Requirements (current version is 3.2):

> *This category includes practices that demonstrably reduce net GHG emissions of grassland ecosystems by increasing soil carbon stocks, reducing N_2O emissions and/or reducing CH_4 emissions, noting the following:*
>
> *a) Soil carbon stocks can be increased by practices that increase belowground inputs or decrease the rate of decomposition. Such practices include increasing forage productivity (e.g., through improved fertility and water management), introducing species with deeper roots and/or more root growth and reducing degradation from overgrazing.*
> *b) Soil N_2O emissions can be reduced by improving nitrogen fertilizer management practices on grasslands as set out in Section 4.2.2(1)(b) above.*
> *c) N_2O and CH_4 emissions associated with burning can be reduced by reducing the frequency and/or intensity of fire.*
> *d) N_2O and CH_4 emissions associated with grazing animals can be reduced through practices such as improving livestock genetics, improving the feed quality (e.g., by introducing new forage species or by feed supplementation) and/or by reducing stocking rates.*

11.3.3.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

This methodology contains no specific limitations based on geographic location or soil types, but does require the project area to be grassland and to remain classified as grassland throughout the crediting period (i.e., land use change is not allowed in baseline or project scenarios).

The specific applicability conditions for this methodology are:

> *a) the boundaries of the project area must be clearly defined;*

b) the baseline scenario must be grassland management;

c) a soil organic carbon model applicable to the project area and that satisfies the specific conditions of this methodology must be available;

d) improved grassland management activities included in the IGMP [Improved Grassland Management Plan] must:

- *decrease the proportion of bare soil in the landscape;*
- *and/or decrease the time bare soil is exposed;*
- *and/or increase the proportion of perennial species above the baseline scenario;*

e) No improved grassland management activities can result in a land designation change;

f) the project area must remain a grass-dominated system throughout the crediting period;

g) improved grassland management activities must not result in an increase in woody perennials that would reach the threshold for the national definition of forest; and

h) no clearing of vegetation shall occur after the project start date except where the grassland management activities that have been proven to enhance long-term grassland productivity are included in the IGMP.

In addition, this project requires the use of a soil carbon model, so supporting information including 10 years of historical management information for the project area is required.

11.3.3.3 What are the methods for determining additionality?

This methodology directs projects to use the latest version of the VCS "Tool for Demonstration and Assessment of Additionality in AFOLU Project Activities" (see above).

11.3.3.4 How are baseline soil organic carbon levels determined?

Baseline SOC levels and changes are estimated using a soil carbon model. The model is used to predict changes in SOC levels due to baseline grassland management practices and the cumulative change over the project's crediting period is then annualized to achieve a linear change in carbon stocks over this the crediting period.

11.3.3.5 How is soil organic carbon sequestration measured and quantified?

The value and change of SOC levels in the project scenario are estimated using a soil carbon model at the project outset and re-run periodically based on any changes that have occurred in environmental conditions, management practices, etc. that would impact SOC dynamics. These values and stock change estimates must be validated with field sampling of SOC.

There are several requirements for field soil sampling in the methodology, including:

- Sampling using a sub-set of grid points used to generate soil carbon model estimates, with sufficient sampling sites to include at least 3 grid points in strata that cover 80% or more of the project area.
- Sampling to a depth of 30cm, unless justification is provided that model or other conditions require a different depth

- A minimum of 30 samples
- 3 soil cores collected within a 1-5m radius that are pooled and subsampled for analysis following procedures described in "ISO 10381-2:2003 Soil quality – sampling – Part 2: Guidance on sampling techniques."
- SOC analysis must include computation of bulk density and SOC estimates must be given as carbon density.

To ground-truth model estimates of SOC levels, project proponents must apply successive statistical tests (a paired comparison test—e.g., a paired t-test) at the project and stratum levels. Instances where model estimates are statistically significantly less conservative than field measurement result in further statistical analysis (i.e., a sign test which reveals whether one source consistently generates higher estimates). Where field measurements are determined to be more conservative at a stratum-level, the model estimates for every grid point in that stratum shall be removed and those values calculated from field sampling shall be used to generate a new spatial layer with SOC levels interpolated based on field-sampled values.

11.3.3.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

Project proponents that own or control grassland outside the project area and within the same country as the project must demonstrate that project activities have not resulted in the shifting of emissions from the project area to these other properties. All other leakage considered is through market effects.

The methodology addresses market leakage specifically regarding the displacement of grazing activity. Where "direct substitution" can be demonstrated (i.e., "where a demand for production is met by an alternative production source with equivalent or lower GHG emissions than livestock production in the *ex-post* project scenario"), the methodology assumes no market leakage is present.

In cases where livestock production decreases by more than 20% from the baseline scenario, project proponents must determine whether these decreases are within the range (within 5% of the variance) commonly observed using historical national grazing data. If the grazing decreases are larger than this variance threshold, 50% of baseline GHG emissions are assumed to be leaked and are added to project emissions.

11.3.3.7 How is permanence of soil carbon addressed?

Potential reversals are addressed using the latest AFOLU Non-Performance Risk Tool approved by VCS (see above).

In addition, soil sampling and reconciliation with model estimates is required every 5 years.

11.3.4 Methodology for Sustainable Grassland Management (SGM) (VCS)

11.3.4.1 What are the project activities eligible for crediting?

According to the methodology authors:

The methodology aims to estimate the reduction of GHG emissions from grassland and increase grassland soil organic carbon (SOC) stock by applying sustainable grassland management practices (SGM). Carbon stock enhancement within the project boundary in above ground and soil carbon pools is considered. This methodology is applicable to projects that introduce SGM into a grassland landscape subject to conditions such that the SOC would remain constant or decrease in the absence of the project.

In the introductory explanation in which the authors addressed three pre-existing methodologies (also contained in this methodology review), they concluded the applicability conditions (e.g., wild- or prescribed fire, increasing perennial species abundance) and use of soil carbon models were unnecessary restrictions limiting the applicability of those methodologies and the authors desired to support "improved grassland management activities not restricted by the above applicability conditions" and not requiring the use of soil carbon models if they are not available or calibrated for local conditions.

11.3.4.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The methodology does not place limits based explicitly on geographic location or soil types, but does have several important applicability restrictions relevant to the consideration of CCG and AGC project types discussed in the Issue Paper. Most notably, this methodology is limited to dry, arid, and semiarid climate types (where potential evaporation is greater than or equal to precipitation) and prohibits consideration of projects where land use types would change. The full applicability conditions included in this methodology are:

a) Land is grassland at the start of the project;
b) Grassland to be sustainably managed is degraded (due to physical constraints as well as anthropogenic actions) and the lands are still degrading[31];
c) There is no displacement of manure from outside the project boundary to within the project boundary;
d) There is no significant increase of use of fossil fuels, fuel wood from non-renewable sources for cooking and heating as a result of the project activity;
e) There is no significant change in manure management systems within the project boundary;
f) The project activity does not include land use change. To clarify, seeding fodder grasses or legumes on degraded grassland is not considered a land-use change activity;
g) If there are studies (for example scientific journals, university theses, or work carried out by the project proponents) that demonstrate that the use of the selected model[32] is valid

[31] The latest version of the "Tool for the identification of degraded or degrading lands for consideration in implementing A/R CDM project activities" shall be applied for demonstrating that lands are degraded or degrading.
[32] The use of the selected model is appropriate for 2006 IPCC AFOLU Guidelines. The model to be applied in the SGM VCS project should be capable of representing the relevant management practices of the project and that the model inputs (i.e., driving variables) are validated from the project region-specific locations that are representatives of the variability of climate, soil and management systems.

for the project region or a similar agroecological zone (AEZ)[33]*, the model can be applied for estimating of carbon stock changes for the SGM VCS project. Otherwise, direct measurement of actual carbon stocks will be carried out;*

h) Regions where precipitation is less or equal to potential evaporation in same period. The indirect N2O emission from leach and runoff is not considered according to Chapter 11, Volume 4 of 2006 IPCC Guidelines.

In addition, the leakage component of this methodology may restrict the applicability of this methodology because it prohibits projects that qualify to use this methodology's leakage tool and that lead to an increase of at least 50% of grazing animal displacement beyond the displacement rates expected in the baseline. In cases where baseline grazing displacement is minimal or non-existent, any grazing displacement by the project would seem to result in project ineligibility.

11.3.4.3 What are the methods for determining additionality?

This methodology requires project proponents to utilize the latest version of the "Tool for the Demonstration and Assessment Additionality in VCS Agriculture, Forestry and Other Land Use (AFOLU) Project Activities" to determine the additionality of project activities.

11.3.4.4 How are baseline soil organic carbon levels determined?

Due to applicability condition b) which requires project lands to be degraded and degrading, the methodology assumes changes of SOC levels in the baseline scenario (i.e., BRS, baseline removals due to changes in SOC) may conservatively be treated as unchanging (zero). Furthermore, since the methodology is limited to projects with grassland remaining as grassland in both the baseline and project scenarios, there is no need to address changing SOC levels under land use change or other management changes in the baseline scenario.

11.3.4.5 How is soil organic carbon sequestration measured and quantified?

This methodology accommodates the use of soil carbon models and direct measurement as two options for estimating a project's SOC sequestration. Under option 1 (carbon model), project proponents may utilize model-derived estimates for the SOC equilibrium level determined for each grassland area subject to a particular management practice, including transitions to new management practices over time (without a change from grassland land use type). The approach used for this option is practically identical to that used in the approved methodology VM0017 – Adoption of Sustainable Agricultural Land Management (VCS) (see above).

As Option 2, projects may utilize direct field sampling to estimate SOC levels in the project scenario as follows:

$$P_{SOC_{s,i,t}} = SOC_{s,i,t} \times BD_{s,i,t} \times Depth \times \left(1 - FC_{s,i,t}\right) \times F$$

[33] The details of global agroecological zones classification outlined by Food and Agricultural Organization of United Nations (FAO), Rome, Italy and International Institute for Applied Systems Analysis, Laxenburg, Austria are available at: http://www.iiasa.ac.at/Research/LUC/GAEZ/index.htm.

Where:

$P_{SOC_{s,i,t}}$	SOC stock in the top 20 cm of soil for stratum *s*, sampling site *i* under project activity in year t, tC ha^{-1}
$SOC_{s,i,t}$	SOC content in the top 20 cm of soil for stratum *s*, sampling site *i*, under project activity in year t, g C·100g^{-1} soil
$BD_{s,i,t}$	Soil bulk density in the top 20 cm of soil for stratum *s*, sampling site *i*, under project activity in year t, g·cm^{-3}
$Depth$	Top soil depth, for calculating grassland SOC stock in the top 20 cm of soil, m
$FC_{s,i,t}$	%age of rocks, roots, and other dead residues with a diameter larger than 2mm in the top 20 cm of soil, for stratum *s*, sampling site *i* under project activity in year *t*, %
F	Unit conversion coefficient turning soil carbon stock into t C ha^{-1}, in 10000m^2·ha^{-1}
s	Index of stratum
i	Index of sampling site

The estimated SOC stock ($P_{SOC_{s,i,t}}$) is then averaged over monitoring sites and strata to give a project average SOC level in year *t*. This project average SOC stock is then used to calculate annual changes in SOC levels.

Projects using Option 2 must repeat field measurement every 5 years, with 3 samples per sampling site, and samples taken to a depth of 20cm.

11.3.4.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

Due to applicability conditions b) and c) (described above), the authors argue the leakage effects of decreasing SOC, changes in fertilizer, fossil fuels, and non-renewable biomass for cooking from outside the project area may be ignored. The only remaining leakage due to land use change the authors address focuses on the potential for displacement of grazing from inside the project area to surrounding areas. This leakage tool is used for cases where the project proponent continues to own or manage the livestock that are displaced by project activities. Based on the precedent of a cited CDM tool, no leakage is assumed to occur in cases where the grazing animals are sold for slaughter or to an entity not involved in the project.

The methodology directs projects to use the latest CDM A/R grazing displacement tool[34] to calculate leakage in all cases except where the specific lands to which displaced grazing animals are expected to move cannot be directly identified, but can be reasonably determined to be grassland. In the case where the destination of displaced grazing animals can be reasonably expected to be grassland but

[34] The leakage assumptions included in this VCS methodology reflect the assumptions of an earlier (and now obsolete) "Tool for estimation of GHG emissions related to displacement of grazing activities in A/R CDM project activity". The successor tool "Estimation of the increase in GHG emissions attributable to displacement of pre-project agricultural activities in A/R CDM project activity", which is currently active, has removed several material assumptions and applicability conditions and thus may introduce some unintended errors when applied to fulfill this VCS methodology's requirement to apply the latest CDM A/R grazing displacement tool. Both CDM A/R grazing displacement tools are reviewed separately above.

cannot be specifically identified, this methodology introduces new steps to account for this potential leakage.

The remaining steps of the leakage section in this methodology address displaced grazing animals by estimating the animal unit months (AUMs) that have been displaced. This metric is compared to the grazing displacement occurring in the baseline scenario, and a discount factor is applied to project sequestration estimates if the project displaces more grazers than were displaced in the baseline scenario (capped at a 50% increase).

11.3.4.7 How is permanence of soil carbon addressed?

Potential reversals are addressed using the latest AFOLU Non-Performance Risk Tool approved by VCS (see above).

11.3.5 Proposed Module/Tool – Soil Carbon (VCS)

11.3.5.1 What are the project activities eligible for crediting?

This methodology was designed with a goal of broadening the applicability of available tools beyond those currently available. It is generally applicable to agricultural land management activities that are expected to reduce emissions from SOC as well as those that increase SOC sequestration. The applicability conditions (see below) do however appear to limit the applicability of this methodology, apparently excluding its use for avoided land use change projects such as AGC as discussed in the Issue Paper.

In the review of pre-existing methodologies, the authors concluded:

> *All of these existing proposed methodologies focus on specific elements of the ALM* [Agricultural Land Management] *continuum. The use of soil carbon prediction models such as Century and DNDC are widely applied in these methodologies. This methodology is much more general, and is designed to applicable* [sic] *to projects where a wide variety of activities are or may occur under the baseline or project scenario, such as timber harvesting, fertilization, etc.*

In the summary description of the methodology, the authors further elaborate:

> *This method has been designed to be applicable to conservation, ecosystem restoration, and agricultural projects. As well as other projects where the management of soils directly, or management of hydrology, fertility, and vegetation systems can effect changes in soils and soil carbon. The method is applicable to a range of project scenarios designed to improve soils, including changes to agricultural practices, grassland and rangeland restorations, soil carbon protection and accrual benefits from reductions in erosion, grassland protection projects, and treatments designed to improve diversity and productivity of grassland and savanna plant communities.*

11.3.5.2 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

This methodology does not place limitations based on geographic location. Baseline management practices that might be displaced by the project must either be grazing and fodder production, crop production, or timber production. In addition, the project may not include wetlands, peatlands, or mangrove ecosystems as part of the project area. The Soil Carbon Module that is a component of this methodology also explicitly prohibits its use on organic soils.

The following applicability conditions are mandatory for this methodology:

a) *Projects must meet the most recent VCS requirements for one of the following three Agricultural Land Management activities:*
    - *Improved Cropland Management (ICM)*
    - *Improved Grassland Management (IGM)*
    - *Cropland and Grassland Land-use Conversions (CGLC)*

b) *As of the project start date all of the project area consists of grasslands, rangelands, croplands or forest lands. Crops may include woody species grown for food products, fuel products, or timber. The project area may not consist of wetlands, peatlands or mangrove ecosystems, as defined by the VCS.*

c) *The project area may not have been cleared of native ecosystems within a ten year period prior to the commencement of the project activity.*

d) *The only Baseline activities that could potentially be displaced by the project activities are grazing and fodder production, crop production, and timber production.*

e) *Project activities must not include changes in surface and shallow (<1m) soil moisture regimes through flood irrigation, drainage, or other significant anthropogenic changes in the ground water table.*

In addition, the methodology includes several "optional" criteria that can be used to streamline various steps within the methodology. As each criterion is fairly specific and tied to particular step numbers in the methodology, these criteria are not repeated here.

11.3.5.3 What are the methods for determining additionality?

This methodology directs projects to apply the latest "Combined tool to identify the baseline scenario and demonstrate additionality for A/R CDM project activities." This CDM A/R tool is discussed separately in this review (see above).

11.3.5.4 How are baseline soil organic carbon levels determined?

The baseline scenario is determined by applying the latest version of the "Combined tool to identify the baseline scenario and demonstrate additionality for A/R CDM project activities."

Where the baseline scenario indicates SOC is likely to be decreasing (i.e., "degradation"), baseline SOC levels are assumed to remain constant throughout the crediting period. Otherwise, projects are required to apply the Soil Carbon Module developed as part of this methodology.

The Soil Carbon Module contains detailed guidance for designing a sampling program (described in more detail below). The Projection of Future Conditions Module contains detailed guidance for estimating baseline levels for the generic variable X and includes an elaborate decision-tree type approach to determine appropriate methods for projecting the value of the variable over the baseline period. For SOC levels, the outcome of this process is most likely to involve the use of a soil carbon model.

Any model used for the projection of any variables under this methodology must be scrutinized through a variety of tests to confirm its accuracy and precision for the project's purposes, described in more detail in the Projection of Future Conditions Module.

11.3.5.5 How is soil organic carbon sequestration measured and quantified?

Quantification of SOC levels in the project scenario must be based upon field sampling. The Soil Carbon Module contains guidance on pre-sampling, stratification, calculating the number of sample plots, a prescribed plot layout including soil pits and cores, steps to determine the desired depth of sampling, and a variety of trouble-shooting sections to address problematic soil processes such as alluvial deposition, erosion, compaction, decompaction, etc.

The number of plots for soil sampling in the field may be based upon the CDM tool "Calculation of the number of sample plots for measurements within A/R CDM project activities" or a comparable published sample plot estimation methods. The methodology requires the use of permanent sampling plots and a prescribed circular sampling layout. The depth of sampling may vary between projects, but must sample to such a depth that accounts for at least 90% of the expected change in SOC levels throughout the soil profile due to project activities, but capped at a depth of 1 meter. The Soil Carbon Module also contains guidance for confirming the accuracy of soil laboratory analysis. There is no specific monitoring interval prescribed apart from the recommendation that monitoring period be at least one year long.

11.3.5.6 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

The methodology contains two separate modules for addressing market leakage and activity-shifting (or "displacement") leakage. Market leakage is assessed where the project activities result in the reduction of the amount of product sold to local, regional, national or international markets. Depending on the barriers to participating in particular markets (price, access, etc.) and the proportion of that market supply that originated from the project area, the estimated carbon stocks of land impacted by market leakage, and the elasticity of supply and demand, varying levels of market leakage discounts will be assessed. As a general rule, if the reduction in the supply of a product due to project activities represents less than 3% of the market in which it is sold, leakage is assumed to be negligible.

> *"Only leakage which results in changes in carbon pools or carbon emissions from specific areas outside of the project zone" is considered in the Displacement Leakage Module. The authors offer a further example, "market leakage resulting in non-locatable changes in economic activities due to changes in market conditions caused by implementation of the project are not estimated."*

To address displacement leakage, this methodology allows two general approaches, one which monitors a sample of the displaced agents who are likely to be the source of emissions leaked outside the project area, and another which monitors and samples the geographic area (or “leakage zone”) surrounding the project that is expected to be impacted by leakage. Tracking displaced agents involves sampling/interviewing these agents periodically following their displacement from the project are to estimate their ongoing activities that have an impact on the carbon pools of interest. Tracking a leakage zone will involve remote sensing approaches to detect changes in carbon stocks or reliable proxies for carbon stocks in the surrounding areas over time.

11.3.5.7 How is permanence of soil carbon addressed?

Potential reversals are addressed using the latest AFOLU Non-Performance Risk Tool approved by VCS (see above).

11.4 Review of Other Relevant Documents

11.4.1 Ducks Unlimited Avoided Grassland Conversion Project in the Prairie Pothole Region – Project Design Document (CCB)

11.4.1.1 Context

This Project Design Document (PDD) describes a specific project (the Ducks Unlimited Avoided Grassland Conversion Project in the Prairie Pothole Region) intended to demonstrate the project's compliance with the Climate, Community & Biodiversity Standard. Although the document outlines the methodological steps used to determine the anticipated climate benefits for the project, it is not a carbon offset project methodology and thus cannot be applied for certification of other projects or issuance of offset credits.

11.4.1.2 What are the project activities eligible for crediting?

The project activity involves securing perpetual conservation easements on grassland areas at risk of conversion to another land use type (specifically cropland). The properties subject to these conservation easements are aggregated into a single carbon offset project.

11.4.1.3 What are the limitations due to geographic location, soil types, cropping systems or other parameters?

The project is implemented in the Prairie Pothole Region of North Dakota and South Dakota, defined as the portion of those states lying east of the Missouri River.

11.4.1.4 What are the methods for determining additionality?

In addition to describing the conversion threat to grasslands in the project region, the project applied "the project test" included in the *Voluntary Carbon Standard 2007*[35], specifically:

Test 1 - The project test:

Step 1: Regulatory Surplus

The project shall not be mandated by any enforced law, statute or other regulatory framework.

Step 2: Implementation Barriers

The project shall face one (or more) distinct barrier(s) compared with barriers faced by alternative projects.

- **Investment Barrier** – Project faces capital or investment return constraints that can be overcome by the additional revenues associated with the generation of VCUs.
- **Technological Barriers** – Project faces technology-related barriers to its implementation.
- **Institutional barriers** – Project faces financial, organizational, cultural or social barriers that the VCU revenue stream can help overcome.

Step 3: Common Practice

[35] Available online at http://v-c-s.org/sites/v-c-s.org/files/VCS%202007.pdf.

- project type shall not be common practice in sector/region, compared with projects that have received no carbon finance.
- if it is common practice, the project proponents shall identify barriers faced compared with existing projects.
- demonstration that the project is not common practice shall be based on guidance in the GHG Protocol for Project Accounting, Chapter 7.

11.4.1.5 How are baseline soil organic carbon levels determined?

The project quantifies baseline SOC levels through two primary steps:

1. *Calculating the expected annual rate of grassland conversion to cropland in the project region.* Using data from the US Department of Agriculture and spatial and economic risk assessment modeling of land-use conducted by Ducks Unlimited, future annual loss rates were estimated with a resulting range of 2-3% per year. This loss rate was then applied to project lands for 99 years, indicating an expected conversion of 73% of the project area over this period.

2. *Estimation of initial soil organic carbon stocks and changes in these stocks upon conversion to cropland.*
   Field measurements from an earlier study in the project area were compared to published values from scientific literature, and were determined to be sufficiently similar to justify the use of values from the literature in lieu of additional field sampling. Soil organic carbon levels for grassland and cultivated soils were determined from IPCC lookup values. The change in SOC stocks with the transition from grassland to cultivated cropland was calculated using an equation developed by the IPCC which describes a linear transition over the course of 20 years at which point SOC levels reach equilibrium in the new land use type:

   $$SOC_{20} = SOC_{Ref} * F_{LU} * F_{MG} * F_{I}$$

   Where
   SOC_{20} is the carbon stock after 20 years of cultivation,
   SOC_{Ref} is the quantity of SOC in an undisturbed grassland soil,
   F_{LU} is the scaling factor for the effect of land use change over 20 years
   F_{MG} is the scaling factor for the effect of management practices, and
   F_{I} is the scaling factor for the application of inputs over a period of 20 years.

   The values for SOC_{20} and SOC_{Ref} were selected from IPCC lookup tables. Default values provided by the IPCC were used for the scaling factors.

11.4.1.6 How is soil organic carbon sequestration measured and quantified?

This PDD addresses avoided loss of SOC present at project initiation and conservatively assumes that SOC stocks will remain constant over time without disturbance. The PDD does not contain methods for measuring additional carbon sequestered in the project area over time. SOC stocks are assumed to remain constant in the project scenario and are quantified in the baseline scenario using the IPCC equation identified above.

11.4.1.7 What are the methods for determining and/or accounting for leakage due to indirect land use change (ILUC)?

The project evaluated historical rates of easement purchases and compared these to rates of “new breakings” recorded by USDA Farm Service Agency that identifies new acres brought into crop production. This evaluation focused on a “leakage belt” and tested for correlations between easements and new breakings in each county using both absolute (acres) and relative (% of county area) metrics. Statistical analysis showed no significant relationship between easement purchases and new breakings requests, which the PDD uses as evidence that easement purchases do not increase cropland conversion and that leakage may be assumed to be zero.

11.4.1.8 How is permanence of soil carbon addressed?

There are two principal strategies this PDD identifies for addressing soil carbon impermanence, government monitoring of the conservation easements and an internal set-aside “buffer reserve.” The US Fish and Wildlife Service monitors the easements according to the agency’s easement manual. This includes periodic monitoring from the air with follow-up ground visits should a possible violation of easement terms be observed. In addition, DU planned to hold a buffer reserve equal to 10% of the marketable carbon credits to be utilized in the event of an easement violation.

12 Appendix E: Leakage Lit Review

"Leakage" as an offset project term is generally invoked to identify and characterize changes in GHG emissions outside the project boundary that can be attributed to the implementation of an offset project or activity. Although leakage effects can technically be positive (i.e., they induce emissions reductions outside the project boundary) or negative (i.e., they induce greater emissions outside the project boundary), most offset standards preclude the consideration of "positive" leakage effects when accounting for a project's net impact on GHG emissions. Although leakage had been identified from very early offset policy discussions and conceptual treatment of leakage effects became fairly commonplace in the literature, many of the accounting practices and policies used to account for leakage were only developed comparatively recently. Whereas changes in sources, sinks, and reservoirs (SSRs) inside a project's boundary are typically referred to as an offset project's primary emissions scope, those changes outside the boundary, such as leakage, are often defined as part of an offset project's secondary emissions scope (Climate Action Reserve, 2011a; World Resources Institute and World Business Council for Sustainable Development, 2005)

Of particular relevance for the AGC and CCG project types considered throughout this issue paper, any offset project that affects the supply of agricultural commodities may be expected to indirectly affect a change in land use outside the project boundary to compensate for the increased or decreased supply.

A significant volume of literature surrounding the concept of leakage has been devoted to characterizing the indirect land use change (ILUC) effects that were anticipated through the implementation of several renewable fuel policies that are now active in the European Union and at in the United States at the federal level and in several states.

Building from a brief treatment of the conceptual foundations for leakage and ILUC analyses, this appendix then addresses the typical accounting approaches and eventual estimates produced for leakage considering programmatic and project-level policies that have been utilized.

12.1 Leakage Concepts and Typology

Although several early publications proposed slightly different categories for leakage (e.g., Schwarze et al. (2002), Vöhringer et al. (2004)), the conceptual framework for leakage provided by Aukland et al. (2003) offers a classification reflected in many contemporary offset methodologies. To allow for greater clarity in further discussion throughout this paper, the full categorization of leakage by Aukland et al. (2003) is excerpted below and will be utilized as the basis for discussing potential leakage effects and policy options for addressing them throughout the remainder of this chapter.

The classification scheme proposed by Aukland et al. (2003) divides leakage into primary and secondary types[36], each with two sub-types of leakage:

[36] It is important to distinguish primary and secondary leakage types from primary and secondary emissions scopes for offset projects. Primary and secondary leakage types are distinguished to enable tailored project-based monitoring and accounting for situations where leakage occurs directly through displaced baseline drivers and/or

Primary leakage occurs when the GHG benefits of a project are entirely or partially negated by increased GHG emissions from similar processes in another area. Primary leakage essentially results in the displacement of the negative activity tacked by the project (the 'baseline driver'), rather than its avoidance. It is, therefore, directly related to the activities that are modeled in the baseline and the actors responsible for causing them ('baseline agents'). Primary leakage can be divided into the following sub-types:

- *Activity-shifting – the activities that cause emissions are not permanently avoided, but simply displaced to another area. For example, one discrete area is demarcated for preservation causing cattle farmers who were converting the area into pasture to simply move into another area outside of the immediate project boundaries to convert forests there.*
- *Outsourcing – the purchase or contracting out of the services or commodities that were previously produced on-site. For example, a logging company that was previously extracting timber from the project area, purchases timber from other operators to maintain an ongoing supply, e.g., for a sawmill, in the presence of the project. This differs from market effects (see below), since the outsourcing is undertaken by the baseline agent as opposed to third parties.*

Secondary leakage occurs when a project's outputs create incentives to increase GHG emissions elsewhere. Unlike primary leakage, secondary leakage activities are not directly linked to, nor carried out by, the original 'baseline agents'. Secondary leakage can be subdivided into the following sub-types:

- *Market effects – when emissions reductions are countered by emissions created by shifts in supply and demand of the products and services affected by the project. For example, an avoided deforestation or logging project may result in a decrease in the supply of timber, thereby causing a rise in timber prices and an increase in logging activities by third parties….*
- *'Super-acceptance' of alternative livelihood options – this is a particular type of leakage that may result from the alternative activities proposed by a project. For example, as part of a conservation project, alternative livelihood options may be promoted to reduce the need for conversion of the forest to agricultural land. As a result, there may be an influx of people attracted into the area from regions outside of the original 'project boundaries' or target group, who may adopt the activities promoted by the project.*

(Aukland, et al., 2003, pp. 124-125)

The implications of this categorization scheme for offset project accounting are readily observed in the methodological approaches that have been used to date to monitor and quantify each type of leakage for carbon offset projects (discussed further below). Primary leakage in offset project accounting is often based on identifying and monitoring 'baseline agents' either as individuals or as a class. In contrast, secondary leakage is mediated by indirect economic and social influences that are often assumed to follow market signals of supply and demand.

In order to translate the extensive body of work investigating the land use change impacts of biofuels into an offset project accounting context, it is important to recognize that both primary and secondary leakage types defined above for offset projects correspond only to the 'indirect' emissions from land use

indirectly through other agents. Both primary and secondary leakage types would still fit within an offset project's secondary emissions scope, defined by the Reserve's *Program Manual* as "unintended effects on GHG emissions, often associated with leakage" (Climate Action Reserve, 2011a; Climate Action Reserve, 2011b).

change invoked throughout the literature on land use change and biofuels. Plevin et al. (2010) offer the following definitions for direct and indirect emissions from land use change:

> *Indirect land use change (ILUC) emissions occur when grassland and forest are converted to cropland somewhere on the globe to meet the demand for commodities displaced by the production of biofuel feedstocks. Direct land use change, in contrast, occurs when a previous land use is converted to bioenergy crop production.*
>
> *(Plevin, et al., 2010, p. 8015)*

Because the econometric approaches to estimate emissions due to land use change in the biofuels literature are generally applied at national and global scales with a policy or program-level (as opposed to project-level) intervention, they typically obscure the distinction between land use change driven by activity-shifting or outsourcing as compared to market effects. Although some global estimations of biofuel-induced land use change, such as Melillo et al. (2009), partition model estimates into direct and indirect land use change emissions, the underlying models treat both direct and indirect (or primary and secondary) land use emissions in the same manner; that is, both direct and indirect land use changes are modeled as being fundamentally driven by the economic variables typically only considered in offset project accounting and monitoring under the scope of "market effects" (Prins, et al., 2010; Cornelissen & Dehue, 2009). This does not necessarily conflict with project-level scopes defined above, as it is also possible that activity-shifting or outsourcing effects in offset project scenarios respond primarily to market forces, particularly when the activity being displaced by the project involves the production of a commodity that is sold into national or international markets. As Murray et al. (2007) highlight, "common usage has tended to equate distant leakage with "market" leakage; but this can be misleading because both local and distant leakage essentially arise from the same underlying economic adjustments of supply and demand, so distant leakage is not necessarily more market-driven than local leakage."

12.2 Estimating Leakage

In their reviews of the various global econometric models for estimating ILUC emissions due to biofuels policies, both Cornelissen and Dehue (2009) and Plevin et al. (2010) describe the general approach to ILUC quantification with the following general steps:

1. **Determining the market-mediated scale and general location of the land/resource base required to supply the quantity of the commodity demanded.** Global economic models are used to model broad national or regional supply and demand responses based on the 'yield shock' introduced by the biofuel mandate (or by the yield forgone due to an AGC or CCG offset project).
2. **Assigning land use changes to specific locations on the ground in each country or region.** Historical patterns of land use change or models incorporating land suitability in the agriculture and forestry sectors are used to assign the specific land use changes (e.g., forest to cropland, grassland to cropland) to the ground. In biofuels modeling, ILUC considers only the land brought into production to provide non-biofuel crop yields that have been displaced by biofuels production (or offset project activity). That is, ILUC should not include the lands on which biofuel feedstocks are grown.

3. **Determining the GHG impacts of indirect land use changes.** Estimates of the carbon emissions due to ILUC, including foregone carbon sequestration, are quantified for a defined timeframe. The total GHG impact is derived from carbon stock factors and/or land use conversion factors based on available datasets that are then typically amortized linearly over a timeframe such as the period of anticipated biofuel feedstock production (or offset project lifetime).

Each of these steps is carried out through several critical input variables that vary significantly across the model approaches reviewed and described below. In general, much of the information regarding the specific assumptions and factors used for each model run are not readily available. Several of the reviews mentioned below attempt to compare the assumptions and factors across models using a consistent approach, but in general there is typically insufficient information published alongside the various model estimates of ILUC relating to biofuel mandates to allow for a comprehensive and consistent comparison of the specific factors used (e.g., Cornelissen & Dehue, 2009; EC DGE, 2010).

In their review of several model estimates for ILUC effect, Plevin et al. (2010) describe a "reduced-form model of ILUC" that distills the multitude of input factors into a simplified equation. Although this equation was originally designed to address ILUC in terms of CO_2 emissions per megajoule of biofuel-derived energy, it may be readily adapted to offset project leakage accounting as:

Equation 1: Reduced-Form Model of ILUC

$$ILUC = \frac{NDF \times EF}{T \times Y}$$

Where:

$ILUC$	emissions due to indirect land-use change; tCO_2e per unit of yield reduction (e.g., tCO_2e/bushel)
NDF	Net Displacement Factor, the ratio of land area brought into crop production to the area subject to reduced yields due to project activities; dimensionless
EF	average GHG Emission Factor for the land area brought into crop production; tCO_2e/ac
T	timeframe over which project-induced yield reductions are considered (e.g., project life); years
Y	the total yield reduction induced by project activities over the timeframe T; unit of yield reduction per acre per year (e.g., bushel/ac/yr)

Each model that has been used for estimating ILUC effects uses a much more elaborate approach with numerous inputs used to generate this handful of factors, but all essentially follow this generalized calculation. This framework provided by this simplified equation will be used to frame the discussion of model inputs below.

12.2.1 Equilibrium Models and their Primary Inputs

Indirect land use change, both in the context of a biofuels mandate as well as in offset project accounting, is an unobservable phenomenon. The attribution of indirect land use change is fundamentally based upon the difference between land use change emissions in a with-policy or with-project scenario and a without-policy or without-project scenario. For this reason, it is impossible to estimate the indirect emissions effects of land use change without invoking models that can stand in for each counterfactual scenario.

The suite of models that have been used to provide estimates of ILUC must combine economic modeling capacity that can simulate market-mediated supply and demand shifts with biological and physical modeling capacity that can simulate land use dynamics such as land productivity and changing management practices. The models used in the biofuels and ILUC literature almost universally fall into the categories of general equilibrium or partial equilibrium models. General equilibrium models take into consideration all sectors of the economy whereas partial equilibrium models only simulate one or two specific sectors of the economy (e.g., forestry and agriculture sectors). The computational complexity and uncertainties surrounding the simulation of numerous sectors typically requires that these models be run with many important simplifications such as whether to include specific sectors, crops, etc. (Prins, et al., 2010).

Several models have been developed to cover these capacities described above (and as necessary, have been paired together), although this review does not devote significant attention to characterizing each model's specific capacities and methods (apart from those highlighted under discussions of ARB and EPA . For further characterization of the features and sectoral scopes of the common ILUC models used, see the thorough review provided by European Commission Directorate-General for Energy (2010) and a comparatively brief review by Prins et al. (2010).

12.2.1.1 Inputs to Estimate the Net Displacement Factor

The Net Displacement Factor is equivalent to the primary step for calculating ILUC described by the Reserve in the Request for Proposals for this Issue Paper: "How much land will be brought *into* production (or converted to other land uses) to compensate for land taken out or precluded from production/conversion."

The sensitivity analysis from the reduced-form ILUC model of Plevin et al. (2010)—which incorporated the range of parameters used for model estimates for ILUC due to corn ethanol production by Searchinger et al.(2008), Hertel et al. (2010), Al-Riffai et al. (2010), Dumortier et al. (2009), and USEPA (2010)—indicated that NDF contributed 70% of the variance in the ILUC calculation. This should highlight the central role that this factor and the inputs from which it is derived play in the ultimate calculation of ILUC effects. In their sensitivity analysis of the reduced-form model, Plevin et al. (2010) used a range of 25% to 80% to represent a reasonable range of uncertainty for this Net Displacement Factor.

In order to determine how much cropland is brought into production to compensate for the yield reduction induced by the project activity, the NDF must incorporate the combined effects of "(i) price-induced yield increases, (ii) relative productivity of land converted to cropping, (iii) price-induced reductions in food consumption, and (iv) substitution of crop products by biofuel coproducts, such as distiller's grain replacing corn as animal feed" (Plevin, et al., 2010).

A review conducted by the European Commission Directorate-General for Energy (2010) identified several components that typically feed into the yield-based aspects of this calculation, most of which are still subject to considerable debate:

- Change in inputs (e.g., fertilizer, irrigation, herbicides, pesticides, etc.) in response to demand

- Changes in frequency of cropping
- Changes in technological development in response to demand
- Yields on land that has been converted to cropland from grassland or forest
- Yields on land converted from one crop to another

The suite of determinants of NDF includes several economic variables such as the price elasticity of supply and demand for cropland and/or the specific crop commodity. These metrics are closely related to assumptions about how much of the "yield shock" caused by a biofuel mandate could be addressed through intensification of agricultural production on existing cropland and how much would need to be met by converting additional grassland or forest areas to new cropland. These assumptions should reflect the expected change in productivity in both the project area and area where ILUC occurs throughout the timeframe of interest (e.g., the project lifetime). One aspect that makes this parameterization particularly challenging and highly variable among the published model analyses is that these variables regarding economic and land use assumptions, which dominate overall model behavior, have to be determined for each country or region being modeled now and into the future.

12.2.1.1.1 An Approach to Estimating the Net Displacement Factor

In theory, the response of supply and demand of agricultural land may be represented in an aggregate way through shifts in the supply and demand curves for agricultural land. Although it requires a significant simplifying assumption that only one market is involved, Vöhringer et al. (2004; 2006), Chomitz (2002), and Murray and Baker (2010) have each derived comparable equations to quantify the market-mediated effects of a reduction in supply based on an offset project's activity that is intended to function as a project-based discount factor. Applied in the context of land set-asides, the following equation can be applied to generate a proportional leakage estimate on an area-basis[37]:

Equation 2: Simplified Leakage Discount Equation

$$LF = \frac{\varepsilon_S}{\varepsilon_S - \varepsilon_D}$$

Where:

LF	Leakage Factor, the proportion of a project's emissions reductions that is leaked through market effects; dimensionless
ε_S	Price elasticity of supply, $\varepsilon_S \geq 0$; dimensionless
ε_D	Price elasticity of demand, $\varepsilon_D \leq 0$; dimensionless

The dynamics of this leakage factor equation are such that as ε_S increases (i.e., supply responds more strongly to price) the leakage factor increases. Conversely, as demand becomes more elastic (i.e., responds more strongly to price, as indicated when ε_D decreases) the leakage factor decreases. In plain terms, higher price elasticity of supply means more cropland will be brought into cultivation in response to a price increase. Higher elasticity of demand means consumers reign in consumption as prices rise,

[37] That is, when the elasticity factors are defined in terms of the cropland acreage response to changes in prices, the leakage factor quantifies the area that will be brought into crop production elsewhere in the world as a proportion of the area taken out of crop production by offset project activities.

leading to reduced demand for new cropland area. A conservative approach to offset leakage accounting (i.e., one that is more likely to overestimate leakage) would thus err on the side of using higher values for ε_S and ε_D (i.e., values for ε_D closer to 0 since ε_D is negative).

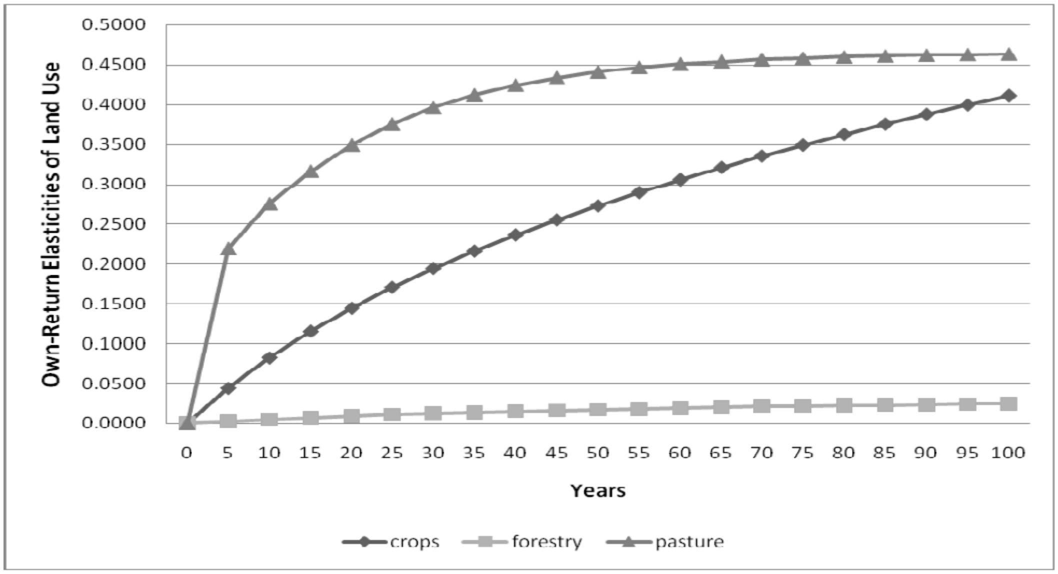
In cases where demand does not respond to price (i.e., $\varepsilon_D = 0$, perfectly inelastic), the leakage factor is 100%. It is important to note that any policy which assumes perfectly inelastic demand would result in both AGC and CCG projects providing zero GHG benefits since all crop production activities precluded by the project would be determined to have completely leaked through market effects or activity-shifting.[38]

Theoretically, the price elasticity of supply and demand for cropland area (distinct from price elasticities for yield) should reflect the various determinants that also go in to model inputs generating the NDF. These values have typically only been provided for price elasticity of supply for cropland area and the values in published studies vary widely. For example, land area supply elasticities in the range of 0.02-0.03 have been observed by Swinton et al. (2011) and Barr et al. (2011) and estimates in the range of 0.25-0.3 found in Roberts and Schlenker (2010) and Berry and Schlenker (2011). Huang and Khanna (2010) found a U.S. cropland area elasticity of 0.26, and reported a range of values from eight other studies not mentioned above with elasticities ranging from as low as 0.05 up to as high as 0.95.

As indicated in Figure 34, below, elasticity of supply with respect to land value is larger over longer timeframes, meaning producers will respond more strongly to higher prices, converting more and more acres to cropland the longer price increases are sustained. The estimates produced by Ahmed et al. (2008) (and shown in Figure 34) show the elasticity of supply for cropland area in the United States growing from approximately 0.05 to 0.08 to 0.17 and 0.27 over five-, ten-, twenty five-, and fifty-year timeframes, respectively.

[38] For example, the Reserve's *Nitrogen Management Project Protocol v1.0* and *Rice Cultivation Project Protocol v1.0* both contain implicit assumptions in their secondary emissions due to shifting crop production that the quantified reduction in yield is 100% leaked (Climate Action Reserve, 2011b; 2012). Because yields are not reduced to zero in these project types, this policy does not necessarily make these project types non-viable. However, project types such as AGC and CCG that reduce yields to zero would be have all primary effect emissions discounted to zero due to the secondary effects calculated under such a policy, making such an approach impracticable.

Figure 34: Own-return elasticities of crop, forest, and pasture land uses in the United States



Source: Ahmed et al. (2008)

Hertel (2010) provides a simple approach to "back out" estimates of the arc elasticity of aggregate land supply with respect to price from some of the biofuel ILUC modeling literature available and finds them to be comparable to those of Barr et al. (2011), which are generally close to zero. Arguing that the biofuel ILUC analyses often rely upon short-term elasticities, however, Hertel concludes:

> *…there are likely very large differences between the short and long run elasticities of land supply to agriculture, and particularly to crops. Furthermore, it appears that those researchers undertaking long run analysis (say over 40 years time) may not fully recognize this large difference – placing excessive weight on the near term changes which can be readily observed in the data – such as the response of world markets to the 2007/2008 commodity price boom. By applying near term land supply elasticities in the context of long run analysis, economists will over-predict the price impacts of exogenous shocks such as climate change and biofuel policies, while also arriving at potentially misleading conclusions regarding global land use change in agriculture.*
>
> *(Hertel, 2010, p. 42)*

This observation by Hertel (2010) suggests that the elasticity values chosen for modeling project-based market effects leakage should ideally have comparable timeframes. Long-run estimates of supply elasticity are typically larger than the values estimated from shorter timeframes. This means that projects with an expected lifetime longer than the timeframe for which the land supply elasticity was determined may be systematically underestimating leakage due to market effects. At the same time, the higher elasticity values often reported over longer (e.g., 50+ year) timeframes may also be excessively large for offset projects designed for 5, 10, or even 20 years.

12.2.1.2 Inputs to Estimate the Average Emission Factor

There are two principal inputs used to estimate the Average Emission Factor of a land-use change: the fraction of land-use change assigned to each land cover type (e.g., grassland, forest, wetland) and the time-bound GHG impacts attributed to the conversion of each land cover type to cropland (Plevin, et al., 2010).

Global models simulating ILUC have taken two general approaches to allocating land conversion across the available land covers: one relies exclusively upon historical rates of conversion; the other relies upon a bio-physical and economic consideration of land suitability in terms of productivity, accessibility, etc. (EC DGE, 2010). Historical rates are often determined through classification of satellite imagery to determine land-use transitions over a particular timeframe for each country or region where ILUC is being modeled. According to the review by the European Commission Directorate-General for Energy (2010), examples of the historical approach include Searchinger et al. (2008) and ARB (2009a); examples of the suitability approach include the CARD (Dumortier, et al., 2009) and IIASA (Fischer, et al., 2009) models; both IFPRI (Bouët, et al., 2010) and USEPA (2010) provide hybrid examples. Reflecting the spectrum of estimates for the proportional allocation of cropland expansion to other land use types, Plevin et al. (2010) give a range of 15% to 50% for forest, 0% to 2% for wetlands, and 48% to 85% for grasslands.

To estimate the GHG impact for each land type subject to conversion, these models generally rely on four datasets/approaches for providing current carbon stocks: IPCC default factors; a global terrestrial carbon dataset from Woods Hole; ecosystem modeling; and direct use of carbon stock values in scientific literature. Because the selection of these carbon stock values remains only one among several important factors needed to determine the GHG impact of converting each land type into cropland, this literature review does not provide further detailed comparison among these datasets.

Following estimation of the carbon stocks for each land type, modelers must then make assumptions about the proportion of carbon lost from pre-existing vegetation and soils upon conversion, GHG emissions from sources related to land conversion and management, as well as if and how to quantify foregone sequestration in forests, grasslands, and wetlands converted to cropland or pasture. As demonstrated in several review articles, each modeling exercise includes a suite of different assumptions for these variables which result in a wide range of GHG impacts assumed to result from land conversion (Cornelissen & Dehue, 2009; EC DGE, 2010; Plevin, et al., 2010; Prins, et al., 2010).

The sensitivity analysis of the reduced-form model from Plevin et al. (2010) used a range of 47-195 tCO_2e/ac (116-481 tCO_2e/ha) for the average emission factor. A comparison in European Commission Directorate-General for Energy (2010) showed a range from 18-142 tCO_2e/ac (44-350 tCO_2e/ha) in average emissions factors applied across five studies.

12.2.1.3 Inputs to Timeframe and Yield Reduction Factors

Both of these variables in the reduced-form ILUC equation are relatively straightforward compared to the variety of approaches that have been taken for the other factors in the reduced-form equation. The timeframe used across biofuel ILUC studies varies, but typically is used as 20 or 30 years. For offset projects, it would be consistent to simply apply the project lifetime as the timeframe (T) in the reduced-form equation.

The Yield Reduction Factor is subject to some of the considerations mentioned above for the Net Displacement Factor, but is comparatively much simpler since it deals with estimating yields only for a single location (i.e., the project area). Several assumptions will still need to be made regarding the yield that would have been expected from the offset project area over the project lifetime such as yield growth and the emissions profile of future yields and agricultural practices.

12.3 Review of Leakage Policies and Estimates

12.3.1 Leakage Policies

Leakage due to indirect land use change has been addressed in biofuel mandate policies as well as through offset project protocols. In general, the biofuel mandate policies have involved elaborate modeling exercises (often with limited publication of parameters) while offset project protocols generally follow simpler approaches that reduce the complexity of the estimation process but also increase the ability to transparently communicate how these factors have been calculated.

12.3.1.1 ILUC Modeling in Recent Biofuel Policies

In preparation for the Low Carbon Fuel Standard, the California Air Resources Board (ARB) utilized the global general equilibrium model that has been developed by researchers at Purdue University known as the General Trade Analysis Project (GTAP) model(ARB, 2009a). ARB also considered using a combination of the FAPRI and FASOM models; GTAP was ultimately considered by ARB to be the most suitable model "based primarily on its global scope, public availability and its long history of use in modeling complex international economic effects" (ARB, 2009b, pp. C-14). Reviews conducted by Cornelissen and Dehue (2009) and European Commission Directorate-General for (2010) provide a more detailed comparison of the use of the GTAP model by ARB to other model analyses.

During the development of the US Renewable Fuel Standard, the US EPA utilized a combination of models to simulate domestic and international ILUC(US EPA, 2010). The Forestry and Agricultural Sector Optimization Model (FASOM) maintained by Texas A&M University was used for domestic modeling and the Food and Agricultural Policy Research Institute (FAPRI) model maintained by the Center for Agricultural and Rural Development at Iowa State University was used for international modeling.

As indicated in several reviews of the models that have been used to estimate ILUC due to various biofuel policies, there are a variety of models that can be reasonably applied to estimate ILUC emissions. In terms of global coverage and applicability, Prins et al. (2010) assert that GTAP, LEITAP, MIRAGE, AGLINK/COSIMO and FAPRI all have the critical components are suitable for estimating ILUC emissions. Among the models covered in the literature discussed in this review, there do not appear to be any clear outliers in terms of models that would be poorly-suited for modeling the effects of an offset project policy for CCG or AGC projects.

One potential alternative to performing a full model analysis for offset protocol development may be to commission further investigation into the range of parameters and factors developed and determined by these models to provide a range of values that could be input to a reduced-form model.

12.3.1.1.1 General Conclusions Regarding Global ILUC Modeling

In general, there are a multitude of choices and simplifications made in each model in order to simulate global economic and land-use responses to a biofuel mandate. Comparative reviews and sensitivity analyses comparing the range of values used in the various models have shown that there is currently no consensus regarding specific ILUC/leakage factors for biofuel mandates and that the range of input values currently in use may lead to dramatically different results (Cornelissen & Dehue, 2009; EC DGE, 2010; Plevin, et al., 2010; Prins, et al., 2010).

Choosing a specific model or combination of models to simulate market effects leakage due to AGC or CCG offset project policies would require a detailed consideration of numerous model characteristics and assumptions beyond the level of detail that can be provided in this review. As discussed in Section 6.5 above of the issue paper, the authors of this report encourage the Reserve to consider a simplified model for estimating market effects leakage, primarily for greater transparency in documentation of policy choices. The discussion of modeling exercises is therefore relatively brief. Should the Reserve nevertheless be interested in pursuing a customized modeling exercise for the purposes of offset project policy evaluation, a more detailed consultation, particularly with the experts affiliated with these models, is recommended.

12.3.1.2 ILUC in Offset Standards and Project Methodologies

Offset standards organizations and project protocols have taken a wide variety of approaches to account for (or dismiss accounting for) leakage. These approaches generally fall into three broad categories:

- **Applicability conditions:** offset methodologies may apply applicability conditions that would effectively ensure activity-shifting or market-effects leakage would be zero or otherwise insignificant. For example, a requirement that projects not result in the reduction of yield compared to baseline would eliminate the risk of market effects leakage (at least in terms of emissions due to displaced agricultural production).
- **Survey:** offset methodologies may require ongoing monitoring and collection of data on baseline drivers. This is often accomplished through remote sensing to detect ongoing land use changes or disturbance in a region surrounding the project area or through ground-based surveys of baseline drivers commonly referred to as “Participatory Rural Appraisals.”
- **Factor-based discounts:** offset methodologies may apply a decision-tree or other approach to categorize the likelihood that a project has produced activity-shifting or market effects leakage. The outcome of these procedures results in the selection of a leakage discount factor.

A brief review of the leakage approaches adopted across 36 program standards, methodologies, and tools is presented in Table 26: Leakage approaches across offset standards and methodologies.

Table 26: Leakage approaches across offset standards and methodologies

	Activity-Shifting				Market Effects			
	Applicability	Survey	Factor	NA	Applicability	Survey	Factor	NA
American Carbon Registry Methodologies								
100-year Improved Forest Management (IFM) on U.S. Timberlands				X			X	
Afforestation and Reforestation of Degraded Lands				X			X	
Forest Carbon Project Standard v2.1		X					X	
Improved Forest Management (IFM) on Non-Federal U.S. Forestlands				X			X	
Improved Forest Management (IFM) on U.S. Timberlands, v1.0				X			X	
N_2O Emission Reductions through Changes in Fertilizer Management	X				X			
N_2O Emission Reductions through Reduced Use of Fertilizer on Agricultural Crops	X				X			
REDD – Avoiding Planned Deforestation		X					X	
REDD Methodology Modules		X					X	
Restoration of Degraded Deltaic Wetlands of the Mississippi Delta	X				X			
Emission Reductions in Rice Management Systems	X						X	
Avoided Conversion of Grasslands and Shrublands to Crop Production		X					X	
Chicago Climate Exchange Methodologies								
Continuous Conservation Tillage and Grassland Conversion				X				X
Forestry Carbon Sequestration				X				X
Sustainably Managed Rangeland Soil Carbon Sequestration				X				X
Clean Development Mechanism Methodologies								
AMS-I.E.: Switch from non-renewable biomass for thermal applications by the user --- Version 5.0		X						X
Calculation of GHG emissions due to leakage from increased use of non-renewable woody biomass attributable to an A/R CDM project activity (Version 01)		X						X
Estimation of the increase in GHG emissions attributable to displacement of pre-project agricultural activities in A/R CDM project activity (Version 01)		X	X					X
General guidance on leakage in biomass project activities (Version 03)			X					X
Climate Action Reserve Methodologies								
Forest Project Protocol, v3.3			X				X	
Nitrogen Management Project Protocol, v1.0			X				X	
Rice Cultivation Project Protocol, v1.0			X				X	

	Activity-Shifting				Market Effects			
	Applicability	Survey	Factor	NA	Applicability	Survey	Factor	NA
Verified Carbon Standard Methodologies								
Adoption of Sustainable Agricultural Land Management, v1.0		X						X
AFOLU Requirements v3.2 – Afforestation, Reforestation and Revegetation		X	X					X
– Agricultural Land Management		X	X					X
– Avoided Conversion of Grasslands and Shrublands		X						X
– Improved Forest Management				X			X	
– Peatland Rewetting and Conservation		X					X	
– Reduced Emissions from Deforestation and Degradation		X					X	
Avoided Mosaic Deforestation of Tropical Forests, v1.1		X				X		
Avoided Unplanned Deforestation, v1.0		X				X		
Calculating GHG Benefits from Preventing Planned Degradation, v1.0		X						X
Carbon Accounting in Project Activities that Reduce Emissions from Mosaic Deforestation and Degradation, v1.0		X			X			
Conservation Projects that Avoid Planned Land Use Conversion in Peat Swamp Forests, v1.0		X					X	
Improved Forest Management Projects in Temperate and Boreal Forests				X			X	
Improved Forest Management through Extension of Rotation Age, v1.0	X						X	
Improved Forest Management: Conversion from Logged to Protected Forest, v1.1	X						X	
REDD Methodology Modules (REDD-MF), v1.1		X					X	
Adoption of Sustainable Grassland Management through Adjustment of Fire and Grazing	X							X
Agricultural Land Management: Improved Grassland Management	X						X	
Avoided Planned Conversion of Grasslands and Shrublands to Crop Production		X					X	
Soil Carbon		X					X	
Sustainable Grassland Management (SGM)			X					X

Notes: "Applicability" refers to methodologies that address leakage through applicability criteria that render the anticipated leakage effects to be prohibited or insignificant. "Survey" refers to methodologies that address leakage through the collection of data in the field through participatory rural appraisal, field sampling, and/or the use of remote sensing imagery, etc. "Factor" refers to methodologies that address leakage through the application of generalized leakage factors based on a decision-tree or similar categorization approach. "NA" means the methodology lacks detailed methodological guidance or steps for addressing leakage. Virtually every standard either prohibits or requires project proponents to report any internal activity-shifting. Those methodologies that do not additionally require monitoring or reporting where other agents may be responsible for activity-shifting leakage are listed as "NA." Rows printed in italic and gray color represent methodologies that are under development and have not yet been fully approved for use.

As shown in Table 26, market effects leakage in offset project protocols is generally handled through the use of discount factors based on a decision-tree type process. Only one standard to date has discussed the use of econometric modeling to estimate market leakage. In its treatment of leakage for Afforestation/Reforestation projects in the Forest Carbon Project Standard v2.1, the American Carbon Registry states:

> *Given uncertainty about the accuracy of econometric modeling, ACR does not apply modeled market leakage rates. If there are multiple, peer-reviewed studies on market leakage rates that*

establish certainty within the industry and demonstrate that leakage likely exceeds the de minimis threshold, ACR may approve a methodology and adopt those leakage rates as part of a future iteration of this standard.

(ACR, 2010, p. 38)

12.3.2 Leakage Estimates

Table 27 below provides a summary of the various leakage estimates and acreage elasticities covered in the literature. Although the literature on leakage includes several studies estimating leakage in forestry projects due to reduced timber production, these estimates are not reproduced in the table below. The studies cited below are considered to be more directly relevant to the consideration of leakage effects for AGC and CCG project types mediated by the US and global markets for commodity crops as opposed to timber markets.

Cropland set-aside programs in the US, particularly the Cropland Reserve Program (CRP), have been a long-standing focus for agricultural economists investigating the rates at which enrolled acres lead to program "slippage," roughly synonymous with current discussions of leakage.

The approaches used to estimate slippage vary considerably across these studies, including general equilibrium modeling (Taheripour, 2006), spatial regression using satellite imagery (Fleming, 2010), and the more common linear regression models (e.g., Love & Foster, 1990; Wu, 2000; Roberts & Bucholtz, 2005). Earlier approaches estimating slippage have also simply compared the change in idled acreage to the change in harvested acreage(e.g., Ericksen & Collins, 1985), but the values in those studies are much more variable and subject to a variety of confounding factors and are not considered alongside more rigorous methods used to derive slippage estimates below. It is also important to consider that earlier studies considering land set-aside programs primarily from an agricultural commodity price-support objective (e.g., Love & Foster, 1990) may quantify slippage based on increases in aggregate yields due to cultivation of new land (i.e., ILUC) as well as due to intensification of existing cropland. Thus, for some of the studies presented below, slippage estimates may be larger than what would be more narrowly considered within the scope of ILUC leakage.

Table 27: Leakage and elasticity factors for cropland area in published studies

Study	Geographic Scope of Expansion	Timeframe	Leakage Estimate (Land Area)	Supply Elasticity	Demand Elasticity	Specific Crop?
Ahmed et al. (2008)	US	1982-1996	--	0.05 (5-yr);	--	--
			--	0.12 (15-yr);	--	--
			--	0.17 (25-yr)	--	--
ARB(2009b)	US	NA	9% to 16%*	--	--	--
	global		21% to 43%*	0.5	--	--
Barr et al. (2011)	US	1995-2009	--	0.01 to 0.03	--	--
Berry and Schlenker (2011)	US	1950-2010	--	0.27	--	--
			--	0.23 to 0.30	--	corn
			--	0.19 to 0.36	--	soybeans
			--	0.34 to 0.37	--	wheat
	global		--	0.09	--	--

Study	Geographic Scope of Expansion	Timeframe	Leakage Estimate (Land Area)	Supply Elasticity	Demand Elasticity	Specific Crop?
Chavas and Holt (1990)	US	1954-1985	--	0.16	--	corn
	(N Central)		--	0.44	--	soybeans
Fleming (2010)	US	1992-2001	3.7% to 3.8%‡	--	--	--
Goodwin and Mishra (2006)	US	1998-2001	--	0.03	--	corn
			--	0.02	--	soybeans
			--	0.03 to 0.04	--	wheat
Hertel et al. (2010)	global	NA	25% to 28%	--	--	--
	US	NA	11%	--	--	--
Huang and Khanna (2010)	US	1977-2007	--	0.25 to 0.26	--	--
			--	0.38 to 0.51	--	corn
			--	0.31 to 0.49	--	soybeans
			--	0.07 to 0.12	--	wheat
Lin and Dismukes (2007)	US	1991-2001	--	0.17 to 0.35	--	corn
	(N Central)		--	0.30	--	soybeans
			--	0.25 to 0.34	--	wheat
Lin et al. (2000)	US	1991-1995	--	0.29	--	corn
			--	0.27	--	soybeans
			--	0.34	--	wheat
			--	0.47	--	cotton
			--	0.28	--	barley
			--	0.55	--	sorghum
			--	0.44	--	oats
Love and Foster (1990)	US	1964-1986	45% to 57%	--	--	corn
			28% to 37%	--	--	soybeans
			27% to 36%	--	--	wheat
Roberts and Bucholtz (2005)	US	1982-1992	2% to 19%	--	--	--
Roberts and Schlenker (2010)	global	1961-2007	--	0.06 to 0.08	--	--
Searchinger et al. (2008)	global	NA	72% or 84%†	--	--	--
Swinton et al. (2011)	US	2006-2009	--	0.02 to 0.04	--	--
Taheripour (2006)	US	1984-2004	18% to 21%	--	-0.46 to -0.33	--
US EPA (2010)	global	NA	29% to 89%†			
Villoria and Hertel (2011)	global	1975-2002	--	0.02 to 0.04	--	--
Wu (2000)	US	1982-1992	15% to 30% (20% avg)	--	--	--

* The leakage values for US and global scopes were calculated from ARB(2009b) by dividing the estimated indirect land conversion areas given in Table IV-10 by the farmland required to produce the corn ethanol feedstock, calculated using a yield of 151 bu/ac, 2.8 gal ethanol/bu, and converting from acres to hectares.

‡ When adjusted for spatial autocorrelation, leakage estimates are reduced to 2.9%.

† The leakage values shown in this table for Searchinger et al. (2008) and USEPA (2010) are not directly provided in these references. Values in the table above are as cited in Plevin et al. (2010). Applying the method described in Plevin et al. (2010), however, the authors of this review derived an estimated leakage factor of 84% for the Searchinger et al. (2008) study (i.e., 10.8 Mha brought into production following diversion of 12.8 Mha to corn ethanol production, 10.8/12.8=0.84).

13 References

ACR, 2010. *Forest Carbon Project Standard v2.1,* Arlington: American Carbon Registry.

AECOM, 2009. *Loss of Central Valley Vernal Pools: Summary Report,* Auburn: Place Land Trust.

Ahmed, S. A., Hertel, T. W. & Lubowski, R., 2008. *Calibration of a Land Cover Supply Function Using Transition Probabilities,* West Lafayette: GTAP Research Memorandum No. 14.

Al-Riffai, P., Dimaranan, B. & Laborde, D., 2010. *Global Trade and Environmental Impact Study of the EU Biofuels Mandate,* Washington, DC: International Food Policy Institute.

ARB, 2009a. *Proposed Regulation to Implement the Low Carbon Fuel Standard, Volume 1. Staff Report: Initial Statement of Reasons..* Sacramento, Air Resources Board, California Environmental Protection Agency.

ARB, 2009b. *Proposed Regulation to Implement the Low Carbon Fuel Standard, Volume II. Appendices B - G,* Sacramento: California Air Resources Board.

Aukland, L., Moura Costa, P. & Brown, S., 2003. A conceptual framework and its application for addressing leakage: the case of avoided deforestation. *Climate Policy,* Volume 3, pp. 123-136.

Baldwin, R., 1995. *Modeling Ruminant Digestion and Metabolism.* London: Chapman and Hall.

Barr, K. J. et al., 2011. Agricultural Land Elasticities in the United States and Brazil. *Applied Economic Perspectives and Policy,* 33(3), pp. 449-462.

Berry, S. & Schlenker, W., 2011. *Technical Report for the ICCT: Empirical Evidence on Crop Yield Elasticities,* Washington, DC: The International Council on Clean Transportation.

Best, E. & Jacobs, F., 1997. The influence of raised water table levels on carbon dioxide and methane production in ditch dissected peat grasslands in the Netherlands. *Ecological Engineering,* Volume 8, pp. 129-144.

Bhandral, R., Saggar, S., Bolan, N. & Hedley, M., 2007. Transformation of nitrogen and nitrous oxide emission from grassland soils as affected by compaction. *Soil and Tillage Research,* Volume 94, pp. 482-492.

Bidwell, T. & Woods, B., 2010. *Reseeding Marginal Cropland to Perennial Grass, Forbs, and Legumes.* Stillwater(OK): Oklahoma State University Cooperative Extension Service.

Black, S., Shepherd, M. & Vaughan, M., 2011. Rangeland Management for Pollinators. *Rangelands,* 33(3), pp. 9-13.

Blanco-Canqui, H. & Lal, R., 2008. *Principles of Soil Conservation and Management.* New York: Springer-Verlage.

Bouët, A., Dimaranan, B. V. & Valin, H., 2010. *Modeling the Global Trade and Environmental Impacts of Biofuel Policies,* Washington, DC: International Food Policy Research Institute.

Breetz, H., Fisher-Vanden, K., Jacobs, H. & Schary, C., 2005. Trust and communication: Mechanisms for increasing farmers' participation in water quality trading. *Land Economics,* Volume 81, pp. 170-190.

Bremer, E., Janzen, H. H. & McKenzie, R. H., 2002. Short-term impact of fallow frequency and perennial grass on soil organic carbon in a Brown Chernozem in southern Alberta. *Canadian Journal of Soice Science,* Volume 82, pp. 481-488.

Brunson, M. & Shindler, B., 2008. *Public and Stakeholder Priorities for Great Basin Rangeland Management.* [Online]
Available at:
http://www.sagestep.org/events/or_presentations_08/Sociopolitical_Brunson/Sociopolitical_Brunson.htm
[Accessed 26 September 2012].

Buell, G. R. & Markewich, H. W., 2004. Data Compilation, Synthesis, and Calculations Used for Organic-Carbon Storage and Inventory Estimates for Mineral Soils of the Mississippi River Basin. In: H. W.

Markewich, ed. *Soil-Carbon Storage and Inventory of the Continental United States.* Reston: US Geological Survey.

CA DOC, 2010. *The California Land Conservation (Williamson) Act: 2010 Status Report,* Sacramento: California Department of Conservation.

Cary, J. & Wilkinson, R., 1997. Perceived Profitability and Farmers' Conservation Behaviour. *Journal of Agricultural Economics,* Volume 48, pp. 13-21.

Caviglia, J. & Kahn, J., 2001. Diffusion of Sustainable Agriculture in the Brazilian Tropical Rainforest: A Discrete Choice Analysis. *Economic Development and Cultural Change,* Volume 49, pp. 311-333.

CCX, 2009. *Sustainably Managed Rangeland Soil Carbon Sequestration Offset Project Protocol,* Chicago: Chicago Climate Exchange.

CDM EB, 2008. *Tool for the identification of degraded or degrading lands for consideration in implementing CDM A/R project activities (Version 01),* Bonn: Clean Development Mechanism Executive Board.

Chanton, J. P. et al., 1995. Radiocarbon evidence for the substrates supporting methane formation within northern Minnesota peatlands. *Geochimica et Cosmochimica Acta,* 59(17), pp. 3663-3668.

Chapin, F. S. I., Matson, P. A. & Mooney, H. A., 2002. *Principles of Terrestrial Ecosystem Ecology.* New York: Springer-Verlag.

Chavas, J.-P. & Holt, M. T., 1990. Acreage Decisions Under Risk: The Case of Corn and Soybeans. *American Journal of Agricultural Economics,* 72(3), pp. 529-539.

Chomitz, K. M., 2002. Baseline, leakage and measurement issues: how do forestry and energy projects compare?. *Climate Policy,* Volume 2, pp. 35-49.

Claassen, R. et al., 2004. *Environmental Compliance in U.S. Agricultural Policy: Past Performance and Future Potential,* Washington, DC: USDA Economic Research Service.

Claassen, R., Carriazo, F., Cooper, J. & Hellerstein, D., 2011a. *Do Farm Programs Encourage Native Grassland Losses.* [Online]
Available at: http://www.ers.usda.gov/amber-waves/2011-september/native-grassland.aspx
[Accessed 27 September 2012].

Claassen, R. et al., 2011b. *Grassland to Cropland Conversion in the Northern Plains: The Role of Crop Insurance, Commodity, and Disaster Programs,* Washington, DC: USDA Economic Research Service.

Climate Action Reserve, 2011a. *Program Manual (October 26, 2011),* Los Angeles: Climate Action Reserve.

Climate Action Reserve, 2011b. *Rice Cultivation Project Protocol, v1.0,* Los Angeles: Climate Action Reserve.

Climate Action Reserve, 2012. *Nitrogen Management Project Protocol, v1.0,* Los Angeles: Climate Action Reserve.

Cornelissen, S. & Dehue, B., 2009. *Summary of approaches to accounting for indirect impacts of biofuel production,* Utrecht: Ecofys.

Davidson, E. & Ackerman, I., 1993. Changes in Soil Carbon Inventories Following Cultivation of Previously Untilled Soils. *Biogeochemistry,* 20(3), pp. 161-193.

de Soyza, A. G. et al., 2000. Indicators of Great Basin rangeland health. *Journal of Arid Environments,* 45(4), pp. 289-304.

Debano, S., 2006. Effects of livestock grazing on above ground insect communities in semi-arid grasslands of southeastern Arizona. *Biodiversity and Conservation,* Volume 15, pp. 2547-2564.

Dell, R. et al., 2012a. *Avoided Conversion of Grasslands and Shrublands to Crop Production (Public Comment Draft),* Arlington: American Carbon Registry.

Dell, R. et al., 2012b. *Avoided Planned Conversion of Grasslands and Shrublands to Crop Production (Public Comment Draft),* Washington, DC: Verified Carbon Standard.

DeLuca, T. & Zabinski, C., 2011. Prairie ecosystems and the carbon problem. *Frontiers in Ecology and the Environment,* Volume 9, pp. 407-413.

Denholm, P. & Margolis, R. M., 2008. Land Use Requirements and the Per-Capita Solar Footprint for Photovoltaic Generation in the United States. *Energy Policy,* Volume 36, pp. 3531-3543.

Derner, J., Boutton, T. & Briske, D., 2006. Grazing and ecosystem carbon storage in the North American Great Plains. *Plant and Soil,* Volume 280, pp. 77-90.

Derner, J. & Jin, V., 2012. Soil Carbon Dynamics in Rangeland Management . In: M. Liebig, A. Franzluebbers & R. Follett, eds. *Managing Agricultural Greenhouse Gases: Coordinated Agricultural Research through GRACEnet to Address our Changing Climate.* Waltham: Academic Press, pp. 79-82.

Derner, J. & Schuman, G., 2007. Carbon sequestration and rangelands: a synthesis of land management and precipitation effects. *Journal of Soil and Water Conservation,* 62(2), pp. 77-85.

Dijkstra, J., Neal, H., Beever, D. & France, J., 1992. Simulation of nutrient digestion, absorption, and outflow in the rumen: Model description. *Journal of Nutrition,* 122(11), pp. 2239-2256.

Doering, O. & Smith, K. R., 2012. *Examining the Relationship of Conservation Compliance & Farm Program Incentives,* Washington, DC: The Council on Food, Agricultural and Resource Economics.

Ducks Unlimited and The Eco Products Fund, 2009. *Ducks Unlimited Avoided Grassland Conversion Project in the Prairie Pothole Region,* Arlington: Climate, Community & Biodiversity Alliance.

Dumortier, J. et al., 2009. *Sensitivity of Carbon Emission Estimates from Indirect Land-Use Change,* Ames: Center for Agricultural and Rural Development, Iowa State University.

Eagle, A. J. et al., 2012. *Greenhouse gas mitigation potential of agricultural land management in the United States: A synthesis of the literature (Third Edition),* Durham: Technical Working Group on Agricultural Greenhouse Gases.

EC DGE, 2010. *The Impact of Land Use Change on Greenhouse Gas Emissions from Biofuels and Bioliquids – Literature Review,* Brussels: European Commission Directorate-General for Energy.

Ericksen, M. H. & Collins, K., 1985. *Effectiveness of Acreage Reduction Programs,* Washington, DC: Economic Research Service, U.S. Department of Agriculture.

Fambrough, J., 2009. *Minerals, Surface Rights and Royalty Payments,* College Station: Real Estate Center, Texas A&M University.

Fargione, J., Kiesecker, J., Slaats, M. & Olimb, S., 2012. Wind and Wildlife in the Northern Great Plains: Identifying Low-Impact Areas for Wind Development. *PLoS ONE,* 7(7), p. e41468.

Feder, G., Just, R. & Zilberman, D., 1985. Adoption of Agricultural Innovations in Developing Countries: A Survey. *Economic Development and Cultural Change,* Volume 33, pp. 255-298.

Fischer, G. et al., 2009. *Biofuels and Food Security,* Vienna: OPEC Fund for International Development.

Fleming, D. A., 2010. *Slippage Effects of the Conservation Reserve Program: New Evidence from Satellite Imagery.* Denver, Agricultural & Applied Economics Association.

Flynn, A. J. et al., 2009. *Soil Carbon Sequestration in U.S. Rangelands: Issues Paper for Protocol Development,* Sacramento: Environmental Defense Fund.

Follett, R., Kimble, J. & Lal, R., 2001. The Potential of U.S. Grazing Lands to Sequester Soil Carbon. In: R. Follett, J. Kimble & R. Lal, eds. *The potential of U.S. grazing lands to sequester carbon and mitigate the greenhouse effect.* Boca Raton: CRC/Lewis Publishers, pp. 401-432.

Frank, A., 2004. Six years of CO2 flux measurements for moderately grazed mixed-grass prairie. *Environmental Management,* Volume 33, pp. S426-S432.

Fry, J. et al., 2011. Completion of the 2006 National Land Cover Database for the Conterminous United States. *PE&RS,* 77(9), pp. 858-864.

Fuglie, K. & Kascak, C., 2001. Adoption and Diffusion of Natural-Resource-Conserving Agricultural Technology. *Review of Agricultural Econmics,* Volume 23, pp. 386-403.
GAO, 2006. *EMINENT DOMAIN: Information about Its Uses and Effect on Property Owners and Communities is Limited,* Washington, DC: Government Accountability Office.
GAO, 2007. *Farm Program Payments Are an Important Factor in Landowners' Decisions to Convert Grassland to Cropland,* Washington, DC: Government Accountability Office.
Gershenson, A., Barsimantov, J. & Mulvaney, D., 2011. *Background Paper on Greenhouse Gas Assessment Boundaries and Leakage for the Cropland Management Project Protocol,* Los Angeles: Climate Action Reserve.
Gibbs, H., Sheehan, J. & Nelson, R., 2010. *Land cover types subgroup final report,* Sacramento: California Air Resources Board.
Gleason, R., Tangen, B. & Laubhan, M., 2008. *Carbon Sequestration In Ecosystem Services Derived from Wetland Conservation Practices in the United State Prairie Pothole Region with an Emphasis on the U.S. Department of Agriculture Conservation Reserve and Wetlands Reserve Programs,* Reston: US Geological Survey.
Goddard, M. A., Mikhailova, E. A., Post, C. J. & Schlautman, M. A., 2007. Atmospheric Mg2+ wet deposition within the continental United States and implications for soil inorganic carbon sequestration. *Tellus B,* 59(1), pp. 50-56.
Goodwin, B. K. & Mishra, A. K., 2006. Are "Decoupled" Farm Program Payments Really Decoupled? An Empirical Evaluation. *American Journal of Agricultural Economics,* 88(1), pp. 73-89.
Grahame, J. & Sisk, T., 2002. *Canyons, cultures and environmental change: An introduction to the land-use history of the Colorado Plateau.* Flagstaff: Northern Arizona University.
Guo, L. B. & Gifford, R. M., 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology,* 8(4), pp. 345-360.
Guo, Y., Amundson, R., Gong, P. & Yu, Q., 2006. Quantity and Spatial Variability of Soil Carbon in the Conterminous United States. *Soil Science Society of America Journal,* Volume 70, pp. 590-600.
Hatfield, R. et al., 2012. *Conserving Bumble Bees: Guidelines for Creating and Managing Habitat for Americas Declining Pollinators.,* Portland: The Xerces Society for Invertebrate Conservation.
Hellerstein, D. & Malcolm, S., 2011. *The Influence of Rising Commodity Prices on the Conservation Reserve Program,* Washington, DC: USDA Economic Research Service.
Hertel, T. W., 2010. *The Global Supply and Demand for Agricultural Land in 2050: A Perfect Storm in the Making?.* Denver, Agricultural & Applied Economics Association, President's Address.
Hertel, T. W. et al., 2010. Effects of US Maize Ethanol on Global Land Use and Greenhouse Gas Emissions: Estimating Market-mediated Responses. *BioScience,* 60(3), pp. 223-231.
Hooker, T. et al., 2008. Distribution of ecosystem C and N within contrasting vegetation types in a semiarid rangeland, in the Great Basin, USA. *Biogeochemistry,* Volume 90, pp. 291-308.
Huang, H. & Khanna, M., 2010. *An Econometric Analysis of U.S. Crop Yield and Cropland Acreage: Implications for the Impact of Climate Change.* Denver, Agricultural & Applied Economics Association.
Ingram, L. et al., 2008. Grazing impacts on soil carbon and microbial communities in a mixed-grass ecosystem. *Soil Science Society of America Journal,* 72(4), pp. 939-948.
IPCC, 2003. *Good Practice Guidance for Land Use, Land-Use Change and Forestry,* Hayama: Institute for Global Environmental Strategies.
IPCC, 2006. Volume 4: Agriculture, Forest and Other Land Use. In: *Guidelines for National Greenhouse Gas Inventories.* Hageyama: Intergovernmental Panel on Climate Change.
Jantz, P. A. et al., 2007. Regulatory Protection and Conservation. In: M. R. Stromberg, J. D. Corbin & C. M. D'Antonio, eds. *California Grasslands: Ecology and Management.* s.l.:University of California Press, pp. 297-318.

Johnson, K. & Johnson, D., 1995. Methane emissions from cattle. *Journal of Animal Science,* Volume 73, pp. 2483-2492.

Jones-Farrand, D. T., Loren W. Burger, J., Johnson, D. H. & Ryan, M. R., 2007. *Grassland Establishment for Wildlife Conservation,* Bethesda: The Wildlife Society.

Jones, J., Wombacher, W., Sherrod, L. & McMillian, H., 2009. *Common Questions on Conservation Easements,* Fort Collins: Center for Collaborative Conservation, Colorado State University.

Kansas State University, 2004. *Kansas State University Range Page.* [Online]
Available at: http://spuds.agron.ksu.edu/
[Accessed 16 September 2012].

Kebreab, E. et al. eds., 2006. *Evaluation of models to predict methane emissions from enteric fermentation in North American dairy cattle.* Wallingford: CAB International.

Kebreab, E. et al., 2008. Model for estimating enteric methane emissions from US dairy and feedlot cattle. *Journal of Animal Science,* Volume 86, pp. 2738-2748.

Keeler-Wolf, T., Elam, D., Lewis, K. & Flint, S., 1998. *California Vernal Pool Assessment: Preliminary Report,* Sacramento: California Department of Fish and Game.

Keeler-Wolf, T. et al., 2007. Community classification and nomenclature. In: M. Stromberg, J. Corbin & C. D'Antonio, eds. *California grasslands: ecology and management.* Berkeley: University of California Press, pp. 21-34.

Kevan, P., 1999. Pollinators as bioindicators of the state of the environment: species, activity and diversity. *Agriculture, Ecosystems and Environment,* Volume 74, pp. 373-393.

Kohn, R. & Kim, S., 2011. A mechanistic model of enteric methane emissions from ruminants. *Journal of Animal Science,* 89(E-Supplement 1), p. 261.

Krausman, P. R. et al., 2011. An Assessment of Rangeland Activities on Wildlife Populations and Habitats. In: D. D. Briske, ed. *Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps.* Lawrence: USDA Natural Resources Conservation Service, pp. 253-290.

Kucharik, C., 2007. Impact of prairie age and soil order on carbon and nitrogen sequestration. *Soil Science Society of America Journal,* Volume 71, pp. 430-441.

Lal, R., 2001. Soil erosion and carbon dynamics on grazing lands. In: R. F. Follett, J. M. Kimble & R. Lal., eds. *The potential of U.S. grazing lands to sequester carbon and mitigate the greenhouse effect.* Boca Raton: CRC/Lewis Publishers, p. 231–247 .

Lal, R., 2002. Soil carbon dynamics in cropland and rangeland. *Environmental Pollution,* Volume 116, pp. 353-362.

Le Roux, P. C. & McGeoch, M. A., 2008. Rapid range expansion and community reorganization in response to warming. *Global Change Biology,* Volume 14, pp. 2950-2962.

Liebig, M. et al., 2010. Grazing Management Contributions to Net Global Warming Potential: A Long-term Evaluation in the Northern Great Plains. *Journal of Environmental Quality,* Volume 39, pp. 799-809.

Liebig, M. et al., 2005. Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada. *Soil & Tillage Research,* Volume 83, pp. 25-52.

Lin, W. & Dismukes, R., 2007. Supply Response under Risk: Implications for Counter-Cyclical Payments' Production Impact. *Applied Economic Perspectives and Policy,* 29(1), pp. 64-86.

Lin, W., Westcott, P. C., Skinner, R. & Sanford, S., 2000. *Supply Response Under the 1996 Farm Act and Implications for the U.S. Field Crops Sector,* Washington, DC: USDA Economic Research Service.

Love, H. A. & Foster, W. E., 1990. Commodity Program Slippage Rates for Corn and Wheat. *Western Journal of Agricultural Economics,* 15(2), pp. 272-281.

Luzar, E. & Diagne, A., 1999. Participation in the next generation of agriculture conservation programs: the role of environmental attitudes. *Journal of Socio-Economics,* Volume 28, pp. 335-349.

Mangino, J., Peterson, K. & Jacobs, H., 2003. *Development of an Emissions Model to Estimate Methane from Enteric Fermentation in Cattle.* San Diego, US Environmental Protection Agency.
Manley, J., Schuman, G., Reeder, J. & Hart, R., 1995. Rangeland soil carbon and nitrogen responses to grazing. *Journal of Soil and Water Conservation,* Volume 50, pp. 294-298.
Mayeux, H., 2000. Foreword. In: R. Follett, J. Kimble & R. Lal, eds. *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect.* Boca Raton: CRC/Lewis Press.
McConnell, S. G. & Quinn, M. L., 1988. Soil productivity of four land use systems in southeastern Montana. *Soil Science Society of America Journal,* Volume 52, pp. 500-506.
McDonald, R. I. et al., 2009. Energy Sprawl or Energy Efficiency: Climate Policy Impacts on Natural Habitat for the United States of America. *PLoS ONE,* 4(8), pp. 1-11.
McLauchlan, K., Hobbie, S. & Post, W., 2006. Conversion From Agriculture to Grassland Builds Soil Organic Matter On Decadal Timescales. *Ecological Applications,* Volume 16, pp. 143-153.
Melillo, J. M. et al., 2009. Indirect Emissions from Biofuels: How Important. *Science,* Volume 326, pp. 1397-1399.
Mensah, F., Schoenau, J. & Malhi, S., 2003. Soil carbon changes in cultivated and excavated land converted to grasses in east-central Saskatchewan. *Biogeochemistry,* Volume 63, pp. 85-92.
Michigan State University College of Law, 2012. *Map of State Endangered Species Laws.* [Online] Available at: www.animallaw.info/articles/armpstateesa.htm
[Accessed 16 August 2012].
Mitchell, J., 2010. *Criteria and Indicators of Sustainable Rangeland Management,* Laramie: University of Wyoming Cooperative Extension Service.
Morgan, D. E. & Calhoun, A. J., 2012. *Maine Municipal Guide to Mapping and Conserving Vernal Pool Resources,* Orono: University of Maine, Sustainability Solutions Initiative.
Mortenson, M., Schuman, G. E. & Ingram, L. J., 2004. Carbon Sequestration in Rangelands Interseeded with Yellow-Flowering Alfalfa (Medicago sativa Spp. Falcata). *Environmental Management,* Volume 33, pp. S475-S481.
Multi-Resolution Land Characteristics Consortium, 2012. *National Land Cover Database 2006 (NLDC2006) Product Legend.* [Online]
Available at: http://www.mrlc.gov/nlcd06_leg.php
[Accessed 5 August 2012].
Murray, B. C. & Baker, J. S., 2010. *An Output-based Intensity Approach for Crediting Greenhouse Gas Mitigation in Agriculture: Explanation and Policy Implications,* Durham: Nicholas Insitute for Environmental Policy Solutions, Duke University.
Murray, B. C., Sohngen, B. & Ross, M. T., 2007. Economic consequences of consideration of permanence, leakage and additionality for soil carbon sequestration projects. *Climatic Change,* Volume 80, pp. 127-143.
Natural Resources Canada, 2011. *Mining Regulations.* [Online]
Available at: http://www.nrcan.gc.ca/minerals-metals/policy/legislation-regulations/3707
[Accessed 18 September 2012].
Neff, J. et al., 2009. Soil carbon storage responses to expanding pinyon-juniper populations in Southern Utah.. *Ecological Applications ,* Volume 19, pp. 1405-1416.
Neff, J. et al., 1994. Fluxes of nitrous oxide and methane from nitrogen amended soils in a Colorado alpine ecosystem. *Biogeochemistry,* Volume 27, pp. 23-33.
NRCS, 2004. HEL Soil Erodibility Index. In: *National Food Security Act Manual, Fourth Edition.* Washington(DC): s.n., p. 511.01.
NRCS, 2006. *Digital General Soil Map of U.S. released on 07/05/2006,* Fort Worth: USDA Natural Resources Conversion Service.

NRCS, 2009a. *Planning site and seedbed preparation for cropland conversion to native species: Plant Materials Technical Note No. 13,* Washington, DC: Natural Resources Conservation Service.
NRCS, 2009b. *Summary Report: 2007 National Resources Inventory,* Washington, DC: Natural Resources Conservation Service.
NRCS, 2010. *Grassland Bird Conservation through Farm Bill Programs.* [Online] Available at: ftp://ftp-fc.sc.egov.usda.gov/VT/Programs/WHIP/G-Bird%20brochure.pdf [Accessed 12 August 2012].
NRCS, 2011a. *Conservation Activity List Ranking Period One 2012.* [Online] Available at: http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1046211.pdf [Accessed 16 August 2012].
NRCS, 2011b. *Forage and Biomass Planting Guide,* Auburn: Natural Resources Conservation Service, Alabama.
NRCS, 2011c. *Minnesota Supplement for Practice Standard 512 - Forage and Biomass Planting.* [Online] Available at: http://www.mn.nrcs.usda.gov/programs/eqip/2012/supplements/Forage%20And%20Biomass%20Planting%20(Formerly%20Pasture%20and%20Hay%20Planting)%20512.pdf [Accessed 16 August 2012].
NRCS, 2011d. *National Range and Pasture Handbook.* [Online] Available at: http://www.glti.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/landuse/rangepasture/?cid=stelprdb1043084 [Accessed 15 August 2012].
NRCS, 2012a. *Distribution Maps of Dominant Soil Orders.* [Online] Available at: http://soils.usda.gov/technical/classification/orders [Accessed 16 August 2012].
NRCS, 2012b. *Easements.* [Online] Available at: http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/programs/easements [Accessed 15 August 2012].
NRCS, 2012c. *National Soil Survey Handbook, Part 622.* [Online] Available at: http://soils.usda.gov/technical/handbook/contents/part622.html [Accessed 15 October 2012].
NRCS, 2012d. *Requirements for Forage Biomass Planting (512): 2012 Environmental Quality Incentives Program (EQIP).* [Online] Available at: www.tn.nrcs.usda.gov/programs/eqip2012/Docs2/RequirementSheetforForageBiomassPlanting.pdf [Accessed 16 August 2012].
NRCS, 2012e. *Wildlife Habitat Incentive Program (WHIP).* [Online] Available at: http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/programs/financial/whip/?&cid=nrcs143_008423 [Accessed 5 August 2012].
Ogle, S., Breidt, F. & Paustian, K., 2005. Agricultural Management Impacts on Soil Organic Carbon Storage under Moist and Dry Climatic Conditions of Temperate and Tropical Regions. *Biogeochemistry,* 72(1), pp. 87-121.
Ogle, S. M., Breidt, F. J., Eve, M. D. & Paustian, K., 2003. Uncertainty in estimating land use and management impacts on soil organic carbon storage for US agricultural lands between 1982 and 1997. *Global Change Biology,* Volume 9, pp. 1521-1542.

Ogle, S. M., Breidt, F. J. & Paustian, K., 2006. Bias and variance in model results associated with spatial scaling of measurements for parameterization in regional assessments. *Global Change Biology,* Volume 12, pp. 516-523.

Paustian, K. et al., 1995. Assessment of the contributions of CRP lands to C sequestration. *Agronomy Abstracts,* Volume 87, p. 136.

Pineiro, G., Paruelo, J., Oesterheld, M. & Jobbagy, E., 2010. Pathways of grazing effects on soil organic carbon and nitrogen. *Rangeland Ecology and Management,* 63(1), pp. 109-119.

Pitesky, M., Stackhouse, K. & Mitloehner, F., 2009. Clearing the air: Livestock's contribution to climate change. *Advances in Agronomy,* Volume 103, pp. 1-40.

Plevin, R. J. et al., 2010. Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated. *Environmental Science & Technology,* 44(21), pp. 8015-8021.

Post, W. M. & Kwon, K. C., 2000. Soil carbon sequestration and land-use change: Processes and potential. *Global Change Biology,* Volume 6, pp. 317-328.

Prins, A. G., Stehfest, E., Overmars, K. & Ros, J., 2010. *Are models suitable for determining ILUC factors?,* Bilthoven: Netherlands Environmental Assessment Agency (PBL).

Prokopy, L., Floress, K., Klotthor-Weinkauf, D. & Baumgart-Getz, A., 2008. Determinants of agricultural best management practice adoption: Evidence from the literature. *Journal of Soil and Water Conservation,* 63(5), pp. 300-311.

Pyke, C. R. & Marty, J., 2005. Cattle Grazing Mediates Climate Change Impacts on Ephemeral Wetlands. *Conservation Biology,* 19(5), pp. 1619-1625.

Rashford, B. S., Latchininsky, A. V. & Ritten, J. P., 2012a. *An Economic Analysis of the Comprehensive Uses of Western Rangelands: A report prepared for USDA Animal and Plant Health Inspection Service,* s.l.: Unpublished.

Rashford, B. S., Walker, J. A. & Bastian, C. T., 2012b. Economics of Grassland Conversion to Cropland in the Prairie Pothole Region. *Conservation Biology,* 25(2), pp. 276-284.

Rau, B., Johnson, D., Blank, R. & Chambers, J., 2009. Soil carbon and nitrogen in a Great Basin pinyon-juniper woodland: Influence of vegetation, burning, and time. *Journal of Arid Environments,* Volume 73, pp. 472-479.

Rau, B. et al., 2011. Transition from Sagebrush Steppe to Annual Grass (Bromus tectorum): Influence on Belowground Carbon and Nitrogen. *Rangeland Ecology and Management,* 64(2), pp. 139-147.

Rinehart, L. & Sullivan, P., 2010. *Converting Cropland to Perrenial Grassland,* Butte: National Center for Appropriate Technology.

Risgaard, M.-L., Frederiksen, P. & Kaltoft, P., 2007. Socio-cultural processes behind the differential distribution of organic farming in Denmark: a case study. *Agriculture and Human Values,* 24(4), pp. 445-459.

Rissman, A. & Bustic, V., 2011. Land trust defense and enforcement of conserved areas. *Conservation Letters,* 4(1), pp. 31-37.

Ritter, M. E., 2012. *The Physical Environment: An Introduction to Physical Geography.* [Online] Available at: http://www.earthonlinemedia.com/ebooks/tpe_3e
[Accessed 16 August 2012].

Roberts, M. J. & Bucholtz, S., 2005. Slippage in the Conservation Reserve Program or Spurious Correlation? A Comment. *American Journal of Agricultural Economics,* 87(1), pp. 244-250.

Roberts, M. J. & Schlenker, W., 2010. *Identifying Supply and Demand Elasticities of Agricultural Commodities: Implications for the US Ethanol Mandate,* Cambridge: National Bureau of Economic Research.

Robles, M. D. & Burke, I. C., 1998. Soil organic matter recovery on Conservation Reserve Progarm fields in Southeastern Wyoming. *Soil Science Society of America Journal,* Volume 62, pp. 725-739.

Samson, F. & Knopf, F., 1994. Prairie conservation in North America. *Bioscience,* Volume 44, pp. 418-421.
Schimel, J., 1995. Plant transport and methane production as controls on methane flux from arctic wet meadow tundra. *Biogeochemistry,* Volume 28, pp. 183-200.
Schuman, G. E., Janzen, H. H. & Herric, J. E., 2002. Soil carbon dynamics and potential carbon sequestration by rangelands. *Environmental Pollution,* 116(3), pp. 391-396.
Schwarze, R., Niles, J. O. & Olander, J., 2002. Understanding and managing leakage in forest-based greenhouse-gas-mitigation projects. *Philosophical Transactions of the Royal Society of London,* 360(1797), pp. 1685-1703.
Searchinger, T. et al., 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land Use Change. *Science,* 319(5867), pp. 1238-1240.
Silver, W., Ryals, R. & Eviner, V., 2010. Soil Carbon Pools in California's Annual Grassland Ecosystems. *Rangeland Ecology & Management,* Volume 63, pp. 128-136.
Smith, E., 2008. The Growing Demand for Oil and Gas and the Potential Impact Upon Rural Land. *Texas Journal of Oil, Gas, and Energy Law,* 4(1), pp. 1-25.
Spivak, M., Mader, E., Vaughan, M. & N.H. Euliss, J., 2011. The Plight of the Bees. *Environmental Science and Technology,* 45(1), pp. 34-38.
Stephens, S. E. et al., 2008. Predicting Risk of Habitat Conversion in Native Temperate Grasslands. *Conservation Biology,* 22(5), pp. 1320-1330.
Stubbs, M., 2007. *Land Conversion in the Northern Plains,* Washington, DC: Congressional Research Service.
Stubbs, M., 2012. *Conservation Compliance and U.S. Farm Policy,* Washington, DC: Congressional Research Service.
Sundquist, E. et al., 2009. *Rapid assessment of U.S. forest and soil organic carbon storage and forest biomass carbon sequestration capacity,* Reston: U.S. Geological Survey Open-File Report 2009-1283.
Su, Y. Z., 2007. Soil carbon and nitrogen sequestration following the conversion of cropland to alfalfa forage land in northwest China. *Soil and Tillage Research,* 92(1-2), pp. 181-189.
Swinton, S. M., Babcock, B. A., James, L. K. & Bandaru, V., 2011. Higher US crop prices trigger little area expansion so marginal land for biofuel crops is limited. *Energy Policy,* Volume 39, pp. 5254-5258.
T-AGG, 2012. *Near-Term Options for Reducing Greenhouse Gas Emissions from Livestock Systems in the United States: Beef, Dairy, and Swine Production Systems,* Durham: Technical Working Group on Agricultural Greenhouse Gases.
Taheripour, F., 2006. *Economic Impacts of the Conservation Reserve Program: A General Equilibrium Framework.* Long Beach, American Agricultural Economics Association.
Tennessee State Farm Service Agency, 2007. *Definition and Clarification for Marginal Pasture Land.* [Online]
Available at: http://www.tn.nrcs.usda.gov/programs/CRP/docs/Marginal_Pasture_Def.pdf
[Accessed 15 August 2012].
The Conservation Registry, 2012. *National Conservation Easement Database.* [Online]
Available at: http://www.conservationeasement.us/
[Accessed 28 October 2012].
United Nations Food and Agriculture Organization, 2012. *What are grasslands and rangelands?.* [Online]
Available at: http://www.fao.org/agriculture/crops/core-themes/theme/spi/scpi-home/managing-ecosystems/management-of-grasslands-and-rangelands/grasslands_what/en/
[Accessed 5 August 2012].
University of Tennessee, 2011. *NRCS National Easement Assessment Project (NEAP).* [Online]
Available at: http://neap.tennessee.edu/
[Accessed 28 October 2012].

US EPA, 2009. *National Water Quality Inventory: Report to Congress, 2004 Reporting Cycle,* Washington, DC: US Environmental Protection Agency.
US EPA, 2010. *Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis,* Washington, DC: United States Environmental Protection Agency.
US EPA, 2012a. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990 – 2010,* Washington, DC: U.S. Environmental Protection Agency.
US FWS, 2011. *Annual Report of Lands Under Control of the U.S. Fish and Wildlife Service. As of September 30, 2011,* Washington, DC: US Fish and Wildlife Service.
US FWS, 2011b. *Habitat Conservation Plans Under the Endangered Species Act,* Arlington: US Fish and Wildlife Service.
USDA Agricultural Marketing Service, 2010. *Side-by-side: A comparison of the current NOP regulation, and proposed and final pasture rules.* [Online]
Available at:
http://www.ams.usda.gov/AMSv1.0/getfile?dDocName=STELPRDC5082667&acct=noprulemakin
g
[Accessed 15 September 2012].
USDA Economic Research Service, 2012. *Commodity Costs and Returns.* [Online]
Available at: http://www.ers.usda.gov/data-products/commodity-costs-and-returns.aspx
[Accessed 5 September 2012].
USDA Federal Crop Insurance Corporation, 2012. *Rainfall Index Plan: Pasture, Rangeland, Forage (PRF) Crop Provisions.* [Online]
Available at: http://www.rma.usda.gov/policies/pasturerangeforage/13riprf.pdf
[Accessed 15 August 2012].
USDA FSA, 2009. *Farmable Wetlands Program.* [Online]
Available at: http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=fwp
[Accessed 9 August 2012].
USDA FSA, 2011. *Conservation Reserve Program Sign-up 41 Environmental Benefits Index (EBI).* [Online]
Available at:
http://www.fsa.usda.gov/FSA/newsReleases?area=newsroom&subject=landing&topic=pfs&new
stype=prfactsheet&type=detail&item=pf_20110128_consv_en_crpebi.html
[Accessed 15 August 2012].
USDA FSA, 2012a. *Conservation Programs: Statistics.* [Online]
Available at: http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=rns-css
[Accessed 27 September 2012].
USDA FSA, 2012b. *Program Fact Sheets: Conservation Reserve Program - Highly Erodible Land Initiative (Continuous Sign-Up).* [Online]
Available at: http://www.fsa.usda.gov/Internet/FSA_File/crp-hel_initiative.pdf
[Accessed 27 September 2012].
USDA FSA, 2012c. *USDA Announces New Highly Erodible Cropland Initiative for Conservation Reserve Program.* [Online]
Available at:
http://www.fsa.usda.gov/FSA/newsReleases?area=newsroom&subject=landing&topic=ner&ne
wstype=newsrel&type=detail&item=nr_20120221_rel_0062.html
[Accessed 27 September 2012].
USDA, 2010. *Grassland Reserve Program: DM 9500-012.* [Online]
Available at: http://www.ocio.usda.gov/directives/doc/DM9500-012.htm
[Accessed 15 September 2012].

USFS and University of Arizona, 2002. *Pinyon-Juniper Woodlands.* [Online]
Available at: http://ag.arizona.edu/oals/watershed/highlands/pinyonjuniper/pjgeneral.html
[Accessed 17 September 2012].

USFS, 2004. *Northeast Forest Inventory & Analysis: Methodology.* [Online]
Available at: http://www.fs.fed.us/ne/fia/methodology/def_ip.htm
[Accessed 5 August 2012].

USFS, 2005. *Management of the Pinon-Juniper Rangelands.* [Online]
Available at: http://www.fs.fed.us/rangelands/ecology/pinonjuniper.shtml
[Accessed 15 September 2012].

USFS, 2012. *The Great Plains including the Black Hills.* [Online]
Available at: http://www.fs.fed.us/rm/grassland-shrubland-desert/research/ecoregions/great-plains/
[Accessed 15 August 2012].

Utah State University, 2007. *What is the Great Basin?.* [Online]
Available at: http://www.usu.edu/weeds/great_basin/greatb.html
[Accessed 5 August 2012].

Van den Pol-van Dasselaar, A., van Beusichem, M. L. & Oenema, O., 1999. Determinants of spatial variability of methane emissions from wet grasslands on peat soil. *Biogeochemistry,* 44(2), pp. 221-237.

vanDoren, J., 2004. *Conservation Easement Violations & Amendments Study- Detailed Analysis,* Washington, DC: Land Trust Alliance.

VCSA, 2012. *VCS Standard, Version 3.2,* Washington, DC: Verified Carbon Standard Association.

Villoria, N. B. & Hertel, T. W., 2011. Geography Matters: International Trade Patterns and the Indirect Land Use Effects of Biofuels. *American Journal of Agricultural Economics,* 93(4), pp. 919-935.

Vöhringer, F., Kuosmanen, T. & Dellink, R., 2004. A Proposal for the Attribution of Market Leakage to CDM Projects. *HWWA Discussion Paper,* Volume 262.

Vöhringer, F., Kuosmanen, T. & Dellink, R., 2006. How to attribute market leakage to CDM projects. *Climate Policy,* Volume 5, pp. 503-516.

Western Pennsylvana Conservancy, 2012. *Seasonal Pools Database.* [Online]
Available at: http://www.paconserve.org/194/seasonal-pools-database
[Accessed 15 August 2012].

Whalen, S., 2005. Biogeochemistry of Methane Exchange between Natural Wetlands and the Atmosphere. *Environmental Engineering Science,* Volume 22, pp. 73-94.

Wienhold, B., Hendrickson, J. & Karn, J., 2001. Pasture management influences soil properties in the Northern Great Plains. *Journal of Soil and Water Conservation,* Volume 56, pp. 27-31.

World Resources Institute and World Business Council for Sustainable Development, 2005. *The GHG Protocol for Project Accounting,* Washington, DC: World Resources Institute.

WRI, 2012. *Grassland extent and change.* [Online]
Available at: http://www.wri.org/publication/content/8269
[Accessed 15 September 2012].

Wu, J., 2000. Slippage Effects of the Conservation Reserve Program. *American Journal of Agricultural Economics,* 82(4), pp. 979-992.

Yavitt, J. B. & Lang, G. E., 1990. Methane production in contrasting wetland sites: response to organic-chemical components of peat and to sulfate reduction. *Geomicrobiology Journal,* 8(1), pp. 27-46.