Background Paper on Greenhouse Gas Assessment Boundaries and Leakage for the Cropland Management Project Protocol

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Table of Contents

1. Introduction

1.a. Overview

The Climate Action Reserve (Reserve) is developing a Cropland Management Project Protocol (CMPP), which, among other things, will specify offset calculation methods for a number of project activities. The CMPP project activities will reduce CO_2 emissions or enhance CO_2 removals from the atmosphere by increasing organic matter (carbon) in mineral soils. This occurs by minimizing soil disturbances and/or maximizing organic matter inputs. There are two distinct categories of project activities that may be covered by the CMPP: (1) those that affect the way cropland is managed, but do not ultimately affect the use of the land and (2) those that represent a significant change in land use. In the first category, changes in cropland management (e.g., using no-till, reducing summer fallow, planting winter cover crops, etc.) can impact disturbance and inputs and, as a result, cause GHG reductions. In the second category, converting cropland that is otherwise capable of producing food, fiber or fuel into a permanent non-tree vegetative cover, such as pasture or natural grasslands, can also increase soil carbon sequestration compared to a cultivated system. This occurs by eliminating soil disturbance, increasing permanent belowground biomass, and possibly increasing overall organic matter inputs from permanent vegetation.

The Reserve has commissioned this study to identify significant leakage risks that may occur as a result of these project activities, as well as inform delineation of the GHG Assessment Boundaries for each activity. Upon determining these leakage risks, the Reserve will determine whether or not they should be included in calculation procedures in the CMPP. To this end, EcoShift Consulting has examined published literature and identified the direction and magnitude of major secondary GHG emissions effects that may occur in cropland management projects. This will create the basis for developing GHG estimates for all GHG sources, sinks, and reservoirs (SSR) that may be included in the GHG Assessment Boundary of cropland management projects. The following section defines and explains these terms and this approach further. In Section 1 we discuss the overall SSRs associated with changes in cropland management. We then discuss the specific leakage risks in Section 2, focusing on the proposed project activities that would be incorporated in the CMPP. In Section 3 we discuss the details of

these leakage risks, as well as offer some suggestions on quantification methodologies for each of the main categories. Our general conclusions are presented in Section 4.

1.b. GHG Assessment Boundary and Leakage

Determining an accurate and sufficiently broad GHG Assessment Boundary for a project is critical for ensuring complete and accurate accounting of a project's GHG impacts, especially when there are "leakage" risks. The IPCC refers to leakage as "the unanticipated decrease or increase in GHG benefit outside the project's accounting boundary as a result of project activities" (IPCC, 2000). Leakage occurs "whenever the spatial scale of the intervention is inferior to the full scale of the targeted problem" (Wunder, 2008). The Reserve uses the term leakage to refer to unintended increases in GHG emissions that may result from a GHG reduction project. The Reserve requires that all significant changes in SSRs associated with project activities be included in the GHG calculation, regardless of their physical location. Relevant text from the Reserve Program Manual explaining criteria and rationale used to delineate GHG Assessment Boundaries is provided in Box 1. These principles guide the consideration of leakage risks in the context of the CMPP as developed in this paper.

Box 1: Reserve Program Manual on Defining GHG Assessment Boundaries and Leakage

The GHG Assessment Boundary delineates the GHG SSRs that must be assessed in order to determine the total net change in GHG emissions caused by a GHG reduction project. GHG Assessment Boundaries are defined for each type of project activity addressed in a Reserve protocol.

The GHG Assessment Boundary is not a boundary related to a project's physical location. Instead, it encompasses all SSRs that could be significantly affected by a project activity, regardless of where such SSRs are located or who owns or controls them. A comprehensive and clearly defined GHG Assessment Boundary is required in order to provide a complete accounting of the net GHG reductions achieved by a project. All SSRs within the GHG Assessment Boundary are included in the calculation of GHG reductions.

SSRs are only included in the GHG Assessment Boundary if a project activity will have a significant effect on their associated GHG emissions or removals. The Reserve determines significance based on an assessment of the range of possible outcomes for a relevant SSR. There is no numerical threshold for significance. Inclusion or exclusion of SSRs is determined for each protocol based on the principles of completeness, accuracy, and conservativeness, and the need for practicality (e.g., related to measurement and monitoring costs). In general, relevant SSRs will only be excluded from the GHG Assessment Boundary if:

1. Projects are likely to reduce GHG emissions (or increase removals) at a SSR, so that excluding the SSR would be conservative (i.e., doing so would result in an underestimation of total net GHG reductions for the project); or
2. The total increase in GHG emissions from all excluded SSRs is likely to be less than five percent of the total GHG reductions achieved by a project.

Physical Project Boundaries

For some types of projects, it is necessary to define a physical boundary for a project in addition to a GHG Assessment Boundary. Physical boundaries are defined in terms of the physical area affected by a project activity and possibly specific equipment or facilities involved. Protocols will only require identification of a physical boundary where a physical boundary is necessary to quantify the magnitude of GHG emissions, removals or storage associated with one or more SSRs included in the GHG Assessment Boundary. The primary example would be forest projects, where the amount of carbon stored by a project depends on the area of land on which the project activity takes place.

Leakage Accounting

The term "leakage" is often used to refer to unintended increases in GHG emissions that may result from a GHG reduction project. Generally, leakage occurs at SSRs that are physically distant from the project itself or otherwise outside the project's physical boundaries. Because the Reserve requires the definition of a comprehensive GHG Assessment Boundary – which must include any and all SSRs associated with significant GHG emissions, regardless of their physical location – Reserve protocols generally do not require an explicit and separate accounting for "leakage" effects. Instead, all effects of a GHG reduction project – both positive and negative – are accounted for without distinguishing one kind of effect from another. This does not mean that Reserve protocols neglect or ignore what other methodologies or protocols identify as "leakage."

Where helpful for conceptual understanding, Reserve protocols may organize SSRs according to whether they are associated with a project's "primary" or "secondary" effects. A project's primary effect is its intended effect on GHG emissions (i.e., intended GHG reductions). Secondary effects are unintended effects on GHG emissions, often associated with leakage.

Changes in agricultural practices can result in significant increases in on-site soil carbon pools, which is the primary effect intended by project activities in the CMPP. But such changes in practice may be offset by increases in on-site and off-site GHG emissions, such as increases in CO_2, CH_4, and/or N_2O from changes in land use, fertilizer application, and other crop management practices. Off-site changes in emissions include those resulting from changes in agricultural inputs (Lal, 2004) or those that occur when agricultural production is displaced to new sites where land use change causes GHG emissions (Melillo *et al.*, 2009).

Like forestry projects, quantifying GHG emission and removals in CMPP projects will likely require delineation of a physical project boundary, especially to quantify the impacts of project activities on changes in soil carbon stocks. However, several other potential SSRs related to crop production may be affected by project activities even though they are well outside the physical project boundary. Therefore, for the purposes of exploring leakage risks in this paper, we further delineate GHG emission source categories as on-site emissions that occur within the physical project boundary, and off-site emissions associated with sources outside the physical project boundary.

1.c. The Importance of Comprehensive GHG Accounting in Agriculture: Examples from the Research Literature

Changes in agricultural practices can result in substantial soil carbon savings, but can also be associated with changes in other GHG emissions that may offset such carbon gains. As an example, Robertson *et al.* (2000) compared changes in agricultural GHG emissions under multiple management scenarios in a Midwestern corn-wheat-soybean system over nine years. Although they found that techniques such as no-till and using cover crops decreased GHG emissions from cropping systems, the increases in other GHG fluxes affected the overall impact of these techniques (Table 1).

Ecosystem Management	CO_2				N_2O	CH_4	Net GWP
	Soil C	N fertilizer	Lime	Fuel			
Annual Crops (Corn-Soybean-Wheat Rotation)							
Conventional Tillage	0.0	-109.3	-93.1	-64.8	-210.4	16.2	-461.3
No Till	445.2	-109.3	-137.6	-48.6	-226.6	20.2	-56.7
Low Input with Legume Cover	161.9	-36.4	-76.9	-80.9	-242.8	20.2	-255.0
Organic with Legume Cover	117.4	0.0	0.0	-76.9	-226.6	20.2	-165.9
Perennial Crops							
Alfalfa	651.6	0.0	-323.8	-32.4	-238.8	24.3	80.9
Poplar	473.5	-20.2	0.0	-8.1	-40.5	20.2	424.9

Table 1. Overall GHG Storage and Release under Various Management Regimes (kg C/acre). Positive values reflect storage; negative values reflect emissions to the atmosphere.

These added emissions may come from changes in inputs or machinery use. Table 2, adapted from a study that synthesized information on the life cycle impacts of agriculture (West *et al.*, 2002a), gives a rough estimate of the relative contribution of each phase of the agricultural process to total GHG emissions, not including soil carbon, for corn, soybean, and wheat. This serves to summarize the SSRs we will discuss in this section. While these values can vary greatly, and not all of these categories are implemented on every field, this study gives an initial and relative look at which areas contribute to overall GHG emissions. Depending on which project activity in the CMPP is implemented, leakage risks may be important in any of these emissions sources. The larger emission sources in Table 2 may be cause for more concern since an increase may lead to a significant leakage; however, it is necessary to first detail the effect and magnitude of changes in these sources for each project activity (see Section 2). Conversely, smaller emissions sources may still be significant leakage risks, depending on how they are affected by a project activity. Therefore, the size of these categories is not an indication of significant leakage risk, those are discussed in each respective section below.

		Corn		Soybean		Wheat	
		kg C / acre	%*	kg C / acre	%*	kg C / acre	%*
Machinery	Moldboard Plow	10.83	100.0%	10.83	100.0%	10.83	100.0%
	Disk	7.06	100.0%	7.06	100.0%	7.06	100.0%
	Planting	2.75	100.0%	2.75	100.0%	2.75	100.0%
	Single Cultivation	1.85	100.0%	-	0.0%	-	0.0%
	Harvest (Combine)	6.67	100.0%	6.67	100.0%	6.67	100.0%
Inputs	Herbicide	6.18	93.0%	3.16	98.0%	1.44	68.0%
	Insecticide	3.00	24.0%	2.31	2.0%	1.99	6.0%
	N	42.34	93.0%	9.27	16.0%	29.50	93.0%
	P_2O_5	3.74	83.0%	3.97	14.0%	5.16	70.0%
	K_2O	3.60	71.0%	4.37	14.0%	4.31	10.0%
	$CaCO_3$	54.95	5.0%	57.84	4.0%	54.95	1.0%
	Seed Production	8.70	100.0%	8.21	100.0%	7.80	100.0%
	Irrigation Water	68.00	15.0%	80.75	5.0%	42.50	7.0%
	Total C Emissions	219.66		197.17		174.95	

* % of planted hectares treated in 1995

Note: Values for herbicide, insecticide, N, P_2O_5, K_2O, $CaCO_3$ include both production and application. Overall, for the average for all three crop types is 40.1 kg C / acre on inputs and 27.9 kg C / acre on machinery.

Table 2. Non-Soil Carbon GHG Emissions from Corn, Wheat, and Soybean Production. Note that, in this table, positive values indicate emissions. (Adapted from West and Marland, 2002)

As these two examples illustrate, the complexity of SSRs associated with cropland management and the associated leakage risks are considerable, and the Reserve needs an in-depth study of the GHG assessment boundary to support agriculture protocol development.

1.d. GHG SSRs Associated with Agriculture

In order to comprehensively analyze potential leakage risks, the first step is to consider the full range of GHG emissions categories related to agricultural production. This section introduces categories of primary and secondary effects. We define primary effects as the changes in soil carbon that occur as a result of any given project activity and secondary effects (including emissions of CO_2, N_2O, and CH_4) as those that occur on-site, upstream or downstream, and are a result of the project activity. Figure 1 illustrates the on-site and off-site SSRs that we will consider for the purposes of this analysis, in order to ensure that impacts of management changes are fully considered within the scope of the CMPP. The rest of this section describes in general each of the SSR categories illustrated n Figure 1.

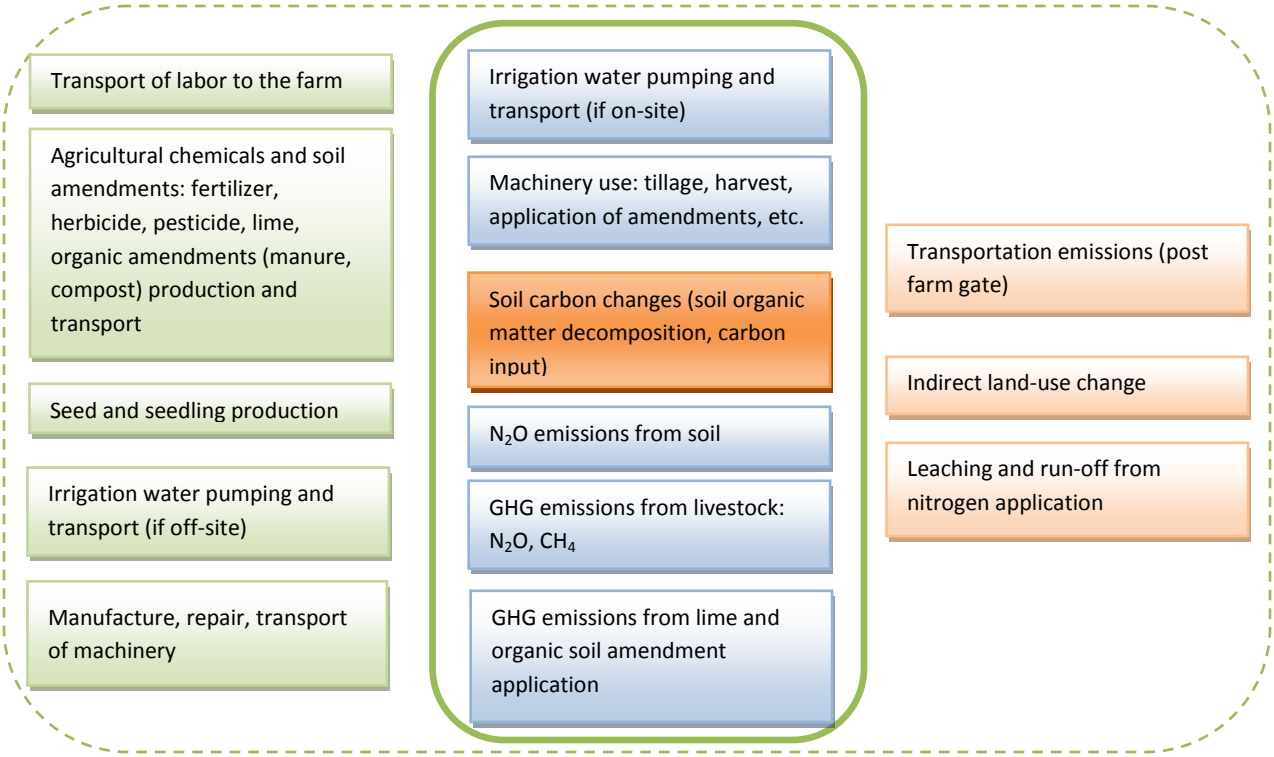

Figure 1. General Illustration of the Physical Project Boundary (Solid green) and Comprehensive GHG Assessment Boundary (dotted green). Light green boxes delineate upstream off-site SSRs, light blue boxes delineate on-site SSRs, and light orange boxes delineate downstream off-site SSRs. The dark orange box delineates primary effects of project activities.

1.d.(i). Primary Effects: Soil Carbon Dynamics

The primary effects of proposed CMPP project activities will be on soil carbon. Soil carbon exists in several distinct pools or fractions. The shortest-lived carbon (<1 year old) is represented by microbial biomass and labile root exudates, which are easily decomposable simple organic compounds released by roots into soil. Medium-term carbon, several years to decades old, is represented by more complex organic materials. Ancient carbon, hundreds-thousands of years old, is represented by humins and humic acids, among other materials (Schlesinger, 1997). There are two main processes by which carbon enters the soil pool, and one primary process for soil carbon loss. Carbon can either enter the soil through litter fall, where it is incorporated into the surface organic or mineral soil horizons, or it can enter the soil pool through rhizosphere processes, which include fine root death and root exudation. Any management activities that increase plant productivity have the potential to sequester carbon, often at long-term time scales

(Paustian *et al.*, 2000). Carbon is lost from soil through microbial decomposition, which is largely dependent on temperature, moisture, and substrate availability (Gershenson *et al.*, 2009).

A major effect of traditional farming techniques on GHG emissions is the change in soil carbon dynamics associated with soil disturbance, primarily through tillage (Lal, 2004). Disturbances associated with tillage and subsequent changes in soil moisture and hydrology of agricultural lands have been found to reduce original soil carbon content by between 20 and 42%, both through enhanced decomposition of organic matter due to physical disturbance, and through reduction of carbon inputs due to removal of biomass during harvest (Post *et al.*, 2000, Houghton *et al.*, 2004). Although agricultural practices, due to associated disturbance, have resulted in soil carbon losses of 8-32 t C/acre, through the use of some management practices it is possible to increase soil carbon stocks by as much as 400 kg C/acre per year, until a new equilibrium of soil carbon content is established, which will differ for different soils (Lal, 2004). Reduced tillage techniques are commonly suggested as the main management practices to reduce soil carbon losses; however, multiple recent papers suggest that the specific effects of no-till or conservation-till adoption are greatly site-dependent (Govaerts *et al.*, 2009). The intended primary effect of the CMPP is to avoid or reverse soil carbon losses by changing tillage management and other disturbance regimes, as well as by altering cropping dynamics, inputs of carbon into the soil, and overall management strategies.

Management techniques that reduce soil disturbance (e.g., reduced tillage intensities) (West *et al.*, 2002b) or change temperature and moisture dynamics to discourage carbon mineralization (Paustian *et al.*, 2000) act to slow down the rates of microbial decomposition of soil carbon, and therefore ensure that more carbon stays in the soil, increasing the overall carbon sequestered. Additionally, any techniques that increase on-site or off-site carbon inputs into soils, such as incorporation of cover crops or organic amendments (Ding *et al.*, 2006), have the potential to increase soil carbon content. Several proposed project activities have the potential to either decrease soil disturbance or increase carbon inputs. Changes in tillage management, increases in organic soil amendments and the use of cover cropping, as well as conversion of croplands to pasture and some form of perennial systems can reduce the rates of soil carbon decomposition, and increase carbon inputs into the soil. Other management practices, such as reductions in

fallow periods, can increase soil carbon inputs through increased productivity, although they may increase the rates of soil carbon decomposition due to increased disturbance. The potential effects of different management techniques on soil carbon are extensively covered in Eagle *et al.* (2010); in this paper, we discuss some of these, but focus primarily on secondary effects (leakage risks) of different proposed project activities.

It is important to note that historically recorded soil carbon losses due to farming practice are non-linear, with recently converted lands experiencing high carbon loss rates initially, and fields with a longer agricultural history exhibiting much smaller amounts of carbon losses on an annual basis (Figure 2). Utilizing carbon-enhancing management practices on recently converted lands may therefore avoid greater carbon losses from mineralization than similar practices taking place on older agricultural fields. However, it has been suggested that older (>20 years) fields are more prone to carbon losses due to erosion, although it is unclear how much of that carbon ends up in the atmosphere versus as permanently buried sediment (Lal, 2004).

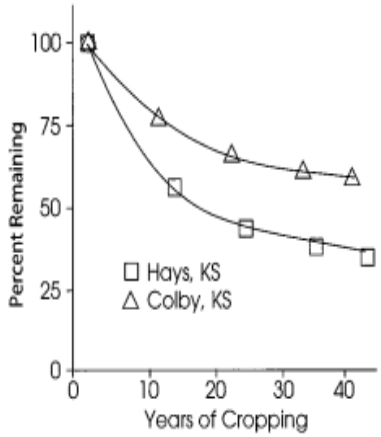

Figure 2. Soil Organic Matter Loss Following Cultivation at Two Sites in the Midwestern U.S. Adapted from Robertson and Grace, 2004.

Any increases in soil carbon content also have important co-benefits. Soils with high organic matter content have several important characteristics, such as increased nutrient availability, increased water retention, and increased aggregate stability, which increase productivity and crop yield (Mäder *et al.*, 2002). They also reduce machinery use requirements due to better tilth characteristics (Favoino *et al.*, 2008). These changes also have the potential of increasing yield. Therefore, the CMPP should provide benefits to the environment beyond carbon storage.

Potentially negative impacts of project activities on other aspects of the environment are discussed in the text when these arise.

1.d.(ii). Secondary Effects

In addition to primary effects of proposed project activities, there are multiple secondary sources of emissions that must be considered when we examine the overall GHG footprint of agricultural production.

N_2O Emissions from Soils: Nitrous oxide (N_2O) is a highly potent GHG that is emitted under certain conditions from agricultural systems, which can be a significant component of overall GHG emissions (Ginting *et al.*, 2003). N_2O is an important GHG because it has 310 times the global warming potential of CO_2 on a per-molecule basis (Solomon *et al.*, 2007). Nitrogenous fertilizer application and cropping practices are estimated to cause 78% of anthropogenic N_2O emissions (U.S. EPA, 2007). A main source of on-site emissions from agricultural chemicals and soil amendments is the biogenic breakdown and/or volatilization of nitrogen-containing soil amendments in the field, leading to N_2O emissions. There are two categories of N_2O from soils (according to IPCC terminology for these sources): direct N_2O emissions that occur on fields where N is applied and indirect emissions that occur elsewhere as a result of volatilization and leaching/run-off. Figure 1 shows these categories separately with direct emissions occurring on-site (N_2O emissions from soil) and a separate category for off-site N_2O emissions occurring outside the project boundary (leaching and run-off from nitrogen application).

Emissions of N_2O depend on several factors, such as input of organic materials and changes in soil moisture dynamics. For instance, Wagner-Riddle *et al.* (1997) have observed that there is a substantial N_2O flux in the spring from winter-fallow plots (0.88 to 2.64 kg N/acre per year (116.8 to 350.4 kg C/acre per year)), due to higher soil moisture content and high availability of organic matter from the previous season. They also observed a significant N_2O flux following incorporation of alfalfa (2.4 kg N/acre per year (318.6 kg C/acre)), which was grown as a cover crop, as well as following application of manure to fallow plots (2.28 to 2.96 kg N/acre per year (302.6 to 392.9 kg C/acre per year)).

The biggest difficulty with estimating the effect of management on N_2O emissions is the tremendous variability in potential emissions due to site-specific micrometeorological factors, types of organic matter or fertilizer that are applied, as well as the mechanism and the timing of application (Smith *et al.,* 1997). To illustrate, a study of different soil/crop combinations in England showed that the rates of N_2O emissions varied more than 16-fold, ranging between 24 and 945 kg C/acre, depending on soil water content, type of crop or grassland, soil temperature, and nitrate content (Dobbie *et al.,* 2003). The authors also found that managed grasslands can exhibit very high rates of N_2O emissions, similar in magnitude to N_2O emissions from cropland. In fact, one of the main predictors of N_2O emissions is soil organic matter content, and since proposed project activities all aim to increase soil carbon content, and therefore soil organic matter, there is a substantial risk of increased N_2O emissions significantly reducing, and in some cases completely erasing, the gains in increased soil carbon sequestration that result from actions such as reduced tillage, enhanced crop residue incorporation, and manure application (Li *et al.*, 2005).

In general, N_2O emissions need to be considered whenever nitrogen-rich content, whether synthetic fertilizer, organic amendment or cover crop-based, is applied to soils with high moisture content and inadequate drainage, although the amounts are highly variable and depend on multiple factors. N_2O emissions are likely to be a factor in almost all proposed project activities. Project activities that involve increased rates of organic matter addition, such as cover cropping, organic amendments, especially manure, and conversion to pasture, significantly increase the amount of nitrogen-rich organic matter and have the potential to stimulate N_2O production. Additionally, practices such as reduced tillage and incorporation of perennial agriculture have the potential of altering soil moisture dynamics, thereby affecting N_2O emissions. Because N_2O emissions are highly variable, modeling tools, such as the DAYCENT model (Del Grosso *et al.,* 2006) and the IPCC methodology for estimating N_2O emissions, are necessary to calculate actual emissions. However, process-based models such as DAYCENT may be more applicable since the IPCC methodology is designed for estimates on larger geographic scales. Compared with direct on-the-ground measurements, the data requirements for such models are fairly minimal, and calculations of N_2O fluxes are relatively robust.

Machinery Use: Agriculture relies on fossil fuel-powered machinery during many stages of the production cycle. Machinery used in agriculture creates GHG emissions through the burning of fossil fuel as well as through the manufacture, transport, and repair of this equipment. Specific stages of the agricultural process in which machinery is used include tillage, planting, fertilizer and pesticide application, harvesting, and potentially drying. Fuels used to power agricultural machinery include diesel, gasoline, natural gas, electricity, and propane. GHG emissions associated with on-farm machinery constitute a significant (>5%) portion of total agricultural GHGs (Table 1). Because many project activities will create changes in these emissions, it will be necessary to account for changes in GHG resulting from machinery use in the CMPP.

Since nearly all farm activities involve the use of machinery, many project activities have the potential to change machinery use patterns. This will result in either an increase or decrease in GHGs from machinery depending on the activity. For example, eliminating summer fallow may increase carbon storage in soil, but the GHGs associated with machinery use in planting the cover crop should be included in the GHG accounting boundary to prevent leakage. On the other hand, switching from conventional tillage to no-till to increase carbon storage may achieve additional GHG savings by reducing the emissions from machinery used in tilling.

One source of emissions associated with machinery that may be necessary to consider is emissions that occur during manufacturing, transport and repair of machinery. These are highly varied and equipment specific; however, West *et al.* (2002a) suggest that this category of emissions is responsible for 16% of total primary energy associated with on-farm machinery use. We suggest a potential mechanism to include these emissions, if the Reserve decides to do so, in Section 3.

Transportation: Transportation occurs in all phases of agricultural production, including upstream, on-site, and downstream. Upstream transportation emissions refer to those associated with moving inputs to the farm. For agricultural chemicals in particular, these embedded emissions are often included together with production related emissions estimates so we will not continue to discuss these upstream emissions separately as transportation emissions. One exception is transportation of labor to the farm, which has not received systematic treatment in

the literature. In Section 3 we discuss the potential and limitations of including these emissions. On-site transportation emissions are covered under the category of on-site machinery use.

Transportation emissions in this report relate to downstream transportation emissions from the movement of agricultural products to market. This is a highly variable category of emissions (dependent on mode and distance) and also very difficult to track. These emissions could be considered from the farm gate to distribution center, to final point of sale or even to point of consumption. Clearly, the broader the boundary, the more difficult the computation will be. In some cases, project activities will affect the volume or weight of agricultural product, which can affect GHG emissions of vehicles needed to transport product. This would possibly occur, for example, in a removal of summer fallow, converting from annual to perennial cropping systems or converting cropland to pasture. In some cases, it may be important to estimate emissions from downstream transportation, and we discuss issues related to this in Section 3.

Agricultural Chemicals and Soil Amendments: The manufacture, transportation, and use of fossil fuel-based fertilizers and pesticides are associated with significant contributions to overall GHG emissions associated with agriculture (Table 2). Fertilizer production alone is believed to constitute approximately 1.2% of global energy demand (Wood *et al.,* 2004). These inputs helped drive agricultural intensification in the 20th century, which some have argued has led to important avoided GHG emissions from land use change (Burney *et al.,* 2010). Off-site emissions from producing and transporting agricultural chemicals and soil amendments also pose potential leakage risks, which are covered in detail in Section 3. These include emissions from fertilizer, lime, and pesticide production, which are a significant portion of agricultural emissions (Table 2). The relationship between energy use in agricultural chemical manufacturing and GHG emissions will also depend on the carbon intensities of the power mix where they are produced.

Biogenic fertilizers such as manure have lower energy intensities in production, up to three orders of magnitude lower (Lal, 2004) and, in some cases, manures sequester carbon when applied on-site (Smith *et al.,* 2000). Evidence suggests that there are potential GHG benefits associated with composting manure prior to application. By stabilizing organic matter, compost appears to reduce post-application respiration losses and reduces N_2O emissions. In the cases

where parent material is not composted, there may be issues of additional emissions during the application process. In fact, although decomposition of manure in a landfill or a slurry pond would probably produce greater CH_4 emissions through more anaerobic decomposition, application to fields is likely to produce higher N_2O emissions within the physical boundary of the project. In the case of manure, the relationship between baseline fate and alternative fate is complicated, but Mosier *et al.* (1998) estimate that N_2O emissions from manure in a grazing system are 20 times higher than anaerobic lagoons and liquid systems, and is approximately equal to dry lot storage. Although anaerobic and liquid systems are likely to produce more CH_4, since N_2O is a much more potent greenhouse gas, these increased emissions will overwhelm the reductions in CH_4 production when converted to CO_2e. As such, it is critical that leakage risks from additional N_2O emissions are considered, either using the IPCC methodology or a model such as DAYCENT. Although complex, this is not a trivial matter given the potential magnitude of emissions under various alternate fates of feedstock.

Irrigation (off-site and on-site): Irrigation is a widely used practice that enhances yield and increases carbon input into soils due to increased productivity (Follett, 2001, Lal, 2004). Irrigation in dry regions also has an important effect on soil carbonate chemistry since it can stimulate precipitation of soil carbonates, further removing carbon dioxide from the atmosphere (Jarecki *et al.*, 2003). Depending on the ecosystem and the farming methods, irrigation has the potential of increasing soil carbon stocks by 20-80 kg C/acre/year (Jarecki *et al.*, 2003). However, the gains in increased soil carbon content can be offset by several factors, such as increased energy requirements for water delivery (Schlesinger, 1999, Mosier *et al.*, 2005), as well as changes in N_2O emissions due to higher soil moisture content (Liebig *et al.*, 2005) (covered in N_2O emissions section above). Because irrigation is an energy intensive management strategy, whether it is an on-site use of energy for pumping and delivery or energy embodied in delivered water from off-site, changes in irrigation practices can affect overall GHG emissions. This is treated in greater detail in Section 3.

Seed and Seedling Production: In a similar manner to the other agricultural inputs mentioned above, the production of seeds and seedlings also creates emissions of GHGs. The production of seeds and seedlings requires energy for growing, as well as packaging and transporting the seeds

and seedlings to the project area. Specific GHG impacts are discussed in greater detail in Section 3.

Indirect Land Use Change (ILUC): A somewhat unique type of leakage is ILUC, which is a category that is difficult to estimate, and is more of a function of economics than the other categories discussed above. ILUC is caused when a project activity indirectly affects the crop management on other lands to make up for decreased food, fuel or fiber supply. The most GHG intensive ILUC occurs when natural systems such as peat lands, forests or grasslands are converted into agricultural systems to make up for lost supplies (Searchinger *et al.,* 2008). The IPCC has developed a methodology for estimating emission factors from land conversion, which are used in models by the U.S. EPA in the Renewable Fuel Standard (RFS) and the California Air Resources Board Low Carbon Fuel Standard (LCFS) calculations. Most estimates of ILUC utilize economic models to estimate the extent to which decreased supply is made up off-site, resulting in a highly uncertain range of values.

Several project activities may either increase yield (irrigation or removal of summer fallow) or decrease yield (conversion to pasture or conversion to non-cultivated). Any change in yield will also have an ILUC effect, either positive or negative. For example, increased yield may have positive ILUC carbon implications by reducing the need to produce elsewhere, while reducing yield can displace agriculture onto other land types and lead to significant carbon losses (Guo *et al.,* 2002).

GHG Emissions from Livestock: According to the U.S. EPA GHG Inventory, GHG emissions from livestock in 2009 were primarily from enteric fermentation (139.8 Tg CO_2e) and manure management (49.5 Tg CO_2e) (EPA, 2011b). Cropland Conversion to Pastureland, a project activity analyzed in this paper, can increase the number of livestock and have important effects on fluxes of CH_4 and N_2O from agricultural ecosystems. The estimates of emissions are highly variable, since the GHG impact of grazers depends on the type of animals, as well as stocking density, housing practices, manure management practices, and even the type of feed. Some studies of emissions from livestock estimate lower enteric fermentation GHG emissions for cattle

fed distillers grain, a co-product of ethanol (Liska *et al.*, 2009). We discuss the special case of conversion to pasture in Sections 2 and 3.

1.e. Considerations Specific to Emissions Capped Under GHG Regulations

Several of the proposed project activities that we discuss below are likely to increase the use of electricity or fuel, such as increases in irrigation or machinery use. However, there may be some cases in which such project leakage does not produce a net increase in emissions. This would occur in situations where leaked emissions occur in a jurisdiction in which environmental regulation exists that creates a hard cap on GHG emissions.

For example, consider a project activity such as switching from dry land farming to irrigation, which increases off-site electricity usage by requiring additional pumping of irrigation water. In addition, for this example, the electricity generation occurs in a state that has a hard cap on GHG emissions from the energy sector and a GHG permit trading program. In this example, the increased electricity usage would not create a net increase in carbon emissions. Instead, it would create additional demand for electricity (and permits), causing the price of permits to increase (ever so slightly). A complimentary reduction would occur elsewhere in the system for a user whose marginal cost of GHG reduction is lower than the ensuing permit price, since there is a hard cap on total emissions.

On the other hand, a net increase in GHG emissions would occur in this scenario if the state had environmental regulation that did not have a hard cap. This would occur under such policy tools as a renewable portfolio standard or a carbon tax. Although the price of the additional electricity production would be higher due to the regulation, if it were profitable to pay the additional cost, a net increase in GHG emission would still occur.

In all cases, the carbon intensity of purchased or generated electricity would need to be considered, as renewable energy used for additional pumping would not create carbon leakage. In accounting for leakage risk that results from changes in fuel or electricity use, the Reserve should consider the regulatory environment of the project location in order to avoid potential dual penalty for the project developer, in the cases of regulatory caps on these categories.

2. Project Activities and Significance of Secondary Effects: Recommendations for Delineating the GHG Accounting Boundary and Estimates of Leakage Risks

After developing a complete list of potentially important GHG categories, the second step in developing a GHG assessment boundary is to evaluate how "significant" each of the possible GHG categories will be in terms of how they are potentially affected by a project. As noted in the introduction, the CMPP could encompass a very broad range of project activities, with each having a different potential impact on the various GHG categories outlined in Section 1. Therefore, Section 2 discusses and summarizes the potential significance of leakage in each of the GHG categories separately by CMPP project activity.

Eagle *et al.* (2010) describe the carbon storage potential (or primary effect) from possible project activities (Table 3) and assess in a general sense other on-site and off-site secondary effect emissions in terms of broad categories, specifically "land emissions" and "upstream and process emissions." Our goal is to identify the specific secondary SSRs that create leakage risk for each particular project activity and thus may be considered for inclusion in the GHG Assessment Boundary, and estimate their direction and magnitude. This analysis will enable the Reserve to determine whether there is significant leakage for each SSR. This section indentifies leakage risks for each project activity, and in Section 3 we give more detailed information as to how and why each SSR may contribute to leakage, and provide information on how leakage effects could be quantified.

Project Activity	Primary Effect (Soil Carbon: kg C/acre)		Max U.S. Area (million acres)
	Min	Max	
Conventional to Cons. Till	0.0	200.8	29
Conventional to No Till	-28.7	286.8	29
Eliminate Summer Fallow	-97.1	259.2	8
Use Winter Cover Crops	40.8	357.4	30
Diversify Annual Crop Rotations	-275.8	332.0	40
Perennial Crops in Rotation	-193.1	242.7	23
Annual to Perennial Crop	0.0	515.2	5
Organic Soil Amendments	19.9	562.6	4
Dry Land to Irrigated	125.8	526.2	NA
Irrigation Improvements	19.9	64.0	8
Cropland to Pasture	0.0	518.5	Unknown
Cropland to Uncultivated Use	-16.5	522.9	6

Table 3. Average GHG Impacts and Applicable Area in the U.S. (Eagle *et al.* 2010)

2.a. Project Activities Representing a Management Change in Existing Croplands

2.a.(i). Change from Conventional to Conservation and No-Till Systems

Estimates suggest that implementing reduced tillage in the U.S. alone has the potential to sequester 19.0 to 143.4 Tg C/y under the most aggressive climate policy (Thomson *et al.*, 2006), which would, at its maximum, serve to offset all of the methane emissions from enteric fermentation in cattle in the U.S. (EPA, 2011b). As an example, it has been estimated that switching from conventional tillage practices could reduce GHG emissions from machinery use in Canadian farms by roughly 43% for reduced till and 58% for no-till (Dyer *et al.*, 2003). However, in certain systems no-till management does not appear to have any soil carbon benefits at all (wheat-fallow systems, West *et al.*, 2002b).

In addition to these mostly positive primary effects, there are many secondary effects of this practice, including N_2O emission, use of herbicides, fertilizer, and lime (Table 4). Generally, reduced tillage is also economically beneficial, since expenditures associated with fertilizer and machinery use decrease, and net income can increase as much as 25% (Archer *et al.*, 2002). On the other hand, reductions in tillage intensity also have the potential to increase herbicide

application and emissions associated with their manufacture, transportation, and application rates. Eagle *et al.* (2010) suggest that additional emissions from such production are negligible, but economic analysis shows that incorporation of conservation tillage can result in increases of herbicide use by as much as 10% (Archer *et al.*, 2002), which may create negative environmental impacts which are additional to the increase in GHG emissions from the additional herbicide. No-till management also has the potential of significantly increasing N_2O release from agricultural systems due to increases in soil moisture as a result of reduced soil aeration (Venterea *et al.*, 2008), so accounting for such sources of leakage is of paramount importance. The main risk factor for such potential for increased N_2O emissions is in soils with high soil moisture content or improper drainage, and in these systems emissions are highly variable.

Additional major secondary changes relative to the baseline[1] include a potential for reduced fertilizer use and associated on-site and upstream emissions (Section 3) due to better nutrient availability as a result of reduction of upper soil disturbance. Even though fertilizer requirements may be reduced, economic pressures and risk aversion may push farmers to maintain fertilizer levels (Archer *et al.*, 2002), so this link would need to be verified on a case-by-case basis. In some cases, reduced tillage increases soil acidity, necessitating increased inputs of lime, which carry significant leakage risks from manufacture, transport, and application (Section 3). Finally, there are certain secondary effects that may contribute to a reduction in GHG emissions, including machinery use and irrigation, since tillage requires machinery and contributes to drier soils. The change in soil moisture may lead to additional N_2O emissions from no-till systems. For instance, Li *et al.* (2005) note that, when taking into account N_2O increases due to reduced tillage, the carbon sequestration gains are significantly diminished in a corn-soybean system, and overall emissions can increase upwards of 75% of the baseline. Therefore secondary effects are an extremely important consideration for this project activity.

An important uncertainty is the efficacy of no-till systems in sequestering soil carbon at all. Significant arguments have been raised that initial estimates of the benefits of no-till methods for

[1] The baseline is a reference point or scenario for GHG emissions/sequestration that would occur in the absence of a project. GHG reductions from offsets projects are assessed as the difference in project and baseline emissions/sequestration.

soil carbon sequestration are due to methodological mistakes, rather than actual benefits of no-till systems (Baker *et al.*, 2007). Although a controversial subject, a follow-up analysis of multiple studies shows that, while there is an increase in surface carbon following no-till adoption, there is a corresponding decrease in soil carbon in subsurface layers, possibly due to reduced transport of carbon into deeper soil profiles, and that overall the soil carbon gain following adoption of no-till or conservation till techniques is questionable at best (Luo *et al.*, 2010). In order to accurately evaluate the effects of no-till management on soil carbon stocks, it is critical to monitor soil carbon beyond the traditional 30cm depth, as it does not accurately capture soil carbon changes to look only at the first 30 cm.

In projects that incorporate no-till or conservation-till practices, our analysis suggests that the project developer should ensure that (1) the gains in soil carbon are considered within the whole soil profile, not just the surface 30cm, (2) the gains in soil carbon are adjusted by corresponding changes in N_2O emissions from the field, especially in soils with poor drainage, (3) any additional inputs of lime are taken into account, both in terms of impacts of production and application and (4) that the change in management does not result in an increase in fertilizer use.

Category	Potential Effect on GHG Emissions (kg C/acre)		Potentially Significant? (>5%)*	Comments
	Conservation Till	No Till		
Soil Carbon[a]	0 – 200.8	-28.7 – 286.8	NA	
Soil N_2O[b]	Highly variable, soil condition dependent		Yes	Emissions of N_2O can constitute a leakage risk, constituting up to 75% the soil carbon benefits of tillage changes. Highly variable
Machinery[c]	~ 10.8	17.9 – 19.7	No	Reduction in tillage results in decreased machinery use
Herbicides/ Pesticides[c]	-0.20 – 0.05	-1.13 – -0.22	No	Herbicide use increases slightly, insecticide use decreases slightly
Fertilizer[c]	-15.7 – 15.4	-15.8 – 6.6	Maybe	Fertilizer increases for corn and soy, decreases for winter wheat
Irrigation[c]	4.3 – 8.5	8.5 – 17.0	No	Effect due to changes in moisture
Lime Production and Application[d]	Variable, significantly smaller than no-till	-44.5 – 0	Maybe	Highly variable, depends on soil pH changes and rates of application
Indirect Land Use Change[e]	-24,500 – 0 Grassland -71,700 – 0 Forest -330,800 – 0 Peatland		Maybe	In worst-case scenario, new land is brought into agriculture from natural systems. The values would be modified by a net displacement factor that captures the numbers of acres brought into production

Table 4. Leakage Risks for Switch from Conventional to Conservation or No-Till Practices.

Note: Negative values indicate additional emission potential, positive values indicate carbon sequestration potential.

[a] Eagle 2010; [b] Li *et al.*, 2005, Six *et al.*, 2002; [c] West & Marland 2002; herbicide, pesticide and fertilizer figures include both production and application; [d] Robertson *et al.*, 2000; [e] Emissions factors converted from Plevin *et al.* 2010. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

2.a.(ii). Elimination of Summer Fallow

The use of summer fallow, defined as leaving a field unplanted for a period of several summer months between planting, is commonly implemented every several years for purposes of water conservation and weed and pest suppression. Summer fallow elimination can sequester soil C at rates of over 2.0 t CO_2e/ha yr, with an average of net GHG impact of 0.32 t CO_2e/ha yr (Eagle *et al.* 2010). The baseline scenario negatively affects soil carbon stocks through reduced carbon inputs due to lack of plant productivity, as well as increased carbon losses due to increased erosion and soil respiration. Elimination of summer fallow increases inputs of carbon into the soil through plant productivity and decreases soil lost to erosion. This practice is especially

effective when combined with a no-till or conservation-till management regime (Eagle *et al.*, 2010).

However, elimination of summer fallow and replacement with an additional crop involves several GHG leakage considerations. The main on-site leakage risk is associated with increased cropping operations as a result of putting fallow land back into production, such as increased agricultural inputs, irrigation, and the use of machinery for harvest and subsequent transport (Section 3). Depending on the geographic location of the project, as well as soil moisture content and management practices, the additional benefits of eliminating summer fallow may be outweighed by these sources of leakage (Eagle *et al.*, 2010). Location is important because weather-related impacts could have positive leakage impacts by delaying decomposition in the cooler northern plains or retaining greater soil carbon in crop residues due to reduced water supply in the Great Plains (Eagle *et al.* 2010, p. 10). And finally, there may be additional N_2O emissions (Eagle *et al.*, 2010) due to application of fertilizer, as well as additional organic matter inputs as a result of adding another crop instead of leaving the land fallow, especially where no-till techniques replace summer fallow (Paustian *et al.*, 2000).

In projects that eliminate summer fallow the project developer must ensure that any gains in soil carbon (1) are adjusted by increased GHG associated with agricultural inputs, including production and application of fertilizers, pesticides, and herbicides, seed and seedling production, and irrigation, (2) are adjusted by increased use of machinery for harvest and subsequent transport of crops (Table 5).

Category	Potential Effect on GHG Emissions (kg C/acre)	Potentially Significant? (>5%)*	Comments
Soil Carbon[a]	-97.1 – 259.2	NA	
Soil N_2O/CH_4[b]	-33.1 – 17.7	Yes	
Machinery[c]	-29.15 – -27.3	Yes	Figures are for corn and soy only
Herbicides/ Pesticides[c]	-9.19 – -5.47	Maybe	Figures are for corn and soy only
Fertilizer[c]	-104.63 – -75.4	Yes	Figures are for corn and soy only
Seeds/Seedlings[c]	-8.7 – -8.21	Maybe	Figures are for corn and soy only
Irrigation[c]	-176.7 – 0	Yes	Depends on irrigation type/source
Downstream Transportation[d]	16% - 88% of all secondary effects	Yes	Depends on production volume, transportation mode and distance

Table 5. Leakage Risks from Elimination of Summer Fallow.

[a] Eagle 2010; [b] Eagle *et al.*, 2010 and Wagner-riddle *et al.*, 1997 [c] West & Marland 2002; [d] Meisterling 2009. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

2.a.(iii). Use of Winter Cover Crops

Cover crops are used to protect soil from erosion and increase soil quality through increased soil organic matter and, in the case of leguminous cover crops, increased nutrient availability for the subsequent crop. Compared to the baseline, the use of cover crops to replace a winter fallow can likely increase soil carbon uptake, since the cover crops will increase carbon inputs during the growing stage, and likely add to soil carbon content through cover crop incorporation before spring planting. Cover cropping can potentially reduce fertilizer production and application emissions (Eagle *et al.*, 2010). As a result of the increased nutrient availability, the use of fertilizers can be reduced for the subsequent crop, which can limit both N_2O emissions and nitrate leaching. However, in some cases incorporation of plant residue can increase N_2O emissions, and these increases can be higher than the amount of carbon added to the soils as a result of the practice (Li *et al.*, 2005).

Project developers that implement winter cover cropping must adjust their soil carbon gains by (1) accounting for any increase in inputs used for cover cropping, such as machinery, seed production, and any additional chemical inputs, and (2) accounting for any N_2O emissions that may result from cover crop incorporation (Table 6).

Category	Potential Effect on GHG Emissions (kg C/acre)	Potentially Significant? (>5%)*	Comments
Soil Carbon[a]	40.8 – 357.4	NA	
Soil N_2O/CH_4[b]	-353.3 – 115.9	Maybe	The flux reported was from incorporation of alfalfa cover crop in an alfalfa canola system relative to a corn system, and the outlier is incorporation of other plant residue before planting. Actual amounts between different systems will vary substantially depending on type of cover crop and its use.
Machinery[c]	-9.8 – -2.8	Maybe	Low end for no till, high end for one round of disk.
Seeds/Seedlings[c]	-8.7 – -7.8	Maybe	Figures are for corn and soy, so are only rough estimates for cover crops
Pesticides/ Herbicides[c]	-3.0 – 0	No	Pesticides are usually not used for cover crops, but may be in some cases. Herbicide use will likely decrease relative to the baseline fallow management.
Fertilizer	(+) not estimated	No	Fertilizer use may decrease in subsequent cropping.

Table 6. Leakage Risks from Winter Cover Cropping.

[a] Eagle *et al*. 2010; [b] Eagle *et al.* 2010, Robertson *et al.* 2000, Wagner-Riddle *et al.* 1997, Li *et al.* 2005; [c] West & Marland 2002. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

2.a.(iv). Diversification of Annual Crop Rotations

Diversification of annual crop rotations has the potential to affect soil carbon stocks due to changes in plant productivity and nutrients use, which may lead to increased soil organic matter deposition or a reduction of soil organic matter decomposition. The effects are varied depending on the particular mixtures of crops chosen for this project type. In some cases, this practice could have the opposite effect by causing a reduction in soil organic matter. For instance, converting from continuous corn to corn-soybean appears to reduce soil organic matter, since soybean does not contribute much organic matter, but instead increases soil priming effect, which stimulates soil organic matter decomposition (Johnson *et al.*, 2005). Eagle *et al.* (2010) extensively discuss the impacts of different types of diversification and intensification of crop rotations.

Because of changes to the cropping system, significant changes can occur to overall yields, as well as to the contribution of transportation to GHG emissions of the cropping mix. If a project

results in major changes to the distribution system of the crop (i.e., shifting from a locally consumed/processed crop to a specialty crop that is shipped across the country), the changes in GHG emissions due to increased transportation distances can outweigh the soil carbon savings from a more diversified system. Therefore, this secondary effect should be monitored in this project activity and would require offset developers to gather information on transportation distances in baseline and project scenarios, as discussed in Section 3.

Potential sources of leakage that should be accounted for in projects that incorporate diversification or intensification of crop rotations are (1) significant changes in fertilizer, pesticide, herbicide or lime additions, (2) significant changes in irrigation practices, (3) changes in the distance crops are transported from field to market, and (4) impacts of yield reductions in the baseline crop and potential associated indirect land use change. This last point is due to the fact that crop diversification will lead to reductions in the baseline crop, although will be to some degree negated by the increased yield of new crops (Table 7).

Category	Potential Effect on GHG Emissions (kg C/acre)	Potentially Significant? (>5%)*	Comments
Soil Carbon[a]	-275.8 – 332.0	NA	
Soil N_2O/CH_4[b]	-32.4 – 4.4	Maybe	Crop dependent and highly variable
Machinery[c]	-14.6 – 14.6	Yes	Figures are for corn and soy only
Fertilizer[c]	-52.3 – 52.3	Yes	Figures are for corn and soy only
Irrigation[c]	-88.4 – 88.4	Yes	Depends on irrigation type/source
Herbicides/ Pesticides[c]	-4.6 – 4.6	Maybe	Figures are for corn and soy only
Seeds/Seedlings[c]	-4.35 – 4.35	Maybe	Figures are for corn and soy only
Downstream Transportation[d]	16% - 88% of all secondary effects	Yes	Depends on production volume, transportation mode and distance
Indirect Land Use Change[e]	-24,500 – 0 Grassland -71,700 – 0 Forest -330,800 – 0 Peatland	Maybe	In worst-case scenario, new land is brought into agriculture from natural systems. The values would be modified by a net displacement factor that captures the numbers of acres brought into production

Table 7. Leakage Risks from Diversifying Crop Rotations.**

[a] Eagle *et al.* 2010; [b] Eagle *et al.* 2010 & Robertson *et al.*, 2000; [c] West & Marland 2002; estimated here as half of total range in both directions, assuming that a change in cropping pattern will produce at most half of the total range that would result from new production; [d] Meisterling 2009; [e] Emissions factors converted from Plevin *et al.* 2010. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

** Depending on the crop mixes, the effects can be highly variable, so the values in this table are not as exact as for other project activities.

2.a.(v). Inclusion of Perennial Crops in Rotations

Including perennial crops in rotations every few years can increase soil carbon due to increased belowground carbon allocation by perennial crops. These effects are highly variable, and depend on the type of plants included in rotation, as well as the duration of perennial management and the overall input intensity (fertilizer, irrigation, lime, etc.) of the perennial crop. Some of the main positive impacts, aside from potential increases in soil carbon, include a reduction in machinery use for tillage, as well as a likely reduction in chemical inputs and associated GHG emissions. However, especially in the case of leguminous perennials such as alfalfa, there is a potential for a substantial increase in N_2O emissions (Robertson *et al.*, 2000).

After demonstrating that proposed perennial inclusions will, in fact, sequester soil carbon, project developers must adjust such estimates by the following potential sources of leakage: (1) changes in GHG emissions associated with seed production, and (2) any increases in irrigation, if

applicable, (3) indirect land-use change that is likely due to a reduction of yield of the main crops, (4) changes in the distance crops are transported from field to market (Table 8). We do not foresee other major sources of leakage for this project activity.

Category	Potential Effect on GHG Emissions (kg C/acre)	Potentially Significant? (>5%)*	Comments
Soil Carbon[a]	-193.1 – 242.7	NA	
Soil N_2O/CH_4[b]	-28.4 – 60.7	Yes	Variable, depends on the perennial crop type and management
Seed Production[c]	-8.7 – -7.8	Maybe	Figures are for corn and soy, so are only rough estimates for perennial crops
Irrigation[c]	-176.7 – 0	Yes	Full range of irrigation leakage included here, high end unlikely
Indirect Land Use Change[d]	-24,500 – 0 Grassland -71,700 – 0 Forest -330,800 – 0 Peatland	Yes	In worst-case scenario, new land is brought into agriculture from natural systems. The values would be modified by a net displacement factor that captures the numbers of acres brought into production
Downstream Transportation[e]	16% - 88% of all secondary effects	Yes	Depends on production volume, transportation mode and distance

Table 8. Leakage Risks for Inclusion of Perennial Crop Rotations.

[a] Eagle *et al.* 2010; [b] Eagle *et al.* 2010 & Robertson *et al.*, 2000; [c] West & Marland 2002, [d] Emissions factors converted from Plevin *et al.* 2010; [e] Meisterling 2009. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

2.a.(vi). Organic Soil Amendments

Organic amendments are added to agricultural soils to improve soil quality and to increase nutrient availability for crops. Organic soil amendments include animal manures, municipal refuse, logging and milling waste, sewage sludge, food processing waste, and industrial organics. Organic soil amendments are either (1) applied directly to fields or (2) are initially processed or composted and subsequently applied.

The GHG Assessment Boundary and leakage risks for this project activity are highly dependent on the baseline scenario, i.e., the most likely alternative use for the organic matter. Although it is difficult to determine, Eagle *et al.* (2010) argue that "full life-cycle analysis is especially important with this activity" (p. 19) to understand the net GHG impacts. If organic amendments

were simply applied in one location instead of another, then the net change across the system would be equal to zero. Incorporation of organic amendments would simply move carbon that would have been incorporated anyway (Johnson *et al.*, 2007b). However, if it would otherwise decompose in anaerobic conditions, a change in GHG emissions would occur. This issue is detailed further in Section 3. And finally, the type of organic amendment applied has a major effect on the potential for leakage. For example, direct applications of manure have been shown to completely erase soil carbon increases, and emit three times more N_2O, in CO_2 equivalent, than the amount of soil carbon stored (Li *et al.*, 2005).

Projects that increase soil carbon content through incorporation of organic amendments should (1) ensure proper aeration of material to reduce GHG losses, and (2) adjust the amount of offset by (a) increased machinery requirements to apply organic material, although these may cancel out with emission from baseline fertilizer application and/or may not be significant, (b) possible increase of N_2O and decrease of CH_4, especially in the case of added manure, and (c) emissions associated with producing compost if applicable. Identifying the source of organic amendments and the baseline scenario for their treatment will be critical for this aspect of GHG accounting. Table 9 summarizes leakage risks per acre, and Table 10 summarizes emissions from compost production per ton of feedstock, since figures per acre were not available.

Category	Potential Effect on GHG Emissions (kg C/acre)	Potentially Significant? (>5%)*	Comments
Soil Carbon[a]	19.9 – 562.6	NA	
Soil N_2O/CH_4[b]	Highly variable	Maybe for compost Yes for manure	Depending on the type of organic amendments, this may be a highly significant source of leakage, especially with manure, but compost additions can also generate significant N_2O emissions depending on soil characteristics
Machinery, Application	Depends on amount applied, see Table 10	Probably not	
Compost Production	Depends on amount applied, see Table 10	Yes	

Table 9. Leakage Risks for Organic Soil Amendments.

[a] Eagle *et al.* 2010; [b] Eagle *et al.* 2010 & Li *et al.*, 2005. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

	CARB (2010)	EPA WARM Model	ROU (2001)
Source	**kg C/ton**		
Increased Soil Carbon Storage	70.9	71.9	-
Decreased Water Use	5.5	-	-
Decreased Soil Erosion	35.5	-	-
Decreaesd Fertilizer Use	35.5	-	-
Decreased Herbicide Use	0.0	-	-
Transportation to Processing	-2.2	-4.5	-0.1
Processing	-2.2	-7.4	-5.5
Biogenic CH4	-21.3	-	-51.6
Biogenic N2O	-6.8	-	
Mechanical Application	-	-	-0.9
Total	**114.8**	**59.9**	**na**

Table 10. Carbon Storage, Savings, and Loss from Compost Production and Application; Comparing Three Studies.

2.a.(vii). Switch from Dry Land Agriculture to Irrigated

A switch from dry land to irrigation farming would create additional C sequestration due to increases in plant productivity, as well as potential for a larger number of rotations per year, resulting in higher C deposition belowground. The most important leakage consideration for this project activity is the additional energy required to deliver irrigation water to the field.

Depending on the method of delivery and other factors that determine GHG emissions due to irrigation (Section 3), the additional energy used to deliver water could outweigh the carbon sequestration benefits of irrigation. This is entirely dependent on understanding the specific characteristics of irrigation at any given site. If, for example, irrigation water is delivered by gravity or pumps powered by renewable energy, there is no leakage and this project activity could realize a net gain in carbon sequestration. On the other hand, if irrigation water is delivered from off-site, with a large elevation differential, and in a state reliant on coal-fired power plants, it is likely that this project activity will not produce net GHG emission reductions. In addition, there may be some additional on-site machinery usage necessary to install or move irrigation systems, but this is likely a small, likely negligible amount, and it is not treated in the literature.

Finally, there may be some increases in N_2O emissions from irrigation. As mentioned earlier, and as discussed by Eagle *et al.* (2010), soils with higher water content have greater N_2O emissions, which may negate benefits from carbon sequestration in soils, especially when combined with energy requirements for irrigation. Simojoki and Jaakkola (2000) have shown that the N_2O flux is greatest when soil pore spaces are 60-90% filled with water due to over-irrigation. Projects that increase soil carbon content by switching from dry land to irrigation should (1) demonstrate that over-irrigation does not occur to ensure against large N_2O emissions, adjust the amount of carbon offset by (2) any GHG emitted though delivery of irrigation water, either from off-site or on-site sources, and (3) changes in the impact of transportation from field to market due to either higher crop production or crop switching.

Category	Potential Effect on GHG Emissions (kg C/acre)	Potentially Significant? (>5%)*	Comments
Soil Carbon[a]	125.8 – 526.2	NA	
Soil N_2O/CH_4[a]	-115.8 – -5.5	Maybe	Greatly depends on irrigation efficiency and type, as well as soil characteristics
On-Site Groundwater Irrigation[b]	-115.4 – -50.6	Yes	Depends on elevation change and energy source
On-Site Surface Water Irrigation[b]	0	No	If gravity-fed, no emissions
Off-Site Surface Water Irrigation[b]	-176.7	Yes	Only national average available. Large variation exists due to differences in elevation and fuel type.
Downstream Transportation[c]	16% - 88% of all secondary effects	Yes	Depends on production volume, transportation mode and distance

Table 11. Leakage Risks for Switch from Dry Land Agriculture to Irrigation.

[a] Eagle *et al.* 2010; [b] West & Marland 2002; [c] Meisterling 2009. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

2.a.(viii). Irrigation Improvements

Irrigation improvements are usually realized by switching from furrow irrigation to sprinkler systems or from either furrow irrigation or sprinkler systems to drip irrigation, as well as utilizing sub-surface irrigation. These activities can increase carbon stored in agricultural systems by reducing soil carbon losses due to erosion (Lal, 2004) and potentially increasing crop production (Eagle *et al.*, 2010). Eagle *et al.* 2010 provide an estimated carbon storage range of 19.9 – 64.0 kg C/acre. More efficient irrigation, especially with improved fertilizer management practices, can also decrease N_2O emissions characteristic of irrigated croplands. All of these options will reduce the amount of water used to irrigate land and therefore will not create the main leakage problem of increased energy usage discussed in the section above. The only leakage issue that may occur is the embodied energy of new irrigation systems and any machinery use required to install or maintain them. The embodied energy of new irrigation systems installed for the purposes of GHG offsets would be quite challenging to quantify and is likely relatively small. Estimation of GHG emissions resulting from fuel use by machinery used to install the irrigation system is feasible; however, it would also be a relatively small and

therefore negligible source. Since there are no significant leakage considerations, we do not provide a summary table for this project activity.

2.b. Project Activities Representing a Land Use Change from Cropland to a New Use

2.b.(i). Change from Annual to Perennial Crops

Converting annual croplands to perennial crops has clear benefits in increasing ecosystem carbon through increased productivity, increased aboveground carbon stocks, and increased soil carbon inputs through root deposition and litter deposition. Elimination of tillage and elimination or significant reduction in the use of fertilizers and other chemical inputs, combined with the increase of carbon in the system, have significant GHG mitigation potential (Eagle *et al.*, 2010).

Although the exact issues will depend on the type of perennial crops that are chosen to replace annual systems, the biggest leakage issue in converting annual to perennial cropping systems is indirect land-use change. Even in the cases of conversion of degraded cropland, there is potential of shifting the original crop production elsewhere, either by bringing new agricultural lands into production or through encouraging intensification (Section 3). The loss of yield of the original crop must be accounted for in calculating the GHG emission reductions from such projects.

Additionally, there may be a small source of leakage associated with seed and seedling production in these systems and other agricultural inputs. Although we do not present data on preparing woody species seedlings, it will likely be greater than for alfalfa since the seedlings will require more energy input to grow (see Section 3). Similar small changes may occur in fertilizer usage, herbicide/pesticide usage, and irrigation, although any of these may be either positive or negative in direction. It may be easiest to adjust the soil carbon gains by a small factor to take this potential leakage into account. Also, in the case of using leguminous crops such as alfalfa, there may be some increases in N_2O emissions due to increased nitrogen-rich organic matter deposition, although without residue incorporation, this risk is small. Finally, changes may occur in the distance crops are transported from field to market, which may create a significant leakage risk.

Category	Potential Effect on GHG Emissions (kg C/acre)	Potentially Significant? (>5%)*	Comments
Soil Carbon[a]	0 – 515.2	NA	
Soil N_2O/CH_4[b]	-28.4 – 169.9	Maybe	Depends on the type of perennial; nitrogen-rich short-lived perennials such as alfalfa may contribute more than longer-lived woody perennials
Seeds/Seedlings[c]	-10.85 – 0	Maybe	The estimate is only for alfalfa, actual estimates will depend on the type of perennial seed/seedling
Irrigation[c]	+/- 176.7	Maybe	In case the project scenario increases irrigation relative to baseline, and depending on the type of irrigation, this may become significant
Indirect Land Use Change[d]	-24,500 – 0 Grassland -71,700 – 0 Forest -330,800 – 0 Peatland	Yes	In worst-case scenario, new land is brought into agriculture from natural systems. The values would be modified by a net displacement factor that captures the numbers of acres brought into production
Downstream Transportation[e]	16% - 88% of all secondary effects	Yes	Depends on production volume, transportation mode and distance

Table 12. Leakage Risks for Switching from Annual to Perennial Crop.

[a] Eagle *et al.* 2010; [b] Eagle *et al.* 2010 & Robertson *et al.*, 2000; [c] West & Marland 2002; [d] Emissions factors converted from Plevin *et al.* 2010. [e] Meisterling 2009. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

2.b.(ii). Cropland Conversion to Pasture

GHG emission reductions from land use change can be achieved by converting cropland to pasture. Grazing lands have higher carbon sequestration potential than croplands because there is less intensive disturbance, and plant species favored by grazers allocate more carbon belowground than typical crop species. However there are many leakage risks to address, although most are difficult to calculate, depend on many production variables, and may be either positive or negative.

Baseline emissions of N_2O can be reduced because there are fewer nitrogen-based fertilizers to volatilize. However, N_2O may also increase due to greater soil compaction and subsequent changes in soil moisture dynamics. In addition, the introduction of grazing animals in the pasture system may also cause increased N_2O emissions due to manure and urine inputs. Especially in

the case of ruminants, there is a high probability of increases in CH_4 emissions as well, although these can vary due to diet, variety of cattle, intensity of grazing, and dietary supplements that can reduce the activity of fermenting bacteria (Section 3).

GHGs from on-site activities will also be affected since there is a change in management. Clearly, there would be a reduced need for on-site farm machinery. Eagle *et al.* (2010) suggest that converting cropland to pasture could save 14.3–78.3 kg C/acre per year from fuel use alone. 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 would remain constant. There may also be increased energy required to house animals and GHG emissions from manure management at the housing location. This 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 cropland in order to sustain healthy pasture.

Changes in off-site emissions include possible reductions in use of chemical inputs for cropping, but this may be accompanied by increases in input-related GHGs associated with livestock such as those used for feed and transportation. While some intensively grazed pastures are fertilized, Eagle *et al.* (2010) conservatively suggests that the conversion from cropland to pasture reduces fertilizer use by 25%. Additionally, especially in converting marginal croplands to pasture, there may be increased upstream emissions due to seed production (especially critical in the case of alfalfa). There may also be changes in off-site downstream emissions, including emissions from cattle in later production and processing stages (Pitesky *et al.*, 2009).

Finally, shifting of croplands into pasture also creates potential ILUC leakage issues, since even removal of marginal croplands from production could result in potential shifting of the original crop (Section 3). Therefore, ILUC must be included in this set of leakage risks.

A review of the literature on cropland to pasture conversion suggests an average net impact of 0.48 t C/acre-yr (Eagle *et al.* 2010), with values ranging from –0.07 (the lower limit from Martens *et al.*, 2005) to 0.64 t C/acre-yr (the higher end from McPherson, 2006). Studies of

cropland conversions to pasture (Reeder *et al.*, 1998) and tilled row crops to pasture (Martens *et al.*, 2005) report soil C sequestration potentials in the negative range, so in these cases there can be increased carbon emissions from this conversion. Eagle *et al.* (2010) suggest that process and upstream emissions total an addition average carbon savings of 50.7 kg C/acre, with a range of 26.5-76.1 kg C/acre, although it is unclear if they consider all potential leakage risks mentioned here.

Projects that propose converting cropland to pasture should account for leakage due to (1) increased inputs of seed, fertilizer, and irrigation, if applicable and if higher than baseline, (2) increased on-site emissions associated with housing and maintaining grazing animals, (3) increased emissions of CH_4 and possibly N_2O, in the likely case that input of manure will be higher than baseline fertilizer applications, and due to urine and manure inputs and soil compaction (4) emissions associated with indirect land-use change (Table 13). Off-site effects of housing and processing the animals may or may not be included, depending on how the Reserve ultimately determines boundaries; however, the impacts are not included for food processing activities associated with agricultural crops.

Category	**Potential Effect on GHG Emissions** (kg C/acre)	**Potentially Significant?** (>5%)*	**Comments**
Soil Carbon[a]	0 – 518.5	NA	
Soil N_2O/CH_4[b]	Extremely variable, especially based on type of animals	Yes	Emissions will vary based on manure management, grazer type, and soil properties, and need to be determined on a project-specific basis
Fertilizer	Variable	Probably not	Depends on the difference between baseline and pasture
Seeds/Seedlings	Variable	Probably not	Depends on the type of forage seed that is applied
Irrigation	Variable	Probably not	Depends on irrigation type/source
Indirect Land-Use Change[c]	-24,500 – 0 Grassland -71,700 – 0 Forest -330,800 – 0 Peatland	Yes	Cropland loss is likely offset by crop production in other locations.

Table 13. Leakage Risks from Converting Cropland to Pasture.

[a] Eagle 2010; [b] Eagle 2010 & Brown *et al.*, 2001; [c] Emissions factors converted from Plevin *et al.* 2010. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

2.b.(iii). Cropland Conversion to Uncultivated Use

Removing cropland from production can reduce GHG emissions due to elimination of soil disturbance, as well as avoided on-site emissions from agricultural input manufacturing, application, and volatilization, and from machinery use. Eagle *et al.* (2010) find the average net impact of cropland to set-aside or herbaceous buffers (strips of land planted to reduce wind and water erosion) is 0.51 t C/acre per year. Off-site, downstream emission reductions include avoided emissions from transporting crops off-site, as well as avoided emissions from agricultural inputs production.

Any emission reductions from cropland conversion to uncultivated use would have to consider ILUC from displaced agricultural production to prevent leakage (Table 14). Offset design should consider whether cropland conversion to uncultivated use will lead to land use changes from uncultivated use to cropland elsewhere. There are also many important co-benefits from this approach such as those to wildlife and water quality that should also be considered in evaluating impacts of offset design.

Category	**Potential Effect on GHG Emissions** (kg C/acre)	**Potentially Significant?** (>5%)*	**Comments**
Soil Carbon[a]	-16.5 – 522.9	NA	
Indirect Land Use Change[b]	-24,500 – 0 Grassland -71,700 – 0 Forest -330,800 – 0 Peatland	Yes	Cropland loss is likely offset by crop production in other locations.

Table 14. Leakage Risks for Switching from Cropland to Uncultivated Use.

[a] Eagle *et al.* 2010; [b] Emissions factors converted from Plevin *et al.* 2010. All units have been converted to the C equivalent, expressed as kg C/acre.

* Leakage is counted as potentially significant when it is both negative and the range exceeds 5% of the range of potential soil carbon storage. "Maybe" is given when the leakage risk would be greater than >5% only when soil carbon gains are at the low end of the range.

3. Details On and Potential Quantification Methods for Relevant Secondary GHG Emission Sources and Sinks Associated with Cropland Management and Land Use Changes

3.a. On-Site Emissions

3.a.(i). Machinery Use

It will be important to consider GHGs associated with machinery in the GHG Assessment Boundary for the following cropland management project activities: winter cover crops, reducing summer fallow, organic soil amendment, and potentially diversifying crop rotation. Other activities may cause a decrease in GHGs emitted from machinery use, including switching from conventional to conservation and no-till agriculture, as well as conversion to grazing or uncultivated use.

To consider the importance of computing leakage associated with changes in machinery use, it is useful to understand the contribution of machinery use to the overall agricultural carbon budget. For various types of corn-soybean-alfalfa cropping systems, Adler (2007) found that annual GHGs associated with machinery use ranged from 23.2-43.1 kg C/acre (Table 15, adapted from Adler 2007). West and Marland (2002a) found an average of 27.9 kg C/acre for corn, soybean, and wheat under conventional practices. This compares to an average of 40.1 kg C/acre on average for agricultural inputs, after normalizing for the percent of area treated, including fertilizers, pesticides, lime, and seed production (excluding irrigation). Therefore, according to these studies, machinery use comprises about 41% of total farm GHG emissions, not including changes to soil carbon. In the small percentage of fields in which all inputs are used (see Table 2), the proportion attributed to machinery can be as low as roughly 15%, but this is not the norm.

Since different project activities will only affect certain phases of machinery use, it is important to determine the GHG significance of various machinery practices. Adler *et al.* (2007) analyzed GHGs associated with machinery usage for six bio-energy crops, and values for a corn-soybean and a corn-soybean-alfalfa rotation show the relative contribution of each phase of machinery use to overall GHG emissions from machinery (Table 15). In this example, tillage, harvesting, and drying are each roughly one third of total machinery-related GHG emissions (note that drying is not required for many agricultural goods.) West and Marland (2002a) found similar results. They use a finer categorization of machinery use, exclude drying, and include GHGs

associated with manufacture, transport, and repair of farm equipment (Table 16). Their results are similar to those of Adler *et al.*, although West and Marland show that GHGs associated with harvest seem to be lower and those associated with planting, pesticides and fertilizer application seem to be higher.

Adler *et al.* (2007) also found that manufacture, transport, and repair of farm equipment accounts for 16% of total energy associated with machinery use (Table 16). This suggests that this category of upstream emissions may be important to include in calculations of GHG emissions associated with machinery.

	Corn-Soybean		Corn-Soybean-Alfalfa	
	kg C / acre	% of total	kg C / acre	% of total
Tillage	12.40	28.8%	7.77	26.90%
Harvesting	13.22	30.7%	9.74	33.70%
Planting, pesticide, fertilizer application	3.27	7.6%	3.44	11.90%
Drying (propane)	14.17	32.9%	7.98	27.60%
Total	43.06		28.90	

Table 15. GHG Emissions from Agricultural Machinery and Inputs for Two Cropping Patterns. Adapted from Adler 2007.

	Operation (MJ/acre)	Manufacture, Transport, Repair (MJ/acre)	Total Emisisons (kg C / acre)	Percent of Machinery GHGs
Moldboard Plow	454	41	10.83	30.8%
Disk	140	22	7.06	20.1%
Planting	103	23	2.75	7.8%
Single Cultivation	68	17	1.85	5.3%
Fertilizer Application	205	24	5.00	14.2%
Pesticide Application	25	23	1.03	2.9%
Harvest (combine)	232	75	6.67	19.0%
Total	1,227	226	35.17	100%
Percent of Total MJ	84%	16%		
Note: assumes two rounds of disk tillage				

Table 16. GHG Emissions from Agricultural Machinery and Inputs. Adapted from West and Marland 2002.

A final consideration for emissions from machinery involves not leakage of GHGs, but additional emissions of other air pollutants. Increasing air pollutants as a result of machinery use, especially in Clean Air Act non-attainment areas, would be counter to Reserve principles. The Reserve requires project developers to demonstrate that their GHG projects will not undermine progress on other environmental issues such as air and water quality, endangered species and natural resource protection, and environmental justice (Reserve Program Manual, p. 12). Offset rules should ensure that any significant increases in air pollutants covered under the Clean Air Act do not result from particular projects. Some examples where criteria air pollutants could increase include projects included in air basins, such as the California Central Valley. Consultation with air pollution control boards may help mitigate potential issues that related to significant contributions to criteria air pollutants. In some situations air quality would improve if projects reduce dust emissions from erosion, as would happen with cover cropping and no-till agriculture.

Emissions associated with fuel use for machinery can be calculated using standard emissions factors. The Reserve has directed all project developers to use the most recent EPA eGRID emissions factors, which allow straight forward conversions from electricity use to GHG emissions. Directly accounting for GHG emissions from machinery requires detailed record keeping at the project level that would allow computation of gallons of fuel or kWh used per acre. In addition, farm machinery would need to be designated by project in order to allocate percentages of use of a particular machine for a particular project area/crop for any given project. The most straightforward calculation method would be the following, for each piece of farm equipment, along with appropriate unit conversions:

C/Project Acre = Total Gallons X Percent Time Used for Crop / Acres of Crop X Fuel Emissions Factor

Variations on this approach may be necessary depending on how/if records are kept. For example, records for equipment use may be kept in number of field passes or not kept at all. Total gallons may be known for the farm but not per piece of machinery on a project level, in which case factory specified MPG and total miles driven could be used. For other types of

machinery, hours per gallon of operation may be available, rather than MPG. In these cases, hours per gallon or MPG could be used in conjunction with time in use or miles driven to determine fuel usage. In sum, these calculations should be straightforward, but the Reserve protocol should specify what type of records should be kept during the project to ensure data are available to estimate project emissions. Baseline emissions, however, may need to be based on historically available records. In which case, some flexibility may be allowable for the input data required to estimate baseline emissions.

When sufficient data are not available, estimation methods may be used to understand GHG emissions produced from machinery, although this clearly will not be as accurate. One way to normalize these calculations may be to use existing data such as the American Society of Agricultural Engineers machinery management data (American Society of Agricultural Engineers, 2000 in Adler et al., 2007). An example of normalizing calculations using this data can be found in the Integrated Farm Systems Model (Rotz, 2004 in Adler et al., 2007). In addition, manufacturer data can also be used to estimate GHGs from machinery. Thus, using existing data to create a Reserve-specific estimation methodology is one potential avenue to developing a methodology, particularly for baseline estimation.

3.a.(ii). Application of Organic and Other Amendments

Crop production relies on inputs either produced off-site, such as manure, compost, herbicides, lime, and pesticides, or on-site, such as cover crops. Volatilization of some of these inputs can lead to GHG emissions. This section deals specifically with emissions of GHG associated with application of inputs, but not machinery related emissions. These emissions may be important to address in any project activity that changes the levels of their use.

Incorporation of organic soil amendments is beneficial for various reasons. Such practices supply nutrients in a time-release form, increase soil organic matter content resulting in better soil aggregation, improve other soil quality factors, and store carbon. Incorporation of organic material can be a result of cover crop use, in which case the material is plowed under using machinery or it can be a result of import of off-site organic matter, such as compost or animal manure. Regardless of treatment, all studies of organic soil amendments show sequestration

potential. A wide range of 0.02 to 0.5 t C/acre yr^{-1} is reported by Franzluebbers (2005) for poultry litter in the Southeast U.S., while other ranges include 0.08 – 0.2 t C/ acre yr^{-1} (Follett, 2001). There are two main GHG considerations associated with soil amendments, including changes in GHG emissions resulting from incorporation of material into soil, and emissions associated with the production of and transport of amendments. These two topics are treated separately below, first focusing on the on-site emissions and later on the off-site emissions related to production and transport of organic amendments (see Section 3.b.(iv)).

Incorporation of Organic Material into Soil: The increase in soil organic matter as a result of compost and other off-site organic residues can constitute a substantial carbon sink that greatly depends on the intensity and duration of the practice. In addition, there are several secondary effects that need to be taken into consideration in the incorporation of organic matter inputs. Aside from increasing organic matter content and improving soil quality, incorporation of organic materials reduces the need for carbon-intensive fertilizers and increases soil tilth, thereby reducing machinery fuel use. On the other hand, application of organic materials can also increase N_2O and CH_4 emissions if not managed properly, which can constitute an additional source of leakage. However, this may have also occurred in a baseline scenario if this material was allowed to decompose under anaerobic conditions. As described later, understanding the alternative fate of organic amendments is both difficult and crucial, as it has important implications for the actual net GHG effects.

Compost: Incorporation of compost changes soil physical characteristics, such as soil moisture, soil temperature, and nitrogen content, and these changes depend greatly on the original feedstock material (Favoino *et al.*, 2008). Depending on the state of decomposition and the lignin-nitrogen ratio of the parent material (especially if the compost is partially or wholly based on animal manure or any other nitrogen-rich substrate), there may be additional N_2O and CH_4 emissions after application due to continued decomposition of primary material under an oxygen-limited environment. Due to the extremely variable range of these emissions, the actual conditions for individual projects should be monitored and evaluated using available modeling tools, such as DAYCENT.

Manure: Franzluebbers (2005) notes that, although there is a general lack of studies of the use of manure as fertilizer, the range of soil organic matter benefits is large and such applications increase soil organic matter content sometimes by an average of 15% of the original weight of amendments. Although the amount of carbon sequestered increases with the amount of manure applied, increases above a certain threshold are likely to result in negative environmental effects such as nitrate leaching (Wood *et al.*, 1996, Johnson *et al.*, 2007a). Also, depending on the moisture content of the applied manure, substantial CH_4 and N_2O emissions are possible (Amon *et al.*, 2006).

Cover Crops: The use of cover crops is widespread in organic agriculture since the incorporation of cover crop before planting greatly enhances soil quality and nutrient availability, especially in the cases of leguminous cover crops. Although cover crop incorporation has the potential of producing N_2O emissions, the effects appear to be short lived, and are limited to crop residues with a low C:N ratio (Baggs *et al.*, 2000). Aside from additional fossil fuel requirements to operate machinery, there are no major emissions from incorporation of cover crops, and on balance cover cropping appears to be more important than tillage practices for retaining and increasing soil carbon, significantly increasing soil carbon content under all tillage scenarios (Veenstra *et al.*, 2007, De Gryze *et al.*, 2009).

Lime Application: Another GHG-intensive input in agriculture is lime, which is typically a mix of limestone and dolomite. The primary purpose of soil liming is to influence soil pH and decrease soil acidity. Applications of lime result in CO_2 emissions due to the chemical reaction between lime and acids present in the soil. Although this is not a widespread practice, the EPA estimates that the use of lime in agriculture contributes almost 2% of all agricultural GHG emissions (EPA, 2011b). However, West and McBride (2005) suggest that the estimates that the IPCC and the EPA use to characterize emissions from lime use may be higher than the real emissions, primarily due to riverine transport of bicarbonates. Nonetheless, applications of agricultural lime result in significant CO_2 emissions and produce, at minimum, 20% emissions per weight of lime used. West and McBride (2005) estimate that from the 20-30 Tg of lime used in the U.S., 4.4-6.6 Tg CO_2 are produced. Aside from the energy needed to produce and transport lime to the field (see below), applications of lime can be a significant source of leakage, and

should be considered within the scope of the CMPP. This is especially important in the cases of no-till or reduced till projects, since these approaches are frequently accompanied by liming to reduce acidification of soils, typical following adoption of such practices (Fox *et al.*, 1981, Blevins *et al.*, 1983, Robertson *et al.*, 2000). These applications of lime in order to manage soil pH would constitute additional sources of leakage.

Special Considerations Regarding Increased Application of Pesticides and Herbicides: We have not identified significant sources of leakage from proposed project activities through increased application of pesticides and herbicides, although if project activities result in major changes in the rates of application, this issue may need to be revisited. However, in addition to potential GHG emission issues, the use of these chemicals poses significant and widespread environmental issues (Lal, 2004). Any increase in use, especially due to no-till or conservation till practices, may have increased adverse impacts on the surrounding ecosystems and native plant and animal populations. Therefore the Reserve may choose to limit increases in pesticide and herbicide use in all project activities.

3.a.(iii). N_2O Emissions from Soil

The first significant issue is potential for increases in N_2O emissions following conversion of conventional-till to conservation-till management. Several studies have suggested that N_2O emissions can constitute a considerable GHG flux from no-till systems under conditions of inadequate soil aeration and excessive soil moisture (Six *et al.*, 2002, Rochette, 2008, De Gryze *et al.*, 2009). The flux can be large enough to offset any of the soil carbon sequestration benefits of conservation tillage. There is a considerable level of uncertainty in quantifying the additional fluxes of N_2O due to changes in tillage practices; however, projects on poorly drained soils have a higher likelihood of such effects and should account for this potentially considerable source of leakage. The second issue is that in such soils, acidification may require liming, the effects of which are discussed above (and cause CO_2 emissions). Third, N_2O emissions can also increase due to irrigation in no-till systems, due to poor aeration of soils under no-till management, and increasing soil moisture (Rochette, 2008).

Finally, N_2O emissions can increase from fertilizer use. The primary issue with GHG emissions from fertilizer application is the rate of emissions of N_2O. Total N_2O emissions from U.S. fields are 204 Mt CO_2e/yr (55.6 Mt C/yr) (Paustian *et al.* 2004 cited in Eagle *et al.* 2010). Up to a certain level of application, additions of nitrogen-based fertilizers increase productivity, and therefore increase carbon storage. However, above an optimum level of fertilizer additions, if the applied nitrogen is not taken up by plants or the microbial biomass, there is significant potential of denitrification of NO_3 and nitrification of NH_4 by micro-organisms, which are mainly responsible for N_2O emissions. These emissions can more than offset the increased gains of carbon in the soil, and thus can result in increased emissions in some circumstances.

To the extent that project activities under the CMPP are likely to affect fertilizer management, on-site N_2O emissions from nitrogen-based fertilizers could be a source of leakage if the project increases fertilizer application (Kim *et al.*, 2008), such as in the case of eliminating summer fallow, as well as changing cropping regimes to incorporate crops with potentially higher nitrogen requirements. In addition, excessive inputs of both nitrogen and phosphorus fertilizers results in increased concentration of these fertilizers in streams due to runoff, which contributes to estuarine eutrophication, and associated ecosystem disturbance (Tilman *et al.*, 2002). Therefore, if project activities do increase nitrogen-based fertilizer use, then there will be leakage risks that should be accounted for.

Since N_2O emissions vary significantly based on overall nitrogen concentration in the soil, as well as other soil properties, such as soil moisture content, an estimation approach is necessary. There are several ways that this can be accomplished, using both top-down estimates such as the IPCC Tier 1 emission factor methodology or using process-based models, such as DAYCENT or DNDC (which is consistent with IPCC Tier 3 approaches). Li *et al.* (2001) compared the IPCC Tier 1 methodology with DNDC, and found that, although on a national scale the IPCC Tier 1 methodology arrived at similar results as the DNDC model, there were very important regional dynamics that the IPCC Tier 1 methodology missed. For the purposes of the CMPP, since the scale of projects is relatively small, compared to the area of whole nations, the IPCC Tier 1 methodology may be inadequate to capture site-specific N_2O emissions. Using process-based models is likely a better alternative, even though they require more data in order to be effective.

Additionally, since models such as DAYCENT and DNDC also account for soil carbon changes, using such models can accomplish several tasks that are required to estimate the impact of the proposed project activity at once.

3.a.(iv). Irrigation: On-Site Water Pumping and Transport

Delivering irrigation water to crops requires transporting water from a source to the field. Although non-irrigated areas comprise 85% of U.S. corn crops, 95% of soybean crops, and 93% of wheat crops by areas (USDA, 1997), when irrigation is used, it can be a large contributor to agricultural GHG emission. On balance, increased carbon storage as a result of changes in agricultural practices may be offset to some degree by GHGs emitted in the delivery of water (Schlesinger, 1999, Mosier *et al.*, 2005). Therefore, energy use to pump water is a leakage concern for these project activities: diversifying crop rotations, inclusion of perennial crops, removal of summer fallow, switching from dry land to irrigation, and conversion to pasture. Reduced emissions from irrigation may occur for these project activities: switch to conservation or no-till practices or from diversifying crop rotations, irrigation improvements.

Unless irrigation water is delivered by gravity, energy is required to pump water from on-site groundwater sources (as well as from off-site surface water sources, see Section 3b). Since fossil-based energy is used in the vast majority of cases, this creates a source of GHG emissions. More than half of U.S. on-site irrigation pumping is done using electricity, with diesel and natural gas comprising roughly 23% and 17% respectively, and gasoline and propane-fueled motors at less than 5% each (West *et al.*, 2002a). If a project activity of the CMPP requires additional irrigation, this potential source of leakage should be considered within the GHG Assessment boundary. Namely, a project's direct emissions may increase as a result of a project activity if diesel, natural gas, propane or gasoline are used, and therefore this leakage would need to be calculated and subtracted from carbon stored through the project activity. If on-site pumps are powered by electricity produced off-site, then a project's indirect emissions would increase, and the carbon intensity of purchased electricity would need to be determined and used to report leakage. This can be done using established Reserve methods from existing offset protocols. For instance, the Reserve Livestock Project Protocol and the Landfill Project Protocol both contain methodologies for calculating emissions from on-site fuel use and electricity use (Livestock

Project Protocol Version 3.0, Equation 5.11, and Landfill Project Protocol Version 3.0, Equation 5.10, 5.11, and 5.12).

According to Schlesinger (1999), GHGs emitted in pumping irrigation water ranged from 89-336 kg C/acre, depending how water is delivered. West and Marland (2002a) estimated GHG emissions from different types of fuel using national level data on total U.S. area by irrigation type, total energy expenditures and energy costs (Table 17). For on-site pumping, figures range from 50.6-115.4 kg C/acre, for a U.S. average of 96.83 kg C/acre. On-site surface water irrigation, which is delivered by gravity, does not require pumping and therefore does not emit GHGs. Therefore, the CMPP should distinguish the method of on-site water delivery. Of total on-site irrigation water in the U.S., by area irrigated, 83% is from groundwater and 17% from surface water. Off-site irrigation water, discussed below, is nearly twice as greenhouse gas intensive on average, and may reach the levels mentioned in Schlesinger (1999).

	U.S. Area Irrigated (million acres)	GJ/acre	kgC/acre
Electricity	3.24	2.15	107.69
Natural Gas	1.00	7.94	115.44
LGP	0.26	2.71	50.70
Distillate Fuel	1.35	3.05	66.91
Gasoline	0.03	2.40	50.98
Total	*5.86*		
Average		3.36	*96.83*

Table 17. Area and GHG Emissions from On-Site Groundwater Pumping by Fuel Type. (Adapted from West and Marland 2002)

The values in Table 17 are based on national averages and therefore are a coarse estimate of emissions from any given farm. Emissions from pumping can greatly vary by irrigation requirements (which vary by crop type and region), the vertical elevation distance from source to field, and the pressure required at the site of irrigation (which varies by irrigation technology). Therefore, using these average coefficients may give an inaccurate accounting of GHG emissions. For on-site irrigation, a more accurate approach would be to directly monitor fuel used in pumps, allocate this by crop depending on irrigation requirements, normalize by acre, and multiply by the appropriate emissions factor depending on fuel type. Estimation

methodologies based on existing models are described in Section 3b, but for on-site pumping direct estimation is both preferable and probably feasible.

3.a.(v). GHG Emissions from Livestock

Incorporation of livestock into agricultural practices has the potential to affect multiple sources of GHG emissions. The major benefits of converting agricultural lands to pasture systems are: (1) a reduction in tillage-related GHG emissions both from soil organic matter decomposition and machinery use, (2) a possible reduction of chemical inputs, (3) an associated reduction in upstream emissions from input manufacture and transportation, (4) increases in soil organic matter buildup from plant cover in grazing systems through higher amounts of belowground carbon deposition, and (5) in arid and semi-arid systems a significant belowground increase in soil inorganic carbon, which may be more than twice as large as the increase in soil organic carbon (Reeder *et al.*, 2004).

However, significant sources of GHG leakage exist in pastureland systems that may reduce the total GHG benefits of conversion projects. The majority of emissions associated with cattle are CH_4 resulting from enteric fermentation in ruminants. To a smaller extent, large intestinal emissions of CH_4 from pigs and other non-ruminant animals and birds can also contribute to leakage. The quantities of enteric methane depend on several factors, such as feed, type of cattle (dairy or meat), age of cattle, as well as other factors, and can vary the emissions produced by as much as 14% (Monteny *et al.*, 2006). In addition, animal manure can be a considerable source of CH_4 and N_2O emissions, but the amounts greatly depend on manure management practices. CH_4 emissions are greater from concentrated slurry managed systems which are outside of the scope of this discussion since these systems are not used in pasture managed systems. However, the production of N_2O from manure and urine deposition on the soil is another potential source of leakage. By weight of application, these emissions can exceed N_2O emissions from fertilizer applications by a factor of two (Brown *et al.*, 2001). In addition, N_2O emissions from soils can be increased as much as seven times as a result of soil compaction due to grazing activity (Bhandral *et al.*, 2007).

There are also somewhat more complicated issues to consider when we examine adoption of grazing systems. Different grazing systems (intensive versus traditional) have different rates of soil carbon deposition, and management plays a critical role in the total amount of carbon sequestered (Eagle *et al.*, 2010). Additionally, grazing systems often involve irrigation, fertilization, and other practices that have their own associated leakage issues, although they have the capacity to further increase carbon sequestration. And finally, infrastructure to house grazing animals requires additional energy inputs, and is associated with further emissions of GHGs. Although there can be significant soil carbon gains in pasturelands relative to croplands, there are substantial leakage issues that primarily center around N_2O and CH_4 emissions that should be included in the GHG Assessment Boundary of the CMPP for projects that incorporate replacement of croplands with pasture.

3.b. Off-Site Upstream Emissions

3.b.(i). GHG Emissions from Fertilizer, Herbicide, Pesticide, and Lime Production and Transport

Fertilizer, lime, and pesticides are produced in energy intensive manufacturing facilities and derive from extractive industries. Reductions in fertilizer, lime or pesticide production could lead to significant GHG reductions (West *et al.*, 2002a), while project activities that increase their use would create leakage. Eagle *et al.* (2010) suggests that reduced agricultural chemical use would not be eligible for GHG credit offsets since emissions may be capped at the GHG source; however, that will depend on the policy context and is not unique to this source alone.

Accurate estimates of life cycle energy use from input extraction, manufacturing, and transportation can be obtained from fuel and electricity use and source profiles. The emissions associated with manufacturing these inputs can be estimated using life cycle assessment (LCA) approaches in units of carbon per unit area-year (for example, ton of carbon dioxide equivalents per acre-year). Reported values for the carbon intensity of these inputs are widespread in literature on life cycle analyses of biofuels (Hill *et al.*, 2006). For example, Liska *et al.* (2009) have published emissions factors for various agricultural inputs. The following section highlights some of the sources for carbon intensities and the associated carbon accounting issues that would complicate protocol design and Table 18 gives an overview of primary energy use.

Product	Total Production Energy (Btu/g)
Nitrogen*	45.84
Phosphate	13.31
Potash	8.42
Herbicides	273.26
Pesticides	312.43

Table 18. Energy Used to Manufacture Agricultural Inputs. (CARB 2009)

Fertilizer: Fertilizer production consumes 1.2% of global energy supply (Kongshaug 1998). The goal of applying fertilizer is to deliver nitrogen, phosphorus, and potassium to crops. Nitrogen based agricultural fertilizers include anhydrous ammonia, aqueous ammonia, ammonium nitrate, calcium ammonium nitrate, urea ammonium nitrate, and urea. Carbon intensity values for each are typically addressed in most life cycle analyses (LCAs) of biofuels production because they are the primary nitrogen-based fertilizers used. There are GHG emissions factors for most nitrogen-based fertilizers available in the peer reviewed literature (Table 19) and default values are built into LCA modeling tools such as GREET (Landis *et al.*, 2007). The manufacture, distribution, and transportation of N fertilizer results in between 872-1,126 kg C per ton of N fertilizer manufactured. Total N fertilizer consumption is 13.6 Mt N/yr (Millar *et al.* 2010; USDA ERS 2010a, both cited in Eagle 2010).

Product	Region	g CO_2e/kg N	g CO_2e/kg product	Reference
Ammonia	USA	1491.5	1223	DOE 2000 (Wood)
Ammonia	USA	1536.6	1260	EPA 1993 (Wood)
Nitric Acid	USA	2818.2 – 12,681.8	620 – 2790	IPCC 2000
Nitric Acid	USA (no NSCR)*	13,386.4	2945	IPCC 2000
Nitric Acid	USA (NSCR)	2818.2	620	IPCC 2000
Urea	Europe Ave.	4018.9	1848.7	Davis and Haglund 1999
Urea	Europe Ave.	1326.1	610	Kongshaug 1998
Urea	Europe Modern	913	420	Kongshaug 1998
UAN**	Europe	3668	1173.8	Kuesters and Jenssen 1998
UAN	Europe Ave.	5762.9	1844.1	Davis and Haglund 1999
UAN	Europe Ave.	4093.8	1310	Kongshaug 1998
UAN	Europe Modern	2000	640	Kongshaug 1998
Ammonium Nitrate	Europe Ave.	7030.8	2460.8	Davis and Haglund 1999
Ammonium Nitrate	Europe Modern	2985.1	1000	Kongshaug 1998
Calcium Amm. Nitrate	Europe Ave.	7481.9	1982.7	Davis and Haglund 1999
Calcium Amm. Nitrate	Europe Modern	3018.9	800	Kongshaug 1998
Nitrogen Fertilizer	USA	857.5	–	West and Marland 2001
Mean N Fertilizer	Germany	5339.9	1479.1	Patyk 1996

Table 19. GHG Emission Factors for Nitrogen Fertilizers.

* Non-Selective Catalytic Reduction Technology

** Urea Ammonium Nitrate

Most nitrogen fertilizer is made from ammonia (NH_3). While the nitrogen source is mostly N_2 from air, hydrogen is obtained from natural gas (~80%) or heavy fuel oil. The energy intensity is in the range of 25-35 GJ/ton of ammonia (see Patyk, 1996, Kongshaug, 1998, Davis *et al.*, 1999, DOE [Department of Energy], 2000). An ammonia manufacturing plant produces CO_2 at a rate of 1.15 to 1.3 kg/kg ammonia (Kim *et al.*, 2005).

Nitric acid is a product of ammonia synthesis and is the largest industrial source of N_2O (IPCC). N_2O is formed as a byproduct and emitted to the atmosphere during the nitric acid production process during the catalytic oxidation of ammonia. N_2O emissions from nitric acid manufacturing are highly variable, ranging from 0.550–2.945 kg CO_2/kg nitric acid (Davis *et al.*, 1999, Wood *et al.*, 2004), and in some cases values are as high as 5.890 kg CO_2e/kg nitric acid.

Variations can be explained by differences in N_2O abatement technologies. Emission from nitric acid production may appear low because some LCAs credit steam exports, which means they are credited for displacing fossil fuel use in other parts of industrial parks. In general, accurately quantifying these emissions on a project level is unrealistic and may therefore be determined to lie outside the project boundary.

Ammonium nitrate (NH_4NO_3) and calcium ammonium nitrate (CAN) are commonly used fertilizers with emission data listed in Table 19. Ammonium nitrate is produced from ammonia and nitric acid. Mixing in limestone before the reaction product becomes a solid makes calcium ammonium nitrate (CAN). Most of the emissions from these processes come from the nitric acid and ammonia synthesis above. Transportation of raw materials and intermediate products account for 1-3% of emissions (Patyk, 1996, Davis *et al.*, 1999).

Urea makes up 50% of global fertilizer production (UNEP [United Nations Environmental Program], 1996). CO_2 and ammonia are the raw materials for producing urea, and 0.74–0.75 kg CO_2 produces 1 kg urea (Kim *et al.*, 2005). There are several claims to award carbon credits in life cycle accounting methodologies for urea production because CO_2 is ostensibly removed from the atmosphere (Liska *et al.*, 2009). But since most CO_2 inputs come from typically adjacent ammonia synthesis, the sources of carbon are fossil fuels and should not be counted. The IPCC recommends "no account should be taken for intermediate binding of CO_2 in downstream manufacturing processes and products" (IPCC [Intergovernmental Panel on Climate Change], 1996).

Other fertilizers include phosphorous-based fertilizers. Emissions factors for Single Superphosphate (SSP), Triple Superphosphate (TSP), Diammonium Phosphate (DAP), and Monoammonium Phosphate (MAP) are listed in Table 20. Most are made from phosphoric acid, a product of phosphate rock and sulfuric acid. Sulfuric acid is the most widely manufactured chemical in the world and the largest single buyer is the fertilizer industry. Most of the literature on this topic uses estimates from European manufacturing and most emissions are related to energy consumption. Most of these reactions are exothermic and lead to steam and hot water exports, which may reduce overall energy usage. Some studies included the transportation of raw

materials and intermediate products, which accounted for one third to 20-25% of overall emissions (Patyk, 1996, Davis *et al.*, 1999).

Product	Region	g CO_2e/kg P_2O_5	g CO_2e/kg product	Reference
SSP	Europe Ave.	1051.8	220.9	Davis and Haglund 1999
SSP	Europe Ave.	95.2	20	Kongshaug 1998
SSP	Europe Modern	-238.1	50	Kongshaug 1998
TSP	Europe Ave.	1083.5	520.1	Davis and Haglund 1999
TSP	Europe Ave	354.2	170	Kongshaug 1998
TSP	Europe Modern	-416.7	-200	Kongshaug 1998
MAP	Europe Ave.	1352.4	703.2	Davis and Haglund 1999
MAP	Europe Ave.	596.2	310	Kongshaug 1998
MAP	Europe Modern	-519.2	-270	Kongshaug 1998
DAP	Europe Ave.	1883	866.2	Davis and Haglund 1999
DAP	Europe Ave.	1000	460	Kongshaug 1998
DAP	Europe Modern	-388.9	-152.2	Kongshaug 1998
Mean P Fertilizer	Germany	817.3	263.2	Patyk and Reinhardt 1996
Mean P Fertilizer	Germany	458	177.7	Patyk 1996
Mean P Fertilizer	Germany	700	248.5	Kaltschmitt and Reinhardt 1996
P Fertilizer	USA	165.1	–	West and Marland 2001

Table 20. Emissions Factors for Phosphate Fertilizers.

The above discussion covers most fertilizers and their standard manufacturing processes. Multi-nutrient NPK fertilizers are also used but there are no standard manufacturing approaches, so accounting for leakage would be more complex. In all cases, manufacturing location is also important, which could cause accuracy issues since most LCA studies rely on European data. Other life cycle analysis impact categories of importance in addition to GHGs are the depletion of inputs such as phosphate rock, raw phosphate, potash, and limestone (Brentrup *et al.*, 2004). Because of the nature of chemical manufacturing, there are also possible other environmental and environmental justice considerations.

Insecticides and Herbicides: Pesticides are also energy intensive to produce, package, and transport and are almost exclusively made from fossil fuels. Therefore, in addition to emissions from manufacturing, emissions are embedded in the product itself, as well as in any associated emulsifiable oils, wettable powders or granules. Since most pesticides used in the U.S. are non-persistent, we can expect these to be volatilized within weeks of application. Overall, whether dealing with herbicide (266.56 GJ/Mg) or insecticide (284.82 GJ/Mg), most pesticides are relatively similar in terms of embodied energy (West *et al.*, 2002a). The translation from energy to carbon would depend on where the pesticides were manufactured, since the energy sources will differ between countries or within areas of a country. Calculations would use the EPA eGRID emission factors for different areas; however, it may be too challenging to develop separate estimates for each facility for the purposes of the Reserve. In addition, many chemical plants directly use energy on site to generate steam. Accurate estimates of GHG per unit means accounting not only for fuel consumption, but also for differences in carbon intensities for the fuels used (e.g., natural gas, propane, coal, diesel, etc.). Rather than requiring calculation of GHG emission from this source, a conservative approach would be to require implementation of practices, e.g., integrated pest-management, that ensure chemical applications are limited in the project (Lal, 2004).

The manufacturing process for pesticides is a significant portion of life cycle GHGs of these inputs. Few detailed LCAs exist on herbicide manufacturing in the published literature. Those that use data from the EcoInvent database find that herbicide manufacturing constitutes about 51% (0.35 GJ/ha) of the total contribution to energy use associated with using herbicides (Gasol *et al.*, 2007). Bennett *et al.* (2004) look at the impacts of herbicide manufacturing and use on summer smog, acidification, and toxic particles, which have both negative and positive effects on overall radiative forcing, therefore making direct GHG emission conversions challenging.

Lime: Lime is used as a soil amendment to change the pH of acidic soils. Emissions factors for various products such as quicklime and byproducts such as calcine wastes are available in most standard LCA models. The primary emissions from lime production are CO_2 released from heating limestone and CO_2 associated with energy consumption for source rock crushing. One way to account for these emissions is to use IPCC general methodology, and the IPCC emission

factors (IPCC Tier 1 or Tier 2). However, in order to get more precise numbers, it can be useful to incorporate source-specific and facility-specific calculations (Similar to IPCC Tier 3), since agricultural lime is not necessarily a uniform commodity. The National Lime Association's CO_2 Emissions Calculation Protocol for the Lime Industry (2005) has developed a means to quantify carbon emissions directly from lime production using the following equation based on mixtures found at particular plants.

$$EF = [(\% \text{ total CaO} * 0.7848) + (\% \text{ total MgO} * 1.0918)]$$

The emissions factor (EF) is reported in tons of CO_2 per ton of lime, and the factors for CaO (0.7848) and MgO (1.0918) are based on stoichiometric balances with CO_2. This factor would be multiplied by the tons of lime and added to the tons of CO_2 emitted from the energy consumed in the process. Default values are available for various fuels, but the National Lime Association recommends actual data because coal and petroleum coke emissions factors can vary, and other sources of energy at lime plants such as tired and landfill gas have site specific emissions factors. In some cases byproducts of lime sequester carbon, so emissions reductions also occur. In other cases the calcium oxide (CaO) and stable carbonates formed will reabsorb some of the CO_2 emissions. Some estimates suggest that 20–25% of all CO_2 emissions from lime are reabsorbed, but reabsorption typically occurs off-site and thus should not be included in GHG LCAs (National Lime Association, 2005).

Transportation: Upstream emissions due to transportation may come from transportation of inputs and transportation of labor during agricultural production. Other sections address GHG emissions associated with the production of various agricultural inputs. Most of the studies cited do not address emissions associated with bringing these inputs to the farm specifically, and depending on the study, these emissions may or may not be included. Given that most producers will not know the location of production of the various agricultural inputs used, it would be difficult to accurately account for these emissions. Of course, these emissions will vary greatly depending on the distance from the point of production to the point of use. One study assumed transportation distances for railroad and truck to be 800 and 160 km respectively (Mudahar *et al.*, 1982). Section 3.b.(iv) below on organic amendments shows that, at least in case of that input,

transportation related emissions are a small percent of total emissions. Since these emissions may already be included in some estimates in the literature of GHGs embodied in inputs, and these in general do not seem to be significant, we do not treat them separately in this document, and the CMPP could choose to ignore upstream transportation-related emissions as a separate category.

Comprehensive Calculation Procedures for Chemical Production and Use: There are several examples of efforts to create a comprehensive and accurate characterization of GHG emissions from agricultural chemical production including the International Sustainability and Carbon Certification for biofuels in the European Union and California's Low Carbon Fuel Standard (LCFS). These approaches could be adapted for use in the Reserve protocol as an alternative option to using emission factors noted above for each type of input. For gathering on-site GHG emissions data, ISCC requires collecting information on a) amount of main product and by-product, b) the amount of chemicals used, c) amount of P_2O_5, K_2O, CaO, and N-fertilizer, d) diesel and electricity consumption, e) thermal energy consumption, f) source of energy for processing, and g) amount and use of by-by-products. They also allow information to be collected from the literature including, a) heating values of main product and by-products, b) emissions factors, c) N_2O emissions. The LCFS uses the GREET model to calculate GHG emissions associated with these inputs based on this information, which would be an appropriate approach for the CMPP to adopt. The GREET model does not take into account GHG emissions associated with the transport of agricultural inputs to the field.

3.b.(ii). Irrigation Water Pumping and Transport from Off-Site Sources

Of all irrigation in the U.S., 30%, by area, is from off-site surface water, while the rest is from on-site sources described above. In a similar vein to the discussion about on-site irrigation, project activities could require an increase in irrigation water, which would create an increase in GHG emissions. The general analysis of this issue is nearly identical to that of on-site irrigation water. The differences are the in scope of additional emissions from the point of view of the offset developer, the average carbon intensity of delivered water, and potential methods for calculation of leakage.

As mentioned in the on-site irrigation section, only a small percentage of basic grain production in the U.S. is irrigated, and even less is from off-site sources. However, when off-site irrigation water is used, it can have a large carbon impact, so it is important not to ignore this leakage category. West and Marland (2002) estimated that, on average in the U.S., off-site pumping and transmission of surface water emits 176.7 kg C/acre, or nearly twice the emissions of on-site irrigation. However, there should be considerable regional and local variation in this figure, so average numbers should not be used for calculating leakage associated with off-site pumping. While these emissions are off-site emissions from the point of view of the project, they would clearly count as carbon leakage and should be considered within the GHG Assessment Boundary.

Usage of irrigation can vary dramatically by region as well as by year, which could make accurate calculation difficult and the national average figure quoted above virtually unusable for the CMPP. For example, more than 78 percent of the corn acres harvested in Nebraska required irrigation in 1996, while only 5 percent of the corn acreage was irrigated in Iowa and no corn acreage was irrigated in Indiana, Illinois, and Wisconsin (Shapouri *et al.*, 2002). In addition, the amount of water used for irrigation in Nebraska in 1996 was one third of what was used in 1991.

Greenhouse gases associated with off-site pumping can be determined based on the following factors: (1) The amount of water used, measured in acre-feet or acre-inches per acre. Default values for average gross seasonal water application depth can be obtained by state from the USDA or the amount of irrigation water used can be determined directly with flow meters or other devices. (2) The vertical elevation difference between the source of water pumped and pumping plant discharge point is an important determinant of energy used to deliver water. (3) The pressure, measured in psi, of delivered irrigation water also influences the amount of energy used in water delivery (0-25 psi for most surface applications, 26-59 psi for drip and some sprinkler systems, >60 for other sprinkler systems (USDA-NRCS, 2011). (4) Finally, differences in carbon intensity of electricity used to move pump water plays a role in overall GHG emissions pertaining to irrigation.

These variables have been organized into a simple model by the USDA Natural Resource Conservation Service (USDA-NRCS, 2011), which provides an estimated cost of energy for irrigation as an output. The model requires the input of location, energy type, energy cost, vertical elevation difference, water usage (which can be state average defaults by crop), and pressure at the site of irrigation. Using the model's output of total cost of energy and the cost per unit of energy inputted by the user, a total energy usage can be derived. While this will still require some data collection on the off-site irrigation system, this may prove to be the most straightforward method available to estimate irrigation GHGs, although not perfectly accurate since this model makes some necessary assumptions. Ideally, the local irrigation district would have an emission factor for water computed, which would provide a more accurate and simple method to estimate GHGs from irrigation.

3.b.(iii). Transport of Labor

Another leakage concern associated with upstream transportation is increased GHG emissions due to transporting labor to the farm in order to implement project activities. Accurate accounting of this would require data on the number of additional miles traveled by employees to implement project activities. A U.S. or state average fleet fuel efficiency could be used to determine GHG in this category. We were unable to locate published studies that clearly accounted for the transportation of labor to and from the farm, even though some studies (for example (Hill *et al.*, 2006) include energy use in laborers' households in farm LCAs (which we consider to be outside the potential GHG boundary). However an EcoShift GHG inventory of a California farm showed that transportation of labor was 12.1% of the inventory, which included machinery, irrigation, compost production, and labor transportation. These emissions will be an issue in any project activity that includes an increase in labor, such as cover cropping, removal of summer fallow, and potentially diversifying crop rotations, inclusion of perennials in crop rotations, and switching from annuals to perennials. If the Reserve chooses to include these emissions in the GHG Boundary, it is relatively straightforward to calculate, simply by gathering data on additional labor days, miles travelled to the farm, and miles per gallon of vehicles driven.

3.b.(iv). Soil Amendments: Organic Management and Compost

Compost Production: Preparation of organic amendments, especially compost, creates additional GHG emissions. These emissions can come from two sources: GHG emissions from machinery used to transport and produce compost and biogenic or fugitive GHG emissions from anaerobic pockets within piles that release methane and denitrification processes that release nitrous oxide. A third category, often determined to be outside the project boundary, is capital equipment and infrastructure needed to process compost. This final category has not been addressed in any detail in the studies reviewed for this paper.

A study by the Recycled Organics Unit in New South Whales, Australia cited a range of emissions of 49.8-52.6 kg C/ton of wet feedstock in three studies resulting from compost production (Riffaldi *et al.*, 1986, Jakobsen, 1994, Jackson, 1997, Recycled Organics Unit, 2001), while a California Air Resources Board study found a value of 32 kg C/ton of wet feedstock. Details of aspects of production that led to these calculations follow, and Table 10 above summarizes some of the numerical data.

Machinery Emissions: Machinery use in windrow compost production includes the following, although not all facilities will use all items listed: (a) equipment to transport compostable material from field to composting site and from the composting site to the end user, (b) machinery to agitate or turn compost so that aerobic processes occur, (c) machinery used to force air into compost piles, rather than aerating through agitation, (d) shredding, grinding or screening equipment to prepare material for composting, and (e) equipment used to water compost piles. Windrows or static piles with aeration through agitation or forced air are the main methods of compost production.

A California Air Resources Board study (California Air Resources Board, 2010) estimated transportation related emissions from compost, including feedstock collection and compost delivery, at an average of 75.7 miles, which, using an emissions factor of 0.028 kg C/ton mile, results in 0.60 kg C/ton of feedstock. Two other studies (Blengini, 2008, Martínez-Blanco *et al.*, 2009), conducted in Europe, reported much lower average distances for collection of feedstock, an average of 9 and 16 miles compared to 47.5 miles in the CARB study, and other studies

exclude collection due to lack of data or a tighter project boundary (Recycled Organics Unit, 2001). While the CARB study did not include application and the ROU study did not include collection of feedstock, the total amount allocated to transportation, as a percent of the total, was not too different. On the other hand, the EPA WARM model (EPA, 2011a) concludes that transportation-related emissions only contribute 0.1 kg C / ton of feedstock.

For machinery used to process compost, the CARB study found 2.2 kg C/ton of feedstock, while the ROU study found 4.8 kg C/ton and the EPA WARM model found 5.5 kg C/ton. Finally, the ROU study included emissions resulting from mechanical application of compost at 0.94 kg C/ton, while the CARB study and the EPA WARM model do not include mechanical application. This final area would be covered by the emissions and methods described in the machinery section above.

Biogenic Emissions: Biogenic (or fugitive) emissions from decomposing compost during compost production are by far the most important category of emissions in this process. These emissions were 87% (CARB) and 89% (ROU) of total compost production emissions. Although these are GHG emissions from organic matter decomposition that are upstream of the project, these emissions may have occurred under baseline conditions as well. For biogenic sources, the CARB study found 21.3 kg C/ton for methane emissions and 6.8 kg C/ton for nitrous oxide emissions (for a total of 28.1 kg C/ton), while the ROU study found 51.6 kg C/ton of wet feedstock. The EPA WARM model did not include biogenic sources.

Overall, the CARB study found that using compost can result in an overall savings of up to 147 kg C/ton feedstock, due to increased soil carbon storage, decreased water use, decreased soil erosion, decreased fertilizer use, and decreased herbicide use. On the other hand, leakage that occurs through the production of compost totals 32 kg C/ton feedstock from transportation, processing, and biogenic N_2O and CH_4, suggesting about 22% leakage (see Section 2.a.(iv) and Table 10 for detail).

3.b.(v). Seed and Seedling Production

Seed and seedling production is a source of GHG emissions, since agriculture relies on purchased seed, and it takes energy to grow, package, and ship seeds to the farm. Depending on the crop, seed production and seeding can be a considerable source of GHG emissions (alfalfa, Gelfand *et al.*, 2010) or a negligible source (corn-soybean, Landis *et al.*, 2007). Specifically, at the recommended 4.5-6 kg/acre seeding rate for alfalfa (Rankin, 2008), and taking into account GHG emissions of 2.63 kg C/kg seed for alfalfa (West *et al.*, 2002a), the emissions associated with this seed will be between 11.8 and 15.8 kg C/acre. In contrast, the recommended wheat seeding rate is around 45 kg/acre, and at the estimated impact of 0.11 kg C/kg seed for winter wheat (West *et al.*, 2002a) the impact of the seed in the baseline is 4.95 kg C/acre. For corn and soy, the range per acre is 7.8 - 8.7 kg C/acre (West *et al.*, 2002a).

The overall impact for agricultural operations will depend not only on the type of seed used, as well as planting density, but also on the location of production, since the energy mix at origin will vary with geographical location, and shipping distances. Seedling production is specific to vegetable crops, such as tomatoes. Due to high energetic requirements of greenhouse operation, seedling production can constitute a large GHG source (Lagerberg *et al.*, 1999), but these will vary significantly with type of crop and growing conditions.

3c. Off-Site Downstream Emissions

3.c.(i). Leaching and Run-Off from Nitrogen Application

Applications of nitrogen-based agrochemicals or manures can lead to leaching of nitrate into the surrounding system if applied in quantities beyond optimal for plant uptake. Although nitrate by itself does not constitute a greenhouse gas, leaching of nitrate from the upper soil can lead to indirect emissions of NOx and NH_3. Sources of indirect N_2O emissions from croplands include leaching and runoff of nitrogen applied to agricultural soils and formations of N_2O from NH_3 emissions. Project activities that increase the application of nitrogen-rich fertilizers will also increase these off-site downstream nitrogen emissions.

Nitrate leaching is an issue in agricultural production systems including organic (Kirchmann *et al.*, 2001), conventional (Goulding, 2000), and pasture systems (Di *et al.*, 2002). The extent of

leaching varies with best management practices, type of crop or cattle production system, precipitation, soil characteristics, and other factors. Nitrate losses to volatilization, denitrification, and leaching are affected by the cropping systems' nitrogen use efficiency. Encouraging management techniques that ensure optimal nitrogen uptake and maximum nitrogen use efficiency typically attempt to improve synchrony with crop N demand and N supply.

In 2005 the IPCC adopted a methodology for estimating indirect N_2O emissions from nitrate leaching and ammonia volatilization. Indirect N_2O emissions have been shown to be 29% of overall N_2O emissions, but also has the greatest associated uncertainty (126%) of all sources of N_2O emissions (Brown *et al.*, 2001). Indirect N_2O emissions equals the nitrogen amount in the source multiplied by an emissions factor per kg N and a factor for converting N_2O -N into N_2O (44/28). The IPCC default value for nitrous oxide emission factor is 0.025 kg N_2O -N per kg N leached (Van der Hoek *et al.*, 2007). This equation is modified by including a fraction of nitrogen leached into the ground water to estimate emissions from leaching and runoff. The IPCC default value for this fraction is 0.30 (Brown *et al.*, 2001). Increased nitrate loading in streams and estuaries also significantly affects aquatic environments, creating temporary eutrophic conditions, which is a negative environmental impact and has the potential of altering nutrient and carbon dynamics in aquatic ecosystems.

3.c.(ii). Transportation Emissions (Post Farm Gate)

Some proposed project activities may result in increases or decreases in the distribution of agricultural products. In the cases of crop diversification or switching to combinations of perennial crops, the resulting crops could be transported much greater distances than original crops. In the case of elimination of summer fallow, additional crops will be transported. It is important to consider whether or not transportation of crops beyond the farm gate should be included within the boundary of the agricultural GHG emissions for this protocol. On the one hand, the Reserve may conclude that it is either too complex or not necessary to account for emissions beyond the farm gate. On the other hand, any project activity that results in either an increase or decrease in agricultural production will clearly cause a proportional increase or decrease in GHG emissions associated with transporting agricultural goods.

An important consideration here is whether or not the Reserve believes that agricultural production is entirely a function of demand. If it is, any increase in production will be accompanied by a decrease in production elsewhere. However, while demand does drive production, there are numerous additional factors that could influence production, the most important of which are subsidies, which provide various incentives for agricultural production over and above actual demand. In fact, the potential to sell agricultural offsets could be considered one such subsidy. In addition, just because increased production may be accompanied by decreased production elsewhere, without knowing the location of that production, it is impossible to know whether transportation emissions will cancel out. Given this, the Reserve may decide that it is untenable to assume that changes in downstream transportation will be neutral. In this case, the total GHG impact of an increase or decrease in production associated with project activities should include the downstream transportation phase of the agricultural life cycle.

In fact, these emissions can be a significant proportion of the overall agricultural cycle. For example, Meisterling *et al.* (2009) show that lifecycle GHGs of wheat, excluding downstream transportation, are 190g CO_2e/ kg conventional wheat flour and 160g CO_2e/ kg organic wheat flour (Figure 3). Although the change in management (conventional to organic) results in a reduction of GHG emissions, the downstream transportation change can potentially more than offset this difference. The GHG savings in this example are equivalent to the emissions associated with transporting the product 420km. If the distance is increased to 2000 km for the resulting product, this change will correspond to an increase in emissions of 110g CO_2e/ kg wheat flour, assuming 50/50 truck/rail transport mode (Figure 3). This study is based solely on wheat production, but the overall message is that downstream transportation can be a significant part of the overall GHG footprint increase if distribution patterns change.

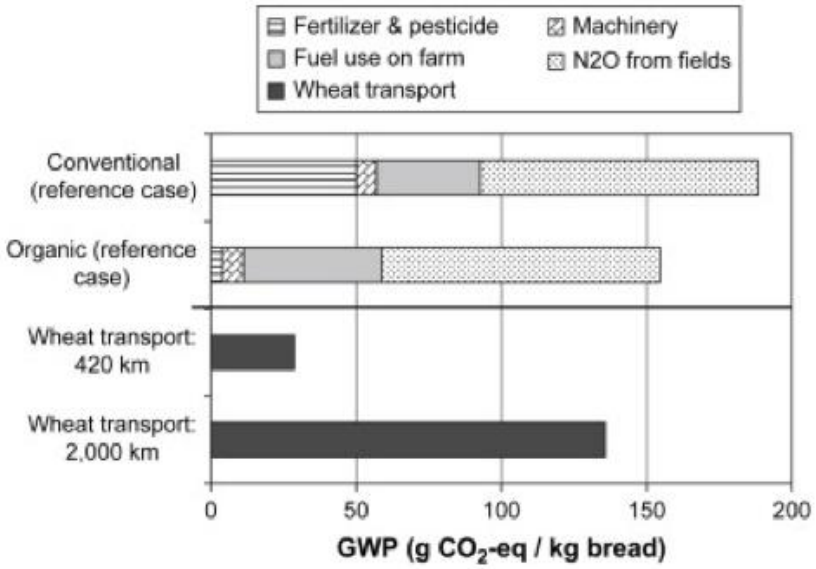

Figure 3. GHG Emissions from Wheat Agriculture Compared to GHG Emissions from Transporting Wheat. (Meisterling et al., 2009)

Accurately accounting for increases or decrease in transportation emissions of agricultural goods would involve data on changes in total yield and computation of GHGs per ton of transportation of the baseline and the project crop. Assigning carbon values to transportation requires assuming a fuel use rate and transportation distance. The fuel use rate depends on the mode of transportation. For example railroad transportation is approximately 0.7 MJ/Mg km, while trucking is more than 1.4 MJ/Mg km (Borjesson 1996 cited in West *et al.* 2002). In addition, the Reserve would have to determine the boundary of transportation emissions, which could be from farm gate to distribution center, point of sale or point of consumption. The broader the boundary, the more accurate but less feasible the calculation will be. For any boundary, the offset developer would need to gather information on the distance from the farm to the determined boundary. The fuel use rates mentioned above could be used or developers could derive their own using information on the transportation mode and average cargo load.

3.c.(iii). Indirect Land Use Change

A key feature of agriculture that makes carbon accounting susceptible to leakage is a fixed land base (the total land in crop production will remain the same) and the possibility that economic behavior (crop production) can shift to meet market demand for agricultural goods (Murray *et*

al., 2007). Any land use or management change that impacts crop yields can have spillover effects onto other parcels where land is brought into production to meet demand for agricultural goods, an effect referred to as indirect land use change (ILUC). ILUC is an important consideration in developing verifiable offsets because it can cause GHG leakage off-site if a crop is simply grown elsewhere to satisfy market demand. By retiring land, altering yields or switching to a different crop variety on the same lands, new cropland, grassland or forest could be brought into production elsewhere.

ILUC differs from direct land use change (DLUC), which occurs when a non-cropland use is brought into crop production or vice versa as part of a project activity. If, for example, a project activity takes agricultural land out of production, that is a carbon-emission reduction from DLUC. For the reduction in GHG emissions to be real, verification that related ILUC does not negate these carbon emission reductions is important. When the risks of this are significant, it is important to include GHG losses due to ILUC in the GHG Assessment Boundary. Risks are significant when the crop is in high demand or fetches high market values. The main concern for the CMPP is whether decreases or cessation of production will lead to significant levels of ILUC. Some emission factors for various land types are in the following table, however, we note that a project activity could result in any percentage, from 0 to 100%, of these values depending on estimates of ILUC.

	Low	High	Units	Ecosystem Conversion Fractions
Forest	38,700	71,700	kg C/acre	0 – 100%
Grassland	8,280	24,500	kg C/acre	0 – 100%
Peatland	110,400	330,800	kg C/acre	0 – 100%

Table 21. Emissions Factors for Clearing Various Land Types in ILUC Models. (Derived from Plevin et al. 2010)

ILUC is not possible to directly measure, but instead relies on a combination of land use change and economic models. Indirect land use change models have been incorporated into life cycle analyses used in California's Low Carbon Fuel Standard (LCFS) for liquid fuels production as well as the U.S. EPA Renewable Fuel Standard (RFS). Hence, there is precedence for attributing

carbon intensity values in a carbon market framework. However, there has been no consensus on preferred models for carbon intensity values. The CARB LCFS uses a model called GTAP that creates "add on" carbon intensity values, while the EPA does a consequential analysis—an analysis that considers how decisions affect levels of production or other impacts outside the system or relative to other scenarios—based on a model developed by Winrock International.

ILUC modeling is based on economic price signals. It is particularly challenging to accurately assess ILUC since the scope of commodity markets in agriculture is quite broad, extending beyond the regional into national and global markets. Popular equilibrium models include FASOM, FAPRI, GTAP, and MIRAGE, but each produces varied results. Prins *et al.* (2010) reviews 14 models and finds that variations are largely driven by (1) different ways that area and intensification changes are incorporated, and (2) how demand for products interact with agricultural area and influences the balance between expansion, intensification, and consumption.

Equilibrium models are broadly characterized as either partial or general models. Partial equilibrium models focus on sectors of interest (agriculture, mining, etc.) while general equilibrium models incorporate on all sectors of the economy. The models estimate how changes in one sector affect prices in other sectors, and use this information to estimate indirect land use change. While agriculture may impact only a few sectors, meaning that partial equilibrium models may suffice, a key distinction is that most partial equilibrium models do not include a market for land (Kretschmer *et al.*, 2010). Since markets for competing land uses are critical for agriculture and indirect land use change, particularly when land is taken out of production, partial equilibrium models are significantly limited.

While some of these models are used to estimate ILUC from biofuels production, these models can be modified for agricultural production more generally and answer, (1) how much land will be brought into production to compensate for land taken out of production, and (2) the spatial proximity of land taken out of production to lands replacing that production (Plevin *et al.*, 2010). These models pose particular challenges because to the extent that they are verified with real data, they rely on historical models of land use change. The problem is that historical drivers of

land use change including subsistence agriculture, beef production, and local food production, are less relevant in the context of a globalized food system where transitions to large scale production are current drivers of land use change particularly in Brazil and Indonesia (Plevin *et al.*, 2010). ILUC emission estimates for crop used to grow biofuels are highly uncertain, but may be much greater than previously estimated (Plevin *et al.*, 2010).

Parameters have been developed for ILUC in the peer reviewed academic literature. In addition, some are explored in greater detail by an expert working group on ILUC for California's Low Carbon Fuel Standard. Values for some of the following parameters evaluated to estimate ILUC in the various models can be found in the literature cited below. These parameters include:

- *Yield per Unit Land*: The amount of product yielded per unit land is important because not all land produces the same yields.
- *Net Displacement Factor*: This is the amount of land brought into production to replace decreased production elsewhere (Searchinger *et al.*, 2008, Hertel *et al.*, 2010, Plevin *et al.*, 2010).
- *Ecosystem Conversion Fractions*: These are fractions of GHG emissions from converting cropping systems, land cover type or region (e.g., percent forest, grassland, wetland, etc.).
- *Land Conversion CO_2 Emission Factors*: The GHG emissions associated with converting specific ecosystems into production (Searchinger *et al.*, 2008, Hertel *et al.*, 2010, Plevin *et al.*, 2010).
- *Production Period*: This is used for biofuel production and is reported in a number of years, since the emissions must be assigned on a per unit-fuel basis. It is usually amortized over 30 years, which is the standard that has been adopted by CARB and the U.S. EPA (Searchinger *et al.*, 2008, Hertel *et al.*, 2010, Plevin *et al.*, 2010).

Carbon emissions from land use change require accurate estimates for above and below ground biomass either before and after or as a function of conversion to cropping practice. One challenge in deriving values for the parameters listed above is estimating accurate emissions factors for particular ecosystems and agroecosystems. The carbon stocks in many places have not been systematically reviewed. IPCC has default values for general land cover types but it has

been recommended that better estimates be developed (Yeh *et al.*, 2010). The Woods Hole Research Center has data for biomass carbon and soil carbon stock values, which is the preferred model for the CARB-LCFS GTAP model. It divides global data into 31 ecosystem types in ten world regions. Each region is broken into ecosystem types. For example, lands available for conversion in the U.S. is comprised of broadleaf forest (2%), mixed forest (34%), coniferous Pacific (2%), and grassland (62%). Crop yield maps have been created that can estimate regionally-specific biomass estimates for different crop types which have been recommended by (Yeh *et al.*, 2010).

One estimate of potential ILUC leakage is <10% for switching from conventional to conservation or no-till management (Murray *et al.*, 2007). An estimate of leakage for converting cropland to uncultivated use is 20% (Wu, 2000 denominated in land, not carbon), but is suggested to be an underestimate by Murray *et al.* (2007). Because retired lands are usually the most marginally productive, project activities might not have much of an effect on commodity supplies. But as Murray *et al.* (2004) found, project activities can lead to leakage of 90% in forest set-aside programs.

GHG leakage from ILUC change is ultimately an economic question. How commodity supplies and subsequently prices change can effect grower decisions to bring other land into to production. ILUC estimates of leakage from proposed project activities are possible using the models mentioned above and data aggregated at the regional or national level to account for a system that captures economic activities such as grower responses to prices. But it is also important to note that many factors drive commodity prices and therefore grower decisions as well. The lack of verified data to assess ILUC in agriculture suggests that existing ILUC models require further development and refinement. Currently, the LCFS standards are in the final stages of development, and the Reserve can utilize the considerable amount of expertise that has been devoted to these models. However, in the cases where there is a potential for ILUC leakage risk, the Reserve may chose to adopt a methodology that takes into account GHG intensity of the crops, rather than modeling projections of ILUC.

ILUC risk is associated with decreased yields due to project activities. As discussed above, decreased yield may result in indirect land use change since production must be replaced elsewhere. Instead of using LCFS methodology for ILUC impacts, the Reserve can discount the value of the total offset by the corresponding reduction in yield to account for leakage. Murray and Baker (2011) suggest that, in order to avoid the complications arising from yield changes that accompany GHG mitigation projects in agriculture, GHG reduction calculations should be yield-based, rather than area-based. Throughout this paper we have reported GHG emissions in kg C or t C/acre, which is the traditional way of accounting for GHG fluxes from agricultural and other activity. However, it may be appropriate, in the cases where a project activity results in a yield loss, and therefore constitutes a potential ILUC leakage risk, to calculate a discount factor based on changes in GHG intensity (i.e., t C/t [product]), and somehow also encourage projects with the highest reduction in GHG intensity.

A GHG intensity approach, while simpler than the ILUC modeling approach, may potentially over-estimate leakage effects (both positive[2] and negative) by assuming that a one-for-one increase/decrease in yield occurs somewhere else to compensate for yield losses/gains from a project. In reality, the change in yield on other croplands will frequently be smaller than the change in yield within the project area. An intensity approach may therefore be conservative when estimating leakage from projects that reduce yields, but will tend to overestimate the GHG reductions associated with projects that increase yields.

4. Summary

All proposed project activities for the CMPP have leakage risks except irrigation improvements. Accounting for on-site GHG emissions is usually easier than off-site GHG emissions simply due to availability of records. In addition, there is significant uncertainty for many types of leakage, including, perhaps most importantly, CH_4 and N_2O emissions from additions of organic materials, conversion to pasture, as well as changes in cropping regimes and tillage. This is critical because these gases have much higher global warming potentials than CO_2, and small

[2] Positive leakage refers to when a project causes emission reductions somewhere else, in this case, by producing more goods and reducing agricultural production somewhere else. The Reserve generally would not count this type of indirect emission reduction in the GHG reduction calculation.

increases in emissions of these gases, especially N_2O, can reduce gains in soil carbon storage. Magnitudes of leakage in these categories will depend on soil type, cropping system, moisture regime, original soil carbon amount, and other factors. While difficult to measure and uncertain, these emissions are critical to include in GHG calculations.

In addition, quantifying some types of carbon storage could require potentially cost prohibitive monitoring. For example, if the Reserve decides to include tillage reduction measures as a project activity, soil carbon gains from reduced or no-till practices may need to be monitored for 10–15 years at depth of up to one meter, depending on soil type, to ensure that additional carbon storage is actually occurring, and that carbon is not merely being redistributed within the soil profile. However, in the majority of cases, especially for estimating changes in soil carbon and nitrogen fluxes, modeling approaches, such as the DAYCENT or DNDC model, constitute sufficiently accurate, cost effective accounting methods. Project activities that potentially affect ILUC can also draw upon existing models, such as the LCFS methodology, but come with significant uncertainties, and the alternative GHG intensity approach may be more readily applied.

The key message from this review paper is that although some practices are considered positive changes for soil carbon storage, there are many issues in determining a net reduction of GHG emissions. Our analysis suggests that, due to the high potential for leakage, it would be inappropriate to use the physical project boundary as the GHG accounting boundary. Rather, drawing a broad boundary that includes most leakage types is critical to ensure high quality offsets. Although uncertainties may make calculation difficult, the only leakage types that are unproblematic to ignore are those with a potentially minimal contribution to the total.

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