

Climate Friendly Farming™ Project Summary

In 2007 the International Panel on Climate Change (IPCC) concluded that human actions are very likely the main driver for the increasing global temperatures, rising seas, and shifting weather patterns known as climate change. With changes already being felt, the effort to minimize future climate change has become more urgent. Climate change has the potential to negatively impact many of our natural resources and natural resource-based industries, including agriculture, which we depend on to feed the world's population and provide a variety of environmental benefits.

Agriculture is both a source and a sink for several of the most important greenhouse gases (GHGs) involved in climate change, including methane (CH₄), nitrous oxide (N₂O) and carbon dioxide (CO₂). Direct greenhouse gas emissions from agriculture are significant, accounting for an estimated 10-12% of total global anthropogenic emissions (IPCC, 2007). However, this figure excludes several “indirect” greenhouse gas emissions that are generated in support of agricultural production but accounted for in other sectors, such as agricultural fuel use, emissions from agrochemical and fertilizer production, and emissions from land use changes to produce agricultural products. If these effects are included, it has been estimated that agriculture accounts for 17-32% of total annual global anthropogenic greenhouse gas emissions (Bellarby et al., 2008).¹ The World Bank (2007) similarly concludes that agriculture accounts for 26-35% of worldwide GHG emissions.²

Reducing emissions and enhancing carbon sequestration in the agricultural sector can help mitigate GHG emissions. In addition, biomass energy products derived from agricultural and organic wastes have the potential to replace some fossil fuel use and thus reduce emissions from other economic sectors.

The Climate Friendly Farming™ (CFF) Project was established to provide research-based information to support agricultural climate mitigation policies and the deployment of “climate-friendly” agricultural practices and technologies. The project was intentionally cross-disciplinary and multi-faceted, combining field research, technology research and development, computer modeling and economic and policy analysis. Funded by the Paul G. Allen Family Foundation along with

¹ Agriculturally-related electricity and fuel use (included in the buildings and transport sector) is estimated to account for an additional 0.2-1.8%, emissions related to the production of agrochemicals and fertilizer (included in the industry sector) an additional 0.6-1.2%, and effects resulting from land use changes as additional land is cleared for agriculture (included in land use changes) an additional 6-17% (Bellarby et al., 2008).

² This figure includes 15% accounted for directly in the agricultural sector according to emissions inventories that governments have submitted to the United Nations Framework Convention on Climate Change, plus emissions from deforestation in developing countries (agriculture is the leading cause of deforestation) (World Bank, 2007). This calculation does not include emissions associated with energy use or production of agricultural inputs.

government and industry grants and contracts, we assessed the greenhouse gas emissions occurring within three of the most common agricultural systems of Washington State, developed concrete strategies for reducing the climate change impacts of agriculture, and provided guidance about how these results might apply to current and future policy efforts.

Further background about agricultural emissions, mitigation options, and the Climate Friendly Farming Project is reviewed in the [Project Overview and Context](#). The most important lessons learned from the CFF Project are summarized below, with further detail provided in separate chapters.

A variety of technologies or management strategies have been developed and implemented in farming systems in the PNW through the Climate Friendly Farming™ Project. Our analysis shows that some of these strategies have more potential than others to reduce the overall greenhouse gas emissions of agricultural systems.

The Climate Friendly Farming Project incorporated work on a variety of potential strategies for mitigating greenhouse gas emissions in three agricultural systems (dairy, dryland, and irrigated) of importance to our region and the world. Specific strategies we investigated included anaerobic digestion, precision nitrogen technologies, conservation tillage, and increasing crop residue returns to the soil. This research illustrated the mitigation potential of agricultural practices in semi-arid areas such as the inland Pacific Northwest (PNW), and thus complemented previous research carried out in the Midwest.

In addition to field studies, the CFF Project developed economic, modeling and decision-support tools for evaluating various GHG mitigation strategies. We applied these tools in the assessment of PNW agriculture, but the tools are useful for assessments in other geographic regions as well:

- [CropSyst](#), a cropping systems simulation model that has been tested extensively in the PNW and in many other regions of the world (Stöckle et al., 1994, 2003);
- [C-Farm](#), a simplified version of CropSyst that is appropriate for working directly with industry, policy-makers and extension personnel (Kemanian and Stöckle, 2010);
- [Linked Excel farm-level budgets](#) that show the economic impact of reduced tillage and possible carbon credits;
- [General Integrated Solid Waste Co-digestion \(GISCOD\) model](#), a tool designed to provide performance expectations for digestion of mixed organic wastes (Zaher et al., 2009); and
- A new [decision support tool for precision N management](#), designed for evaluating and formulating N fertilizer recommendations for dryland winter wheat, but with potential to be adapted to a wide range of crops (Huggins et al., in review).

Anaerobic Digestion

Many larger dairies use a lagoon-based system for livestock waste management, which produces prodigious amounts of methane. On these farms, anaerobic digestion, prior to lagoon storage or direct field application, has the potential to reduce GHG emissions through a variety of mechanisms:

- Reductions in methane emissions that manure management would otherwise generate;
- Generation of renewable energy that can be used for combined heat and power (CHP) or further refined to pipeline quality natural gas or vehicle fuel, replacing current fossil fuels;
- Additional potential emission reductions via co-digestion of food processing wastes that would otherwise release methane under uncontrolled landfill conditions; and
- Substitution of digester byproducts (e.g., fiber, bio-fertilizers) for GHG-intensive products such as peat moss and chemical fertilizers.

Carbon credits give some indication of the potential GHG benefits of AD, though they do not capture all of the potential life-cycle greenhouse gas impacts. By combining existing carbon credit protocols for digester and CHP offset credits with a hypothetical protocol for co-digestion of organic wastes, developed using our research and landfill emissions assumptions from Murphy and McKeough (2004), we estimate potential GHG benefits of over 1 MMT CO₂e per year if AD is installed on 40 large dairies in Washington State, representing a third of the CAFO dairy cow population (Table 1).³

³ MT = metric tons (1 MT = 1 Mg); MMT = million metric tons (1 MMT = 1 Tg). CAFO stands for concentrated animal feeding operation, and for purposes of our analysis, included all dairies with more than 500 animals.

Table 1. Generalization of possible AD greenhouse gas credits, including hypothetical carbon credit for co-digestion of organic wastes, assuming installation of AD on 40 large dairies in Washington State^a

| | Manure Credit | OFMSW Credit ^c | | Electricity Offset ^d | Total | |
|---|--|--|-----------------------------------|--|-----------------------------|--------------------------|
| | | Additional Manure Credit from Improved Digestion | Organic Waste Digestion Credit | | MT CO ₂ e/cow yr | MMT CO ₂ e/yr |
| | MT CO ₂ e/cow yr ^b | MTCO ₂ e/cow yr | MT CO ₂ e/wet t manure | MT CO ₂ e/cow yr ^e | MT CO ₂ e/cow yr | MMT CO ₂ e/yr |
| Manure Only | 4.89 | NA | NA | 0.68 | 5.57 | 0.39 |
| Co-Digestion of Manure and Food Processing Wastes | 4.89 | 8.73 | 0.85 | 1.62 | 15.24 | 1.07 |

^a These calculations depend on the following assumptions: (1) all co-digestion on AD farms occurs at a volumetric flow rate of 15-20% and consists of food processing waste with average biogas production capabilities determined by us; (2) of Washington State's 135 CAFO dairies larger than 500 cows, representing 192,000 WEC (Wet Cow Equivalents), 40 of these can have AD installed; and (3) of those 40 installations, all are scrape facilities and, ten each will be on 1,000, 1,500, 2,000 and 2,5000 WEC farms, totaling 70,000 WEC or a little over a 1/3 of the CAFO population in the State.

^b (US-EPA, 2008)

^c OFMSW = Organic fraction of municipal solid waste, Credit for reductions in emissions from co-digestion of food processing wastes were calculated using landfill assumptions from (Murphy and McKeogh, 2004)

^d Electricity offset includes GHG benefit from replacing electricity generated from assumed baseline coal-powered plant with electricity generated from AD digester.

^e 0.4 MT CO₂e/MWh less 6% parasitic load and assumed 90% runtime (CCX, 2008)

The calculations above do not include the GHG benefits of byproducts from the AD biorefinery (there are no existing carbon credit protocols for these products), but additional emissions reductions could also be achieved. Using figures from our research, existing analyses of the GHG impact of the Canadian peat industry (Cleary et al., 2005; Waddington et al., 2009), and relatively conservative assumptions about marketing potential, we estimate that the potential offset of peat substitute sales from the 40 dairies described above represents roughly 19,000 MT CO₂e annually.⁴

⁴ In 2000, the Canadian peat industry mined and sold 1.3 MMT of peat, mostly to the nursery industry, emitting an estimated 0.89 MMT CO₂e in the process or 0.685 MT CO₂e/harvested ton (including end-use peat decomposition, land use changes, and fossil fuel combustion) (Cleary et al., 2005; Waddington et al., 2009). We assumed a manure production rate of 9.7 m³ fiber/cow year from the 40 dairies described above in Table 1, with 40% of AD fiber marketed as peat replacement. This would generate 28,280 MT peat replacement per year. Our offset calculations also assume that substitution for peat is achieved on a one-to-one basis, without impacting total consumption.

Biofertilizer by-products could also generate emissions reductions, though the impact would depend on the formulation of the products, which are still in development. From a calculation of the potential impact (Table 2), it is evident that the GHG impacts, while present, are not as large as those from manure digestion or co-digestion of organic solid wastes.

Table 2. Estimated hypothetical greenhouse gas credits for recovering nutrients from anaerobic digestion (AD), assuming installation of AD on 40 large dairies in Washington State.^a

| | Nutrient Product MT-P or N/yr ^{b, c} | GWP MT CO ₂ e/MT-P or N ^{d, e} | Total Offset MT CO ₂ e/yr |
|-----------------|--|---|---|
| Bio-phosphorous | 500 | 6.11 | 3,100 |
| Bio-nitrogen | 5,353 | 2.638 | 14,000 |
| Total | 5,853 | ---- | 17,100 |

^a Assumptions about dairy sizes are as in Table 1.

^b Assume 2.65 kg total ammonia nitrogen/m³ manure (Frear et al, 2009), 0.12 m³ manure/cow day, and 80% conversion to ammonia product

^c Assume 0.048 lbs P/cow day and 0.4 kg P/MT food waste (Frear et al, 2009) and an 80% P recovery

^d GWP = global warming potential, a measure of the total contribution to global warming that compares all other gases to that of the same mass of CO₂ (whose GWP is equal to 1). Following the IPCC recommendations, we calculate GWP using the contribution over 100 years.

^e (Davis and Haglund, 1999)

Anaerobic digestion has the potential to make a sizeable contribution to agricultural GHG mitigation in Washington State, across the U.S. and elsewhere in the world. As detailed in the calculations above, we estimate that anaerobic digesters installed on 40 Washington dairies totaling 70,000 wet cow equivalents (WEC) could provide GHG mitigation of roughly 1.1 MMT CO₂e/yr. Moreover, with advancements in policy and continued research to improve the process and economics, as was done in the CFF project, AD will likely represent a win-win strategy that will improve nutrient management, dairy economics, and reduce landfill methane emissions in addition to mitigating agricultural GHG emissions.

Precision Nitrogen Management

Precision N technologies have the potential to improve the efficiency by which crops use nitrogen inputs, thereby reducing potential N losses as nitrous oxide, a potent GHG. These technologies use geographic positioning, variable rate application and sensors to vary the timing and rate of N applications, thereby tailoring N management to seasonal crop demands and environmental conditions. Most of the technologies necessary for implementing precision N have been available, but the CFF Project provided the first rigorous field-scale tests for dryland wheat production in the PNW. Research we conducted over several years showed that up to an 18% reduction in N fertilizer application was possible for winter wheat without negatively impacting yields or grain protein content. Variable rate soil N applications (high and low) used by growers led to increases in yield in both high-yielding and low-yielding areas. However, these benefits did not occur in every year and variations in seasonal weather conditions played a large role in the results,

pointing to a need for decision-support tools that growers can use in tandem with precision-N techniques.

Tremendous field-scale variability exists in many farm fields in the region, affecting grain yield, grain protein concentration, and nitrogen use efficiency (NUE, a measure of how much applied N is recovered in harvested crop). Thus, the uniform field management recommendation traditionally used by wheat growers to determine N requirements can be improved upon. Our CFF research determined that site-specific precision N management strategies for hard red spring wheat could improve NUE as well as yields. We developed indices that farmers can use to evaluate relationships between crop grain yield and grain protein content, and to diagnose field areas with over- or under-application of N, poor N utilization or uptake efficiencies, or significant N loss.

Findings from the CFF Project have helped farmers with their decisions to invest in precision N tools, and they have reported significant savings in agrichemical applications including nitrogen. Continued improvement of precision N decision-support tools will lead to greater adoption by farmers as they represent a low-risk, high-reward strategy for farmers and a small, but readily attainable GHG mitigation strategy for the public. If a 5% reduction of N fertilization was possible for both spring and winter wheat on Washington's annual 2.3 million acres of wheat, the IPCC estimate of 1.25% N fertilizer release as N₂O would predict an annual savings of 166,000 kg N₂O emissions, or 24,700 MT CO₂e.⁵

Because nitrogen is necessary for plant growth, and N₂O emissions are an inevitable byproduct associated with nitrogen transformation in soils, it will be virtually impossible to eliminate these emissions. Even biological sources of nitrogen generate substantial nitrous oxide emissions. Thus, in the long run, developing strategies that increase nitrogen use efficiency and reduce N₂O emissions is likely to become increasingly important.

Conservation Tillage

Conservation tillage including no-till has been widely proposed as a strategy to increase carbon sequestration on agricultural lands (e.g., Lal et al., 1998; Allmaras et al., 2000; Sperow et al., 2003; Purakayastha et al., 2008; Huggins and Reganold, 2008), although not all research supports a carbon benefit for all forms of conservation tillage (West and Post, 2002; Deen and Kataki, 2003; Yang and Wander, 1999). Interest in the GHG benefits of reducing tillage has also grown as voluntary carbon market enterprises have offered carbon credits to producers for implementing conservation tillage and no-till.

Direct measurement of soil organic carbon (SOC) to determine carbon impacts of conservation tillage can require many samples to account for field variability,

⁵ This calculation assumed that fertilizer use followed 2006 average fertilizer use of 101 lb ac⁻¹ on spring wheat and 77 lb ac⁻¹ on winter wheat (NASS 2009).

increasing costs. Therefore a combination of modeling and field approaches have been suggested for estimating the impacts of management changes on greenhouse gas emissions, particularly over larger areas. As part of the CFF Project, we enhanced CropSyst, a robust, process-oriented simulation model, to predict biologically-based soil C changes and N₂O emissions. Because the model is based on mechanistic principles, it can be used to study the effect of climate, soils, and management on cropping systems productivity and the environment in any global location. It has been tested extensively on numerous crops in both the PNW (e.g., Pannkuk et al., 1998; Peralta and Stöckle, 2002; Marcos, 2000; Jara and Stöckle, 1999; Kemanian, 2003) and elsewhere in the world (e.g., Stöckle et al., 1994, Stöckle et al., 2003; Pala et al., 1996, Donatelli et al., 1997; Stöckle et al., 1997, Stöckle and Debaeke, 1997; Monzon et al., 2006; Wang et al., 2006; Benli et al., 2007; Todorovic et al., 2009).

To ensure accurate simulations of N₂O emissions and to better understand the range of fluxes that could reasonably be expected, we carried out field measurements of nitrous oxide fluxes under experimentally-induced conditions in various dryland systems of interest including no-till, organic and native perennial systems. Results confirmed that seasonal climatic conditions, soil moisture and nitrate availability all influenced N₂O fluxes, as expected. Under control conditions, N₂O fluxes were generally constant and low, typically less than 0.5 g N₂O-N per hectare per hour, regardless of management regime. However, simulated rainfall and fertilizer applications transiently elevated N₂O fluxes to quite high values, particularly during the summer.

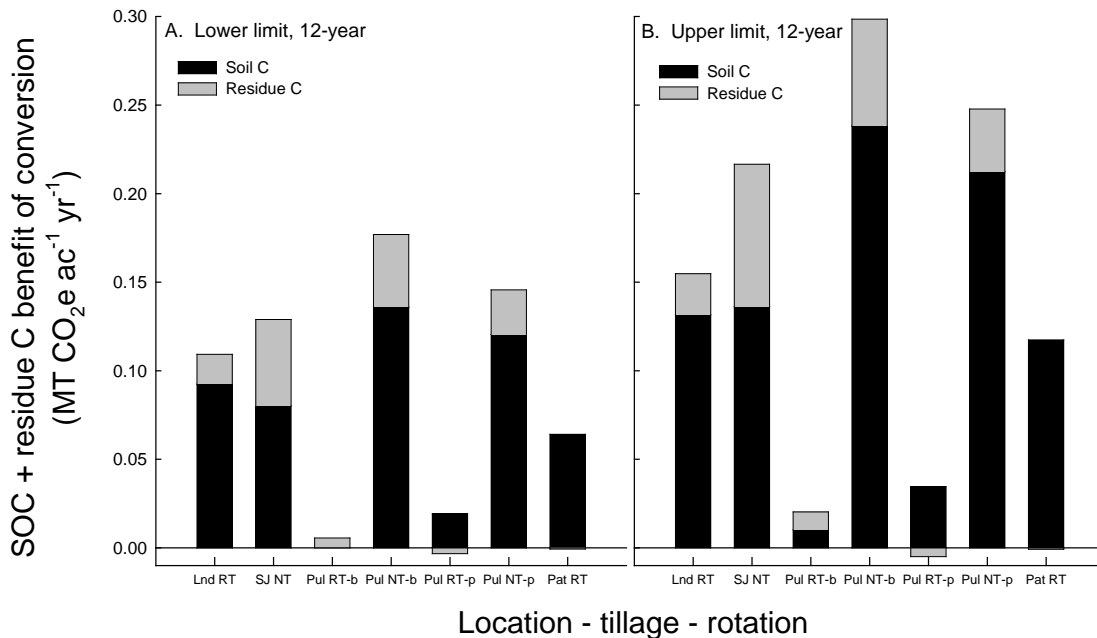
We used CropSyst to model the impact of changing from conventional tillage (CT) to reduced tillage (RT) or no-till (NT) in designated crop rotations in three dryland locations with varying amounts of rainfall (Lind, < 380 mm annual precipitation; St. John, 380 mm–460 mm; and Pullman, 460 mm–550 mm), and one irrigated location (Paterson).⁶ Multiple crop rotations, one with spring barley and one with spring pea, were simulated for Pullman only. To accommodate real-world variability, the modeling used a lower and upper boundary for tillage effects on SOC oxidation rates to obtain a range of expected changes in soil organic carbon (Δ SOC) due to biological processes after adoption of reduced- or no- till. Possible erosion effects were not included.

Regardless of location or rotation, CropSyst predicted an increase in SOC over a 12-year period after conversion to RT or NT from CT (Figure 1). CropSyst predicted increases in soil carbon storage ranging from slightly above zero to 0.3 MT CO₂e per acre per year with the switch from CT to conservation tillage, with location and rotation also influencing the results. Conversion to NT provided substantially greater benefits than conversion to RT, a finding that is consistent with Liebig et al.

⁶ For further information on methods for simulations, see the chapter titled “CropSyst simulation of the effect of climate, tillage and rotation on the potential for carbon sequestration and on nitrous oxide emissions in Pacific Northwest agriculture ([Stockle et al., this report](#)).

(2005) based on their analysis of C sequestration in response to tillage in the northwestern U.S. and western Canada.⁷ The relative benefit of conversion to RT was much greater in the driest location (Lind) than in wetter areas (Pullman). Surface residue C such as partially decomposed straw or plant residues made a significant contribution to the total C conserved. A permanent return to conventional tillage from conservation tillage would generally be expected to increase the vulnerability of SOC to oxidation, however Purukayastha et al. (2008) reported that a one-time moldboard plowing following long-term no-till actually increased SOC presumably because surface residues were buried.

⁷ Leibig et al. (2005) reported an average SOC increase of 0.07 MT CO₂e/ac/yr (0.05 Mg C/ha/yr) for reduced tillage and 0.40 ± 0.28 MT CO₂e/ac/yr (0.27 ± 0.19 Mg C ha⁻¹ yr⁻¹) for no-tillage under continuous dryland cropping.



Lnd = Lind (crop rotation winter wheat – summer fallow)
 SJ = St. John (crop rotation winter wheat – spring barley – summer fallow)
 Pul-b = Pullman (crop rotation winter wheat – spring barley – spring wheat)
 Pul-p = Pullman (crop rotation winter wheat – spring wheat – spring pea)
 Pat = Paterson (crop rotation sweet corn – sweet corn – potato)
 RT = reduced tillage; NT = no tillage (both as compared to conventional tillage)

Figure 1. Simulated average annual change in carbon over the first 12 years after converting from CT to either RT or NT for various crop rotations at four locations in eastern Washington State. Carbon values include both soil organic C in the top 30 cm and residue C. Changes in carbon were simulated with a low SOC oxidation rate (lower boundary, part A) and a high SOC oxidation rate (upper boundary, part B).

Carbon sequestration benefits are very different than most other GHG mitigation in agricultural systems. Unlike reductions in N₂O emissions, which are realized continually once a new management practice is implemented, there is an upper limit to the amount of carbon that can be stored in soils (Huggins et al., 1998; Paustian et al., 1997b). Thus, over time, decreasing amounts of carbon will be stored each year until a new “steady state” is reached (Figure 2). Carbon storage will essentially remain at or near this level until an environmental or management change occurs that raises or lowers the boundary.

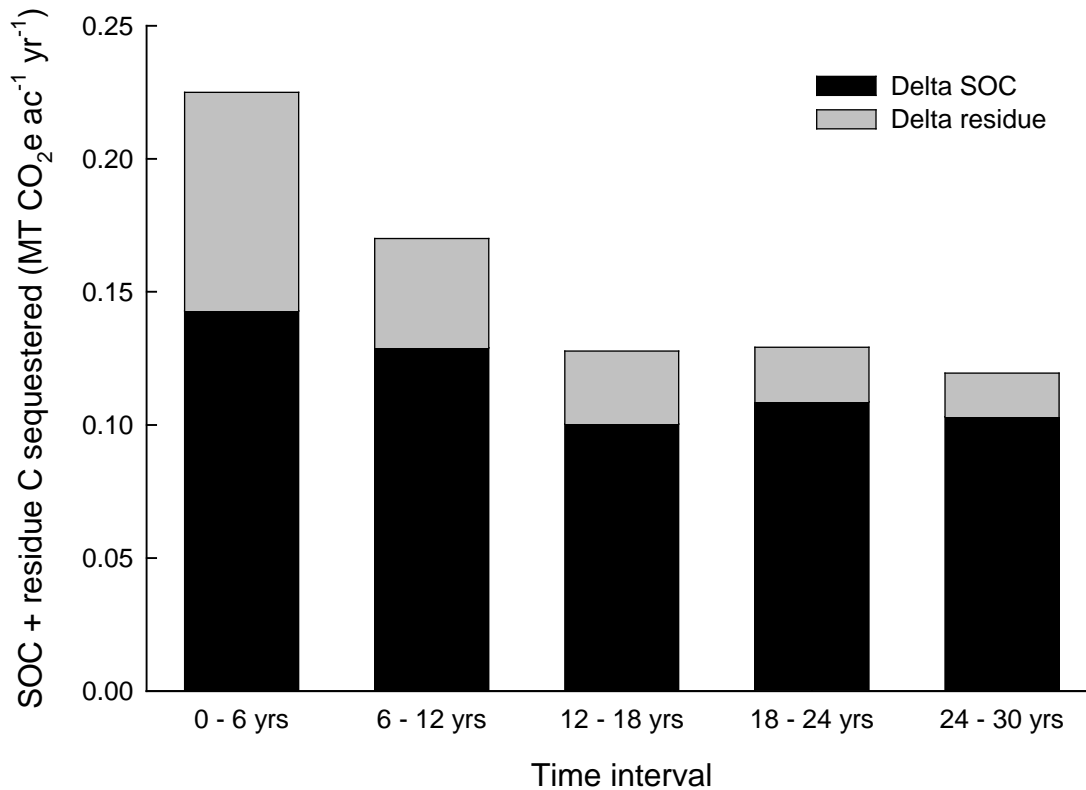


Figure 2. Simulated average annual change in carbon after converting from conventional tillage to no tillage for the winter wheat – spring barley – spring wheat rotation in Pullman. Carbon values include both SOC in the top 30 cm soil and residue carbon, and assume a lower boundary SOC oxidation rate.

It is important to note that while the switch from CT to RT or NT may result in net carbon storage, CropSyst results predicted that the RT or NT system will still have nitrous oxide emissions roughly equivalent to the CT system, mostly ranging from 0.1 to 0.2 MT CO₂e per acre per year. The magnitude of N₂O emissions tends to vary significantly from year to year, and could be reduced in all of the tillage systems by management changes that increase nitrogen use efficiency.

Conservation tillage has so far been much more common in dryland systems than irrigated systems. Our CFF field research also tested reduced tillage in an irrigated system. In a potato – sweet corn – sweet corn rotation, reductions in tillage (from seven tillage operations to four tillage operations) resulted in equivalent yields for potato and the first year of sweet corn. Yields for the second year of sweet corn (normally lower than the first year) were reduced by 15% for RT, compared to an 8% reduction for CT. Given that a potato – sweet corn – sweet corn rotation is not common in commercial production systems in the Columbia Basin, it may be possible to eliminate the comparative yield loss through future adjustments in crop rotation.

While reduced tillage is possible in irrigated regions, and while it provides other important environmental benefits (e.g., erosion control), it may have a minimal impact on GHG emissions. In our field study, reducing tillage did not significantly affect soil characteristics measured in the field. This may be due to the short period under reduced tillage (3 years), but our CropSyst simulation results also suggest that the overall benefit is small. Soils in the irrigated study area are sandy and overall levels of SOC are very low, which likely makes it difficult to sequester C in these soils.

Our results contrast with Entry et al. (2002), who estimated a somewhat larger accrual of SOC from a review of studies of irrigated agriculture in arid shrub steppe ecosystems throughout the U.S. and worldwide, concluding that a gain of 0.47 MT CO₂e per acre per year would be reasonable over the first 30 years if irrigated land managed with moldboard plow was converted to irrigated conservation chisel-tilled crops or 0.074 MT CO₂e per acre per year if converted to native sagebrush vegetation.

Conservation tillage, especially through no-till, is predicted to increase the amount of carbon sequestered in soils throughout the dryland and irrigated areas we evaluated. While C sequestration may be less than in more humid regions, conservation tillage is a cost-effective strategy that also delivers other important benefits such as soil erosion control and water quality improvement which increase agricultural sustainability.

Increasing Residue Inputs to Cropping Systems

Residue inputs such as straw play an important role in soil C levels. The CFF Project developed the [C-Farm model](#), a simplified version of CropSyst that is appropriate for working directly with growers and extension personnel (Kemanian and Stöckle, 2010), and used this model to examine field data concerning the impact of crop residue inputs on SOC storage.

Growers in the PNW Direct Seed Association provided field data to the CFF project from soils typical of the dryland production region, with a representative range of SOC levels. The [C-Farm analysis](#) indicated that the three main factors influencing SOC sequestration potential in no-till dryland systems in the PNW are (in order of importance): initial SOC (low better than high) > residue input to the soil (high better than low) > tillage intensity (low better than high), with most of the potential determined by the first two factors (Figure 3).⁸ Conservation tillage also can conserve soil moisture that leads to greater crop growth and residue levels. Given the importance of residues, proposals to remove crop residues for biofuel use may be in conflict with the goal of maintaining or increasing soil carbon.

⁸ Direct-seeded systems are defined by the Pacific Northwest Direct Seeding Association as “any method of planting and fertilizing done with no prior tillage to prepare the soil” and includes no-till, as well as one and two pass systems.

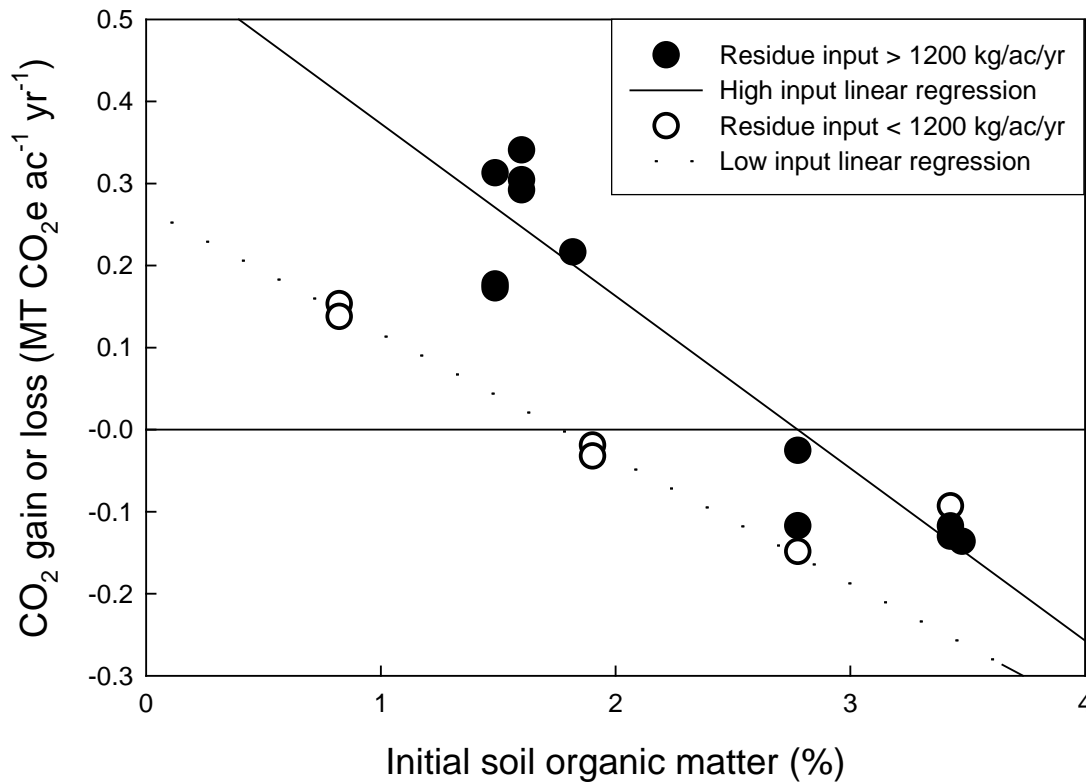


Figure 3. C-Farm simulation of SOC gain or loss as a function of initial soil organic matter and residue input for selected direct-seed cropping systems in the U.S. PNW.

Our work in newly irrigated areas of the semi-arid Columbia Basin also illustrated the potential impact of increasing residue inputs. The conversion of native ecosystems to rain-fed agricultural production usually results in a loss of soil organic matter, with well-documented examples in dryland eastern Washington, the Great Plains and the Corn Belt (Purakayastha et al., 2008; Jennings et al., 1990; Paustian et al., 1997b). However, conversion of arid- or semi-arid lands to irrigated cropping overcomes the native moisture limitation on plant productivity. [In a study of the conversion of native shrub-steppe in the Columbia Basin to irrigated vegetable production](#), SOC increased from 0.43% (native) to 0.64% (farmed) in three years.⁹ Increased plant growth from irrigation and fertilization added more residues to the soil, and compost additions also added stabilized carbon. From a GHG perspective, this increase in SOC must be compared to emissions from (i) fertilizer manufacture, storage, transport, and application, (ii) pumping irrigation water, (iii) farm operations such as tillage and planting, and (iv) dissolved carbonate in irrigation water (West and Marland, 2002; Schlesinger, 1999).

⁹ The difference was statistically significant at $p=0.05$

The benefits of increasing crop residue inputs were also indicated by an analysis we performed of 131 existing published and unpublished data sets measuring soil organic carbon within the approximately 8 to 10 million acres of non-irrigated cropland in northern Idaho, north central Oregon, and eastern Washington. In the same precipitation zone, rotations with summer fallow tended to have accelerated rates of SOC loss compared to annual crop rotations. Repeated burning of straw also accelerated SOC losses compared to no burning ([for data, see Brown and Huggins, this report](#)).

Increasing crop residue inputs over the course of a crop rotation can impact SOC storage, especially in areas where SOC levels are currently low. Management strategies to increase residue inputs (depending on location, etc.) include reducing fallow and burning, using higher biomass crops, and adding green manures to crop rotations. Addition of organic amendments such as manures ([Brown and Huggins, this report](#)), compost, or biochar (Granatstein et al., 2009; [Yorgey et al., Chapter 22 this report](#)) can also provide carbon benefits. Each of these amendments will need further study of economics and long-term impact to determine their potential as a climate mitigation strategy.

Perennial Cropping

Growing perennial rather than annual crops reduces tillage and simultaneously increases biomass additions to the soil through perennial plants' more extensive root systems. Our review of existing SOC data in the dryland PNW (described above) showed that compared to annual cropping systems, mixed perennial-annual systems increased mean SOC stocks in the soil profile by 1.94 (± 1.35) MT CO_{2e} per acre per year in Agroclimatic Zone 2.¹⁰ Considering the high standard deviations associated with these data, we can apply a cumulative probability distribution to the data to determine a more conservative estimate of C sequestration potential for perennial cropping systems. For instance, 75% of the sites in Agroclimatic Zone 2 had SOC gains of at least 0.82 MT CO_{2e} per acre per year.

As with increasing residue inputs to soils, a variety of strategies may be used to achieve an increase in perennialism. Taking cropland out of annual cropping through enrollment in the Conservation Reserve Program (CRP) has positive impacts on SOC. A study of 20 Washington State soils found a C sequestration potential over 4.5 to 5.5 years in CRP soils of approximately 0.27 MT CO_{2e} per acre per year if applied to a 7.5-cm soil profile (Karlen et al., 1999).¹¹

¹⁰ ACZ 2 is defined by wet-cool annual cropping, with more than 400 mm of annual precipitation, and 700-1000 growing degree days. Soil depth is not a factor. ACZ 3 is defined by fallow-transition cropping, with 350-400 mm of annual precipitation, 700-1000 growing degree days, and soil depth greater than 1 m (Douglas et al., 1990; 1992)

¹¹ This value is similar to the 0.30 MT CO_{2e} per acre per year value used by Cook (2007) but much less than the 1.40 \pm 1.28 MT CO_{2e} per acre per year estimated by Liebig et al. (2005) for conversion of cropland to grass.

There may also be opportunities to grow perennial bioenergy crops, or even food crops. Our switchgrass trials showed that this perennial grass can build soil organic carbon levels, even when the aboveground biomass is being harvested and removed for bioenergy production ([for data, see Yorgey et al., Chapter 22 this report](#)). And while most major food crops are annual crops, development of crops such as perennial wheat (DeHaan et al., 2007; Cox et al., 2006; Scheinost et al., 2001) would help enhance this strategy and does show promise.

As with increasing crop residue inputs, increasing perennialism could lead to considerable improvements in SOC.

Many GHG emissions management strategies provide significant environmental and/or economic benefits to farmers, providing other reasons to encourage their adoption. These benefits include nutrient and waste management (for anaerobic digestion), water quality and other environmental benefits (for precision N management), and soil erosion benefits (for conservation tillage). In this context, carbon credits may represent additional revenues that will help induce producers to adopt these technologies.

The beneficial non-climate environmental impacts of anaerobic digestion, precision N management, and conservation tillage have been well documented by others. Therefore, though we review these benefits briefly below, they were not a focus of the Climate Friendly Farming Project. Recognition of these benefits may play an important role in discussions of how to encourage adoption of these strategies in the Pacific Northwest.

Anaerobic Digestion

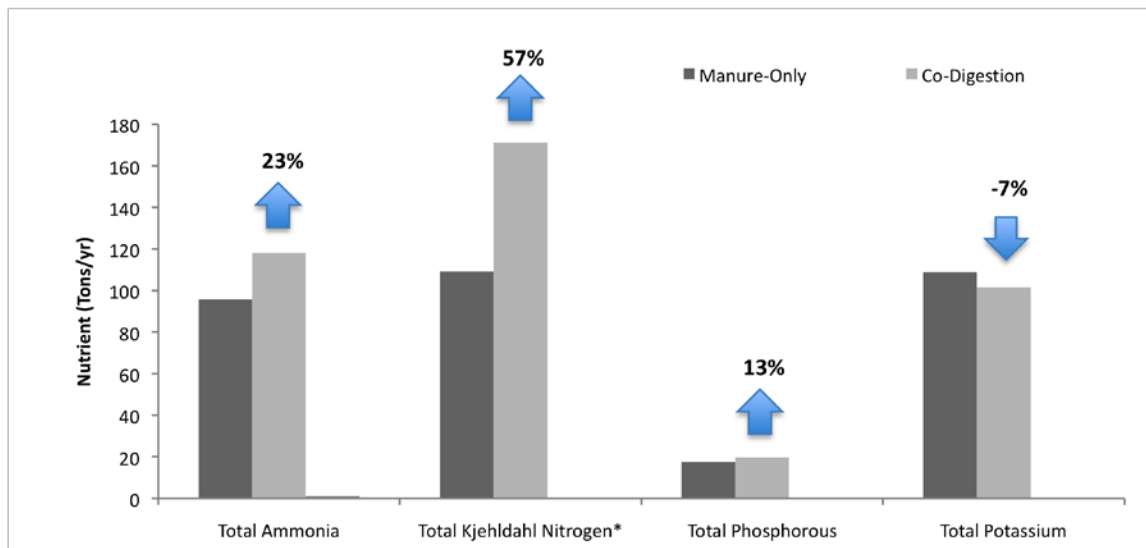
[Modern anaerobic digestion \(AD\) technology](#) is an established wastewater treatment approach that converts complex organic material to biogas containing methane (CH₄) under anaerobic conditions. In the process, it diminishes odors, stabilizes waste, decreases pathogen counts, and reduces GHG emissions (Martin and Roos, 2007; US-EPA, 2004; US-EPA, 2005a; US-EPA, 2008). Under the CFF project, we supported the installation of the first [commercial-scale AD system](#) on a dairy in Washington State. We used this system as a research platform to develop improved technologies and [explore project economics](#). Several more dairy digesters have now been built in the region, and others are being planned.

Anaerobic digesters built in other parts of the U.S. depend mostly on the sale of electricity from biogas-powered generators, but the low price of electricity in the PNW has been a barrier to farm adoption of AD. Our CFF digester research enhanced the potential of AD to serve as a biorefinery that produces other salable products, increases revenue, recycles nutrients, and generates additional energy.

AD systems have significant up-front capital costs, which can also pose a barrier to adoption. Our research targeted capital cost reduction [by developing a mesophillic AD system](#) that reduced waste residence time from 20-25 days to 5-10 days, thus reducing the size of digester vessel needed and lowering capital costs.

In addition to reduced costs, we focused on enhancing revenues to improve financial feasibility. The AD system constructed by our group was designed to accept non-manure organic wastes (substrates) from food processors or other off-farm sources. Our [financial analysis](#) showed that tipping fees from accepting substrates were the single most important contributor to the initial economic feasibility of the commercial scale digester constructed in Lynden, Washington, as measured by our analysis of net present value (NPV). Co-digestion of organic wastes also significantly increased biogas output. Additional revenues from the peat substitutes and bio-fertilizers developed by our research were not included in this financial analysis.

Although the economics of accepting substrates is positive, these materials exacerbate existing nutrient management problems on dairy farms, as more nutrients are imported onto the farm in the substrates (Figure 4). Currently, 36% of dairy producers are overloaded on N and 55% on P (USDA-APHIS, 2004), and land application of these nutrients is often the factor limiting the number of animals that a dairy owner can manage. AD technology by itself does not alter this nutrient balance, but the nutrient recovery processes we are developing show promise for solving these concerns and spurring wider adoption of AD technologies in the PNW and throughout the U.S.



*Total Kjeldahl Nitrogen includes both organically bound nitrogen and ammonia

[Figure 4](#). Farm-level nutrient loads experienced by VanderHaak Dairy (Lynden, WA) when they anaerobically digest dairy manure only, or when they co-digest manure and organic food wastes.

To address these nutrient concerns, we designed and tested an integrated system to recover nutrients, [purify biogas](#), and optimize effluent pH. Nearly 75% of total [phosphorous](#) and 70-80% of [ammonia](#) were removed from the AD effluent, and issues with solids removal, operating pH and temperature, and sulfuric acid dosing were satisfactorily addressed. Total chemical operating costs for lime and sulfuric acid are expected to be less than \$0.85/m³ effluent treated and total operating costs including stripping blower and associated pumps, etc. should be below \$1/m³ effluent treated or less than ½¢/gallon. Once treated, the effluent water could be used on-farm without causing nutrient overloading.

Our analysis shows that in order to offset operating costs alone for effluent treatment (not considering labor, maintenance and capital expenditures), the single combined fertilizer product (1.5% P and 2.5% N dry weight) which combined primary screened fiber with the system's P-rich solids and ammonium sulfate slurry, would need to sell for a market value of roughly \$15/ton excluding shipping costs. Potential markets are still being explored. These recovered nutrients contribute to reducing GHG emissions by substituting for synthesized or mined nutrient products, and may at some point be eligible for a carbon credit.

We also examined [value-added uses for digested fiber](#), a major AD byproduct. Digested fiber is generally used on-farm as an animal bedding substitute. It has been considered as a replacement for peat in container plant media, which would increase its value. Through our research, we developed a patented process for amending AD fiber with gypsum and S⁰, allowing for its use as a 1:1 replacement for peat in container media. We tested it extensively on potted plants, showing for example that *Petunia* plants grown on the fiber-based media produced aerial and root systems that were statistically equal to peat-based media (Figure 5). A major horticulture supply company has now tested the product and found it satisfactory.



Figure 5. Comparison between optimized digested fiber pre-treated media to peat moss control at 1:1 replacement

The CFF Project investigated whether using the [AD biogas as a vehicle fuel](#) would improve the economics relative to using it for electricity generation. This end use will likely add value, but its viability depends upon the presence of vehicles

retrofitted to use the fuel. We have conducted initial bench-scale tests of processes to purify the biogas to vehicle-grade methane, with encouraging results. Lessons learned through these tests were used to build a pilot-scale purification system, and testing is ongoing. Replacing petroleum-based vehicle fuel with biogas reduces emissions not only of greenhouse gases, but also of nitrogen oxides, hydrocarbons, and particles.

Within the context of these multiple product streams, carbon credits comprise only a piece of the financial picture, albeit one that can help enhance overall profitability of dairy farms and offset the considerable economic and non-economic costs of AD adoption. These include the up-front capital costs, and the costs of becoming familiar with new technologies. Other remaining barriers to adoption, which may not be alleviated by carbon credits or other financial returns, include resistance by utilities who may not want to purchase the power generated, and limited policy initiatives in the U.S. to recognize or promote biogas, plan for natural gas/biogas infrastructure, and implement a common long-term purchasing agreement for gas pipeline reception. Lastly, acceptance of off-farm organic wastes may require additional regulatory compliance depending on the jurisdiction.

Precision Nitrogen

Current strategies for managing the supply and fate of N in cereal-based agroecosystems were primarily developed on a regional scale for the uniform, whole-field application of N (e.g., Randall, 1993; Halvorson et al., 1986). Crop recovery of N, however, is often low as N removal in harvested grain is estimated worldwide at only 33% of applied N (Raun and Johnson, 1999). Poor N recovery is a function of uniform N management strategies as field variability in soil N transformations, N movement, and crop N use results in poor synchrony between N supply and demand and low N use efficiency (Fiez et al., 1995; Kitchen et al., 1995; [Huggins et al., Chapter 17 this report](#)). Cropping systems with inefficient N use are increasingly questioned as N movement beyond agroecosystem boundaries results in degradation of air (Mosier et al., 1996) and water (Burkhart and James, 1999; Huggins et al., 2001) at watershed and global scales (Tilman et al., 2001) and as producers seek greater efficiencies in N use to reduce external farm inputs and costs. Excess nitrogen has also contributed to greater nitrous oxide emissions, currently the most significant contributor to stratospheric ozone depletion (Ravishankara, 2009).

Precision N techniques have the potential to create a wide range of public and on-farm benefits, many of them with associated economic value. For example, Tegtmeier and Duffy (2004) estimated that nitrate pollution in ground and surface waters in the U.S. (of which an estimated 80% comes from fertilizers, livestock wastes, and mineralization of crop residues) costs the public \$188.9 million annually in the form of additional water treatment facility infrastructure costs.¹²

¹² 2002 dollars

Currently, farmers do not account for these externalized costs as their decisions are based on maximizing enterprise-level profits, generally based on high yields. Since the direct cost of N fertilizer is relatively low compared to the lost opportunity of increased yield if N is insufficient, growers often fertilize to the most optimistic yield assumption, leading to excess N application in many years. This situation is often exacerbated by uniform applications of N that contribute to poor crop N use efficiencies.

[Precision N management](#) can be a win: win situation for growers, increasing their profits and their NUE. But adoption depends on ease of use, technology compatibility, and effectiveness, issues that were a focus of our research. Our field testing of precision N tools mounted with geo-referencing (GPS) equipment showed that combine-mounted yield monitors and variable rate application controllers worked satisfactorily, but grain protein monitors did not meet grower expectations.

Increased N use efficiency leaves less N in the soil that is subject to losses as N₂O emissions to the atmosphere or nitrates to water. Fertilizer rates may decline on low-yielding areas, but could increase in high-yielding areas underfertilized by the current uniform rate. This efficiency should also improve profitability. Our [research did show the potential for overall fertilizer N reduction at the field scale](#), but this will vary with the field. Nitrogen reductions are not currently eligible for carbon credits, due to concerns about potential double-counting if fertilizer is being produced by an industry that is capped (as has been proposed by several policy schemes). Therefore, we did not investigate the potential carbon credits of precision N techniques.

Conservation Tillage

Conservation tillage provides well-known soil erosion benefits, conserves water, and can reduce the run-off of sediment, fertilizers, and pesticides into waterways (Huggins and Reganold, 2008). Observations of reduced tillage in our irrigated field trials at Paterson indicated that soil erosion reduction benefits may be substantial (Figure 6). Leaving a cover of corn stover (~5000 kg/ha) early in the potato phase of the rotation protects soil from blowing away in the spring of the year when winds are normally strong. Growers benefit economically from reduced erosion in the short term through reduced seedling losses from sandblasting, and in the long term through retained soil productivity and land value.



Figure 6. Potato emergence from adjacent conventional tilled (left) and reduced tilled (right) plots at Paterson, WA. Pictures taken following a period of high winds in April 2005.

Other public benefits from reduced soil erosion include lower costs to maintain roadway ditches (currently estimated at \$1.5 million annually in Whitman County, WA) and longer reservoir life from reduced sedimentation. Conventionally tilled fields in the Columbia Basin are more prone to wind erosion that frequently causes air quality concerns and has led to traffic accidents due to poor visibility (Stetler and Saxton, 1996). The public value of these erosion reductions could be quite significant. In the U.S., the cost of off-site soil erosion damage was estimated at \$37.6 billion annually in 2001 (Uri, 2001).

Farmers may also realize increased profits from adopting conservation tillage. To explore the possible role that carbon credits could have in spurring adoption of reduced tillage and no-till in irrigated and dryland areas of eastern Washington, we combined our [biophysical modeling results](#) with [economic modeling](#). To allow for a clear exploration of the impact of carbon credits on market returns, the analysis did not incorporate government payments or crop insurance indemnities, nor did it incorporate the value of non-carbon environmental benefits. Carbon-only values were used for this analysis because current carbon credit protocols incorporate only the value of sequestered carbon, and do not address simultaneous N₂O impacts.¹³

Our [economic analysis](#) of representative farms for three different dryland areas (Lind, St. John, and Pullman) suggested that without carbon credits, conventional tillage was more profitable than reduced tillage in low (Lind) and intermediate (St. John) rainfall areas of dryland eastern Washington.¹⁴ Assuming equal yields from CT

¹³ As discussed within the report, there are many different considerations necessary to measure carbon sequestration values (see chapters [13](#), [14](#), [18](#), [19](#) and [23](#)). For simplicity within this economic analysis, we used values that showed the lower and upper bound for the net carbon benefit from a change from CT to RT or NT over 12 years in the whole soil profile, including residue.

¹⁴ A representative, or typical, farm was created for each location. The enterprise budgets are meant to be characteristic; they are not a mathematical average of a large number of producers. Costs and

and NT, the gap in profitability ranged from \$13-15/acre/year. The lower profitability of NT in our scenarios was largely the result of higher pesticide costs for NT. Changes in the price of inputs such as Roundup®, and in relative wheat yields, changed the magnitude of, and in some cases reversed, the profitability gap. For example, if NT yields were 10% higher, the two systems would be nearly comparable, with \$17/ac/yr net returns for the NT system in St. John and \$19/ac/yr for CT. Higher yields under NT are not implausible, as large yield increases have been achieved under NT in the ongoing Direct Seed Mentoring project, in which direct seed farmers plant land for conventional growers who are interested in no-till systems (Meyer, 2009; Painter et al., 2010).

Carbon credits could increase profitability of reduced tillage systems, but in these dryland regions with low soil organic matter, gains in carbon sequestration tended to be relatively small. Thus, carbon credit prices would have to be unreasonably high to impact relative profitability. Using the range of C-sequestration values predicted by CropSyst for Lind, the carbon credit would have to be \$70 to \$103 per MT CO_{2e} in order to induce farmers to switch from conventional tillage to no-till (using the upper and lower bound SOC oxidation estimates, respectively, for net CO_{2e} savings in the full profile with residue, over 12 years). For St. John, the similar carbon credit would have to be \$62 to \$102.

Conversely, in high rainfall dryland areas (Pullman), our analysis suggested that profitability increased as tillage decreased. For example, under a very typical winter wheat – spring barley – pea rotation, profitability increased from \$32/ac/yr under CT, to \$34/ac/yr under RT, and \$37/ac/yr under NT. The lower profitability of conventional till was caused by higher weed pressure relative to drier areas, and thus by the need for more tillage in the conventional system, which raised fuel, labor, and machinery costs. As with the drier areas, any number of changes in production costs, relative yields, and prices easily changed the relative profitability of these systems.

This analysis begs the question of why there is not already more adoption of NT in this region, which has consistently been lower than the national average despite serious erosion problems (Conservation Tillage Information Center, 2008). Possible explanations include (i) cash flow problems may prevent purchase of the necessary machinery, (ii) farmers may be reticent to adopt no-till practices due to past failures in the region before necessary management changes were identified and largely solved, or (iii) concerns about yield drag during the transition to conservation tillage, despite evidence to the contrary (e.g., Guy and Lauvier, 2007; Janosky et al., 2002; Camara et al., 1999). A final explanation for lower no-till adoption rates in this region is the underlying profitability of all these scenarios. Crop failures are virtually unknown in this higher rainfall area, unlike in the lower rainfall regions. Farmers may be resistant to change from something that has proven profitable.

returns will differ significantly if farm size, machinery complement, productivity, etc. vary from the assumptions for these representative farms. For further details, see Painter, this report.

These rich, deep soils appear to be endlessly fertile, but estimations of soil erosion suggest that the region has already lost about 35-45% of its topsoil from erosion over the years (Jennings et al., 1990). Including the on-site and off-site costs of the erosion from CT shifts profitability in favor of reduced and no-till systems (Painter, this report).

In this context, carbon credits would represent an added incentive to switch to conservation tillage in the high rainfall zone. Depending on the value of the carbon credit, the financial incentive might be large enough to overcome some of the barriers mentioned above. For example, at \$12 per MT CO_{2e}, whole farm subsidies (using lower and upper bound estimates for the net GHG benefits for the whole profile, with residue, 12-year timeframe CropSyst value) range from just under \$5,000 to \$10,000 for switching to NT, depending on the method used to model net CO_{2e} impact. Subsidies for switching to RT, on the other hand, would be much smaller, due to the smaller carbon sequestration, and likely would not be high enough on their own to induce farmers to change management.

Unlike tillage comparisons in the dryland region, the economic analysis of a representative farm in the irrigated region at Paterson used yield data obtained from our experimental reduced tillage system. Over a four-year period, our sweet corn – sweet corn – potato rotation showed that, despite lower costs of production, reduced tillage was slightly less profitable than conventional till (\$800/ac/year for RT and \$809/ac/year for CT). The gap in profitability was caused by reduced yields in the second year of corn production in our trials. With additional trials, the problems underlying the drop in yield could almost certainly be solved, which would make the RT system more profitable than the CT system. Increases in the costs of fuel or machinery labor would likewise improve the relative profitability of RT.

Though the gap in profitability is small, the value of carbon credits would need to be quite high to induce adoption of reduced tillage, because the soils in irrigated regions sequester only small amounts of carbon. For RT profit to equal CT profit at Paterson, the carbon credit would need to be \$67 to \$123 per MT CO_{2e} (using the upper and lower bound estimates, respectively, for net CO_{2e} savings in the full profile with residue, over 12 years). In reality, the difference between the two systems represents just 1% of net returns over total costs, and thus would be considered inconsequential by many growers. In addition, these results do not include the economic value of reduced wind-driven soil erosion, benefits that, as noted above, are significant.

Where carbon credits are not significant enough to catalyze conversion to RT or NT on their own, other approaches that lower barriers (through mentoring, low interest loans, cost-share or other approaches) and raise incentives (through payments for the public benefits of reduced erosion or other mechanisms) could increase adoption rates. Future economic, technological or policy developments could also be important.

Nitrous oxide emissions from agricultural sites are considerable, and much higher than those from native sites. Though it has previously received less attention than carbon sequestration, efforts to lower the GHG emissions from agriculture in the Pacific Northwest must include a more comprehensive assessment of nitrous oxide emissions. As other researchers have shown, GHG mitigation practices may impact more than one gas, and sometimes in opposite ways. The net impact depends on the combined influence on all gases.

Irrigated and Dryland Cropping Systems

[Field measurements of nitrous oxide emissions in irrigated conventional tillage and reduced tillage systems in the Columbia Basin](#) showed substantial nitrous oxide emissions compared to native vegetation. Total baseline fluxes of N₂O from sweet corn and potato were 16-22 times higher than native plots depending on crop and year, reflecting the overall high N demands of these crops and the application of fertilizer (Table 3).

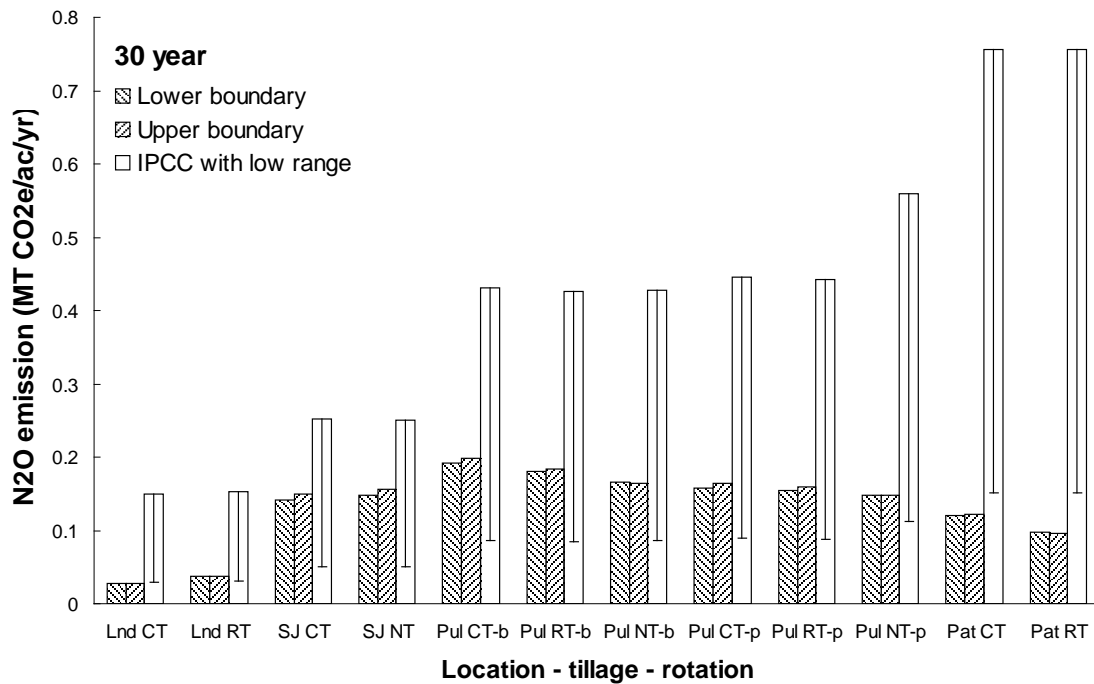
Table 3. Estimated growing season (May 15 through August 28) fluxes and global warming potential (GWP) of nitrous oxide and methane measured in native vegetation (NV), sweet corn, and potato plots at the USDA-ARS Integrated Cropping Systems Research Field Station located near Paterson, WA in 2005 and 2006 on a Quincy sand soil type. Average of conventional and reduced till treatments.

| Vegetation | 2005 | | 2006 | |
|-----------------------|---|--|----------------------------|-------------------------------------|
| | Field flux [†] kg/ha/season | GWP CO ₂ e [‡] MT/acre/yr | Field flux kg/ha/season | GWP CO ₂ e MT/acre/yr |
| N₂O | | | | |
| NV | 0.05 | 0.015 | 0.05 | 0.015 |
| Sweet corn | 0.87 | 0.257 | 1.05 | 0.310 |
| Potato | 0.93 | 0.274 | 0.91 | 0.268 |
| CH₄ | | | | |
| NV | -0.40 | 0.010 | -0.53 | 0.013 |
| Sweet corn | -0.17 | 0.004 | -0.14 | 0.003 |
| Potato | -0.23 | 0.006 | -0.16 | 0.004 |

[†]Field fluxes were calculated from static chamber measurements integrated over the season.

[‡]GWP = global warming potential; CO₂e= CO₂ equivalents based on the global warming potential of greenhouse gasses, relative to CO₂; CO₂ equivalents of N₂O and CH₄ are 298 and 25 times that of CO₂, respectively, over 100 years (IPCC, 2007)

[CropSyst results confirm that nitrous oxide emissions are sizeable](#) in both irrigated and [dryland systems](#), for all types of tillage (Figure 7). In some cases, the predicted GWP of nitrous oxide emissions are of a magnitude similar to the annual carbon sequestration benefits realized by switching to reduced- or no-till (cf. Figure 7 and Figure 1, earlier in the summary), reinforcing the need to understand and account for N₂O.



Lnd = Lind (crop rotation winter wheat – summer fallow)
 SJ = St. John (crop rotation winter wheat – spring barley – summer fallow)
 Pul-b = Pullman (crop rotation winter wheat – spring barley – spring wheat)
 Pul-p = Pullman (crop rotation winter wheat – spring wheat – spring pea)
 Pat = Paterson (crop rotation sweet corn – sweet corn – potato)
 RT = reduced tillage; NT = no tillage

Figure 7. Annual N₂O emissions, averaged over 30 years, either simulated by CropSyst or calculated according to the IPCC equation (1.25% +/- 1% of adjusted applied nitrogen), for various tillage intensities and crop rotations at four locations in eastern Washington State. Simulated data presented for either low SOC oxidation rate (lower boundary) or high SOC oxidation rate (upper boundary) in response to tillage. For IPCC estimates, we show the point estimate (bar) and the lower bound of the IPCC range (lower terminus of the line included within the bar).

In both our field and modeling results, it is somewhat surprising that N₂O emissions from irrigated systems are not larger than we found, given that potatoes require large amounts of N. Overall nitrogen lost as nitrous oxide-N as a percentage of N applied to sweet corn plots was 0.25% and 0.3% over 15 weeks during the growing season in 2005 and 2006, respectively. For potato plots over this same time period, in both years, 0.18% of N applied was lost as nitrous oxide.

We hypothesize that the common method of fertilizing crops in irrigated systems by “spoon-feeding” through fertigation might have lowered N₂O emissions by reducing the amount of nitrate in soil that was available for denitrification (as compared to less frequent, more concentrated applications). Meanwhile, N₂O emissions resulting from nitrification of ammoniacal forms of fertilizers used in the experiment likely

contributed significantly to N₂O losses. These results suggest that further investigation of whether N₂O emissions reductions can be realized from fertigation or other “precision timing” techniques that match the temporal availability of nitrogen with plant needs is warranted.

Methane fluxes were small compared to CO₂ and N₂O fluxes, and methane was consistently absorbed into the soil in irrigated systems, though absorption was higher in nearby native shrub-steppe (Table 3, above).

Measuring N₂O in the field is difficult, due to high temporal and spatial variability. Relationships between N₂O emissions and crop, tillage, and fertilization are complex. Our field measurements showed that the four-way interaction among crop, tillage, position (in row or out of row), and fertilizer level had a significant effect on nitrous oxide flux in 2005.¹⁵ In 2006 the significant interaction was among crop, tillage, and fertilizer level with position no longer contributing to the effect on nitrous oxide emission.¹⁶ For an expanded discussion, see Collins et al. ([Chapter 21, this report](#)). These challenges argue for the use of simulation models to predict N₂O dynamics, and the CFF Project added this capability to CropSyst.

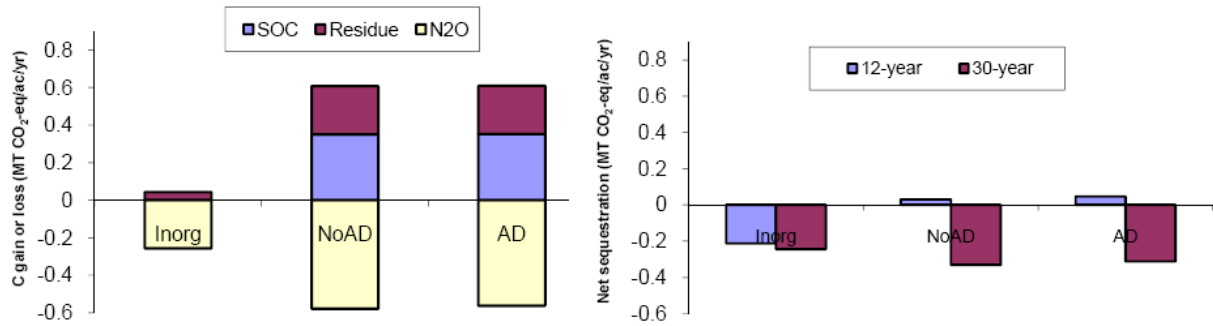
Dairy Systems

To understand the full GHG impacts of anaerobic digestion technologies, we need to understand the [GHG consequences of applying anaerobically digested manure to cropland](#). We conducted field experimentation as well as simulation modeling to evaluate the effect of applying anaerobically digested (AD) manure and undigested (raw) manure on carbon sequestration, methane and nitrous oxide emissions. Applications of nitrogen from AD manure, raw manure and inorganic fertilizer were made to a corn-triticale rotation on the basis of comparable yield expectations in both the field experiment and modeling simulations.

C sequestration, CH₄ and N₂O emission results from the model simulations are presented in Figure 8. These results show the GHG impact during the land application only (and therefore exclude impacts that occur during manure storage). Due to the significant amount of carbon in manure, both AD and raw manure resulted in considerably greater C sequestration than inorganic fertilizer while also generating considerably greater N₂O and CH₄ fluxes. In the 12-year simulation, both AD and raw manure provided superior GHG mitigation potential than inorganic fertilizers due to the high expected rate of C sequestration. However, in the 30-year simulation, inorganic fertilizer provided superior performance, as the amount of new carbon sequestered in soils diminished each year over the longer time horizon as soils approached a new steady state for SOC (for an example of how carbon sequestration diminishes over time, see Figure 2).

¹⁵ p = 0.0005

¹⁶ p = 0.0006



Inorg = inorganic N (chemical fertilizer)
 NoAD = un-digested manure (raw dairy manure)
 AD = anaerobically digested manure

Figure 8. SOC, residue, and N₂O unit GHG fluxes from a 12 year simulation (left) and net GHG fluxes for 12 and 30 year simulations (right) (expressed as CO₂e/ac/yr).

One additional concern that became clear from field experimentation is that CH₄ emissions were much higher than expected, 43 times greater than the uptake potential of the soil, with most of these emissions happening within 48 hours following application. The emissions observed after application can be attributed to the release of dissolved or super-saturated CH₄ produced prior to application during storage (for un-digested liquid dairy manure) or dissolved in the effluent (for the digested manure) and not from ongoing biological processes in the soil. It is apparent that a de-gassing and recovery system for CH₄ is needed for manure (whether digested or not) before application to soil systems. Based on the significant loss of CH₄ observed in our experiments, it is very possible that capture of this super-saturated CH₄ for additional energy production in a digester project could be profitable.

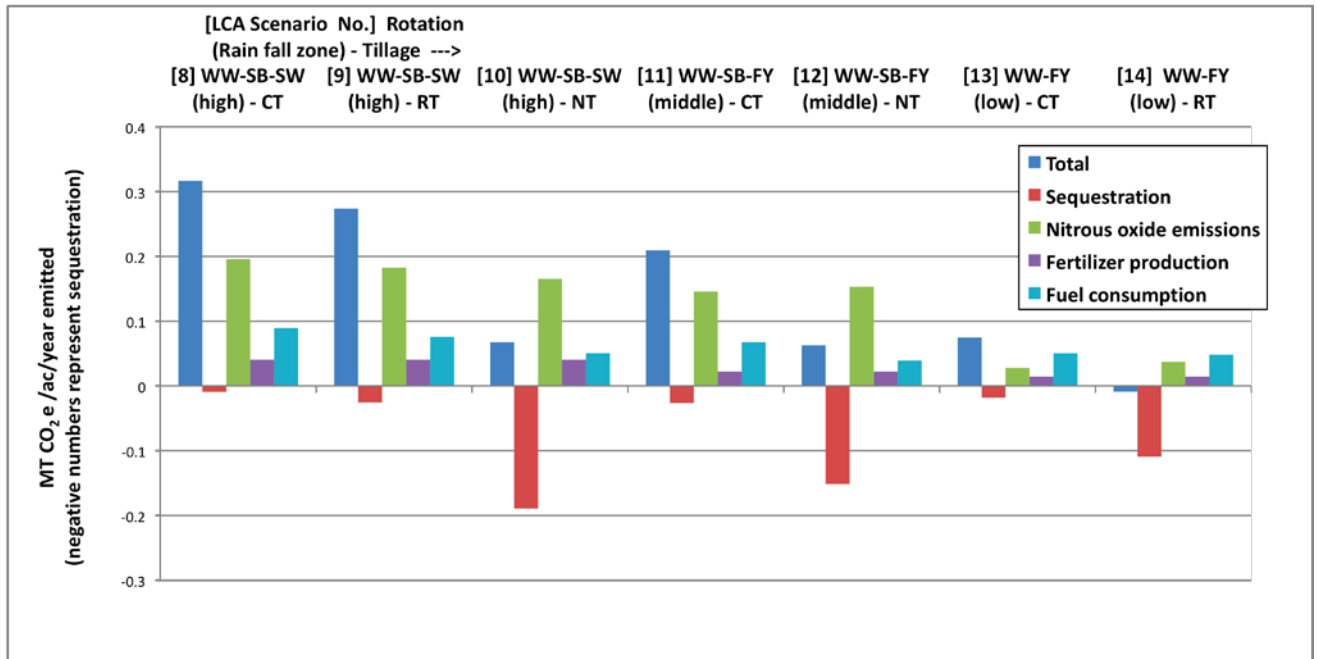
Life Cycle Assessment is a methodology that can compare several different management approaches and account for both direct and indirect emissions (e.g. current manure management practices versus anaerobic digestion; conventional tillage versus no-till or reduced-tillage).

Life Cycle Assessments (LCAs) are being used to compare the system-wide impacts of changes in agricultural management (Haas et al., 2000). LCAs are a scientific accounting tool used for assessing the relative merit of one product or management activity over another, generally with regard to their environmental impacts. [LCA was used in concert with our CropSyst modeling in order to incorporate indirect “upstream” GHG emissions.](#) For example, the CropSyst analysis of the impact of conservation tillage was limited in the sense that it did not incorporate differences in GHG emissions from changes in fertilizer, fuel and pesticide use, nor from reduced emissions associated with changes in farm equipment.

Our LCA analysis incorporated the upstream GHG emissions associated with fuel consumption (assuming consumption of diesel with a 99% conversion efficiency to

CO₂, as described by U.S. EPA, 2005b) and fertilizer (evaluated according to the International Fertilizer Industry Association, 2009), in addition to carbon sequestration and nitrous oxide emissions predicted by CropSyst. While we did not include the GHG emissions associated with pesticides in our analysis, these have generally been shown to be of smaller magnitude for wheat based cropping systems in the PNW (Schenck et al., 2008). Emissions impacts were calculated both on a grain weight basis for winter wheat only and on a per acre basis across the crop rotation.

The results of our LCA confirmed that converting from conventional tillage to conservation tillage reduced GHG emissions (as measured in CO₂e) in all studied locations of dryland winter wheat production. Figure 9 shows the GWP (CO₂e per acre per year) for differing levels of tillage in the different rainfall zones represented by Pullman, St. John, and Lind. These results show that emissions reductions with conversion to RT or NT were mainly due to sequestered carbon, a result that was confirmed when emissions impacts were considered on the basis of dry grain production. The reduced carbon sequestration benefit from conversion to NT and RT in lower precipitation areas reflects the effect of lower residue inputs from the reduced cropping intensity, as well as C losses in the fallow years. A shift from CT to RT or NT also generally reduced fuel consumption, and in wetter areas also reduced nitrous oxide emissions, but by relatively small amounts. Fertilizer-associated emissions did not change much with a switch from CT to RT or NT, but were lower for drier areas, because of the increased frequency of fallow years with no fertilizer application.



CT= conventional tillage; RT= reduced tillage; NT=no tillage
 WW= winter wheat; SB= spring barley; SW= spring wheat; FY= fallow year
 Rainfall zones are high (Pullman, 460 mm – 550 mm annual rainfall), middle (St. John, 380 mm – 460 mm) and low, (Lind, < 380 mm).
 Values are converted to MT CO₂e/ac/year on the basis of global warming potentials of greenhouse gasses, relative to CO₂; CO₂ equivalents of N₂O and CH₄ are 298 and 25 times that of CO₂, respectively, over 100 years (IPCC, 2007). 1 Mg CO₂e = 1 MMT CO₂e

Figure 9. Annual global warming potential per acre for wheat-based rotations under different tillage, precipitation zones, and crop rotations.

These results are preliminary, and are influenced by the system boundaries chosen for this analysis. For example, upstream emissions from fertilizer and fuel were included, but emissions from farm equipment manufacture and pesticides were not. Understanding the boundaries and assumptions utilized in any individual LCA study is critical for determining the appropriate use of the results in a policy framework. While the LCA approach may more accurately portray the full emissions impact of a given agricultural system or product, applying LCA results may or may not be appropriate in a some public policy frameworks as it could lead to double accounting of emissions reductions (e.g. credit applied for fuel reduction on farm while emissions from fuel are “capped” at the refinery).

Our review of historical and current data sets in the Pacific Northwest, and our own field results, found high variability for soil organic carbon levels. This is due to different sampling methodologies, as well as high underlying spatial variability resulting from soil erosion and deposition and other landscape-level properties. This variability makes it more difficult to develop reliable carbon sequestration data for use in carbon markets.

Our analysis of existing data sets of measured soil organic carbon from sites in the dryland PNW (described earlier) showed high variability in soil organic carbon responses to various changes in management depending on location. Data were analyzed according to agroclimatic zones (based on annual precipitation, soil depth, and growing degree days; see Douglas et al., 1990). Average soil organic carbon sequestration rates for the soil profile of 1.05 (\pm 0.94) MT CO₂e per acre per year and 0.32 (\pm 0.15) MT CO₂e per acre per year were reported for NT dryland cropping systems as compared to CT in agroclimatic zones (ACZ) 2 and 3, respectively.^{17 18}

Factors contributing to the large variability of SOC found on farm and research fields (see [Brown and Huggins, Chapter 13, this report](#)) include: (1) different soil sampling protocols such as soil sample depth, degree of residue and root removal (these have relatively high C concentrations), and time of year sampled (more or less crop residues and roots present) and (2) soil erosion processes.

Soil erosion redistributes SOC within the landscape as a function of detachment, transport and deposition processes. The high rates of soil erosion experienced across much of the PNW dryland region have been a major determinant of SOC levels at a given location. For example, assuming SOC levels of 1 to 2%, an annual soil erosion rate of 10 MT/ac means that 0.4 to 0.7 MT CO₂e per acre per year could either be lost from eroded landscape positions or gained in landscape positions

¹⁷ ACZ 2 is defined by wet-cool annual cropping, with more than 400 mm of annual precipitation, and 700-1000 growing degree days. Soil depth is not a factor. ACZ 3 is defined by fallow-transition cropping, with 350-400 mm of annual precipitation, 700-1000 growing degree days, and soil depth greater than 1 m (Douglas et al., 1990; 1992).

¹⁸ These SOC sequestration rates are similar to the global range (0.4 to 1.2 MT CO₂e per acre per year) reported by Smith (2004) for improved management practices. The ACZ 2 rate of change (1.05 MT CO₂e per acre per year) exceeds the upper limit of the 0.4 to 0.9 MT CO₂e per acre per year used by Follett (2001) for NT adoption on CT cropland. The ACZ 2 sequestration rate is also higher than the surface 30-cm rate increase (0.85 \pm 0.21 MT CO₂e per acre per year) reported by West and Post (2002) and the 0.7 MT CO₂e per acre per year estimate for the surface 20-cm reported by Lal et al. (1998) in a global and national analysis, respectively, of NT compared to CT. Furthermore, the ACZ 2 value is much higher than the 0.34 MT CO₂e per acre per year reported by Paustian et al. (1997a) and the 0.4 MT CO₂e per acre per year estimated by West and Marland (2001) comparing NT to CT sites nationally. However, the 0.31 MT CO₂e per acre per year sequestration rate estimated for conversion from CT to NT in ACZ 3 is similar to lower rates reported in many of the same studies (Liebig et al., 2005; Smith, 2004; Follett, 2001; West and Post, 2002; Lal et al., 1998; Paustian et al., 1997a; West and Marland, 2001).

where deposition occurs. Over time, this physical process can result in tremendous field-scale variability in SOC. In one 91-acre field in the Palouse, we found that soil profile SOC ranged from 22 to 110 MT C per acre (equivalent to 81 to 403 MT CO₂e per acre), over 5-fold variability. This variability in SOC was attributed, in part, to soil erosion processes interacting with landscape and soil properties and further complicates field measurements and comparisons of NT and CT ([Huggins and Uberuaga, this report](#)).

Furthermore, we do not know whether eroded SOC contributes more or less to atmospheric CO₂ than SOC that remains in place. Consequently, soil erosion impacts on GHG production are often ignored currently, and will need further research before they can be incorporated into understanding of GHG generation by agricultural activities. The change in SOC that is measured through soil sampling at a given location includes contributions of C inputs and losses from both soil erosion and biological processes. At this time, only the biological processes have been directly linked to GHG production, and CropSyst only includes them as well. Therefore, carbon credit decisions based on field measurements need to be carefully and conservatively evaluated with respect to GHG production, and modeling may offer a preferred approach for estimating soil carbon change if soil erosion processes are controlled.

Given these complications with field sampling, process-oriented modeling can aid in the determination of realistic SOC sequestration rates that are relevant to global climate change by simulating SOC changes due to biological processes. In addition, cumulative probability functions of SOC change, rather than average values, may be more appropriate for some applications.

Our project developed a cumulative probability function of SOC change, in contrast to the average value of SOC change, among studies. This allows the selection of data-supported yet conservative values for assessing expectations for SOC change relevant to GHG production; however using this approach requires sufficient historical data which may be lacking. For example, in the dryland PNW, sufficient data exist only for native conversion to cropland, adoption of NT, and use of a mixed perennial-annual rotation for ACZ 2 and to some extent ACZ 3.

The combination of reported field measures of SOC changes and CropSyst results for ACZ 2 and 3 are presented in Figure 10. Reported field measures of SOC sequestration that result in the wide extremes shown in Figure 10 are not due to biological processes alone but result from the combined influence of different soil sampling protocols, erosion processes and biological factors previously discussed. Using a conservative probability of 30% (30% of reported datapoints below, 70% above) enables us to reduce the uncertainty related to sampling and erosion processes in predicting SOC change and is corroborated with CropSyst simulations.

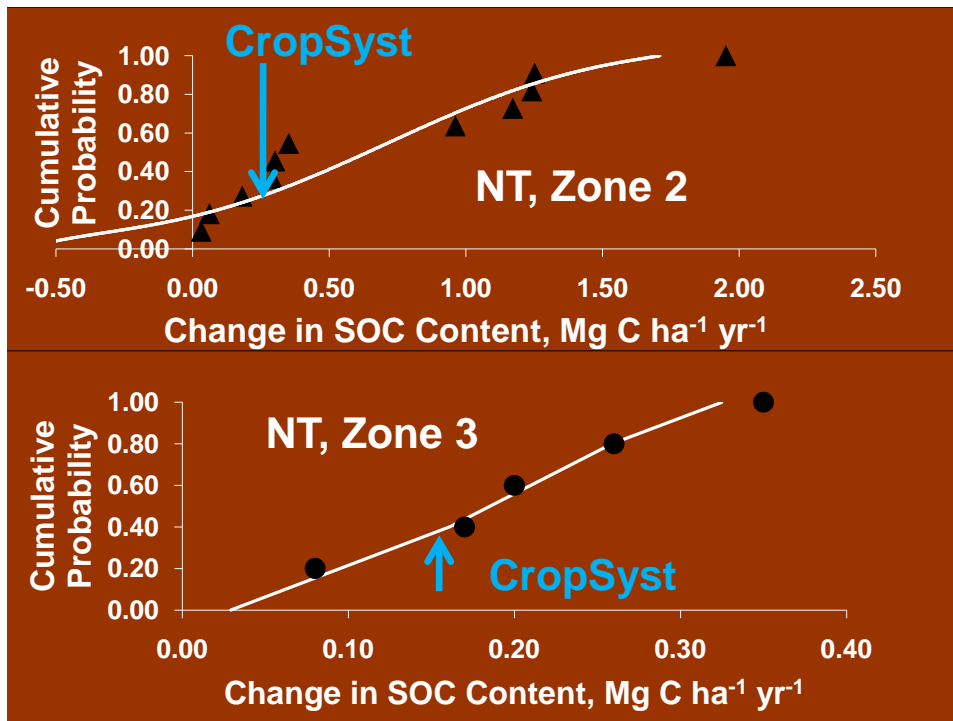
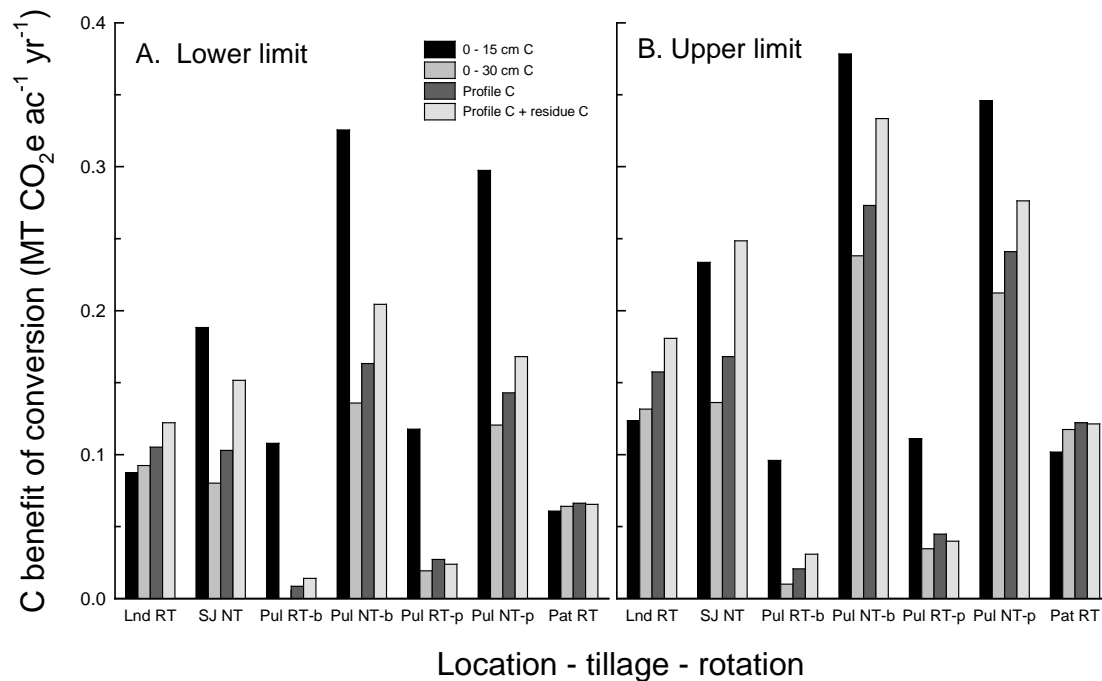


Figure 10. Comparison of soil organic carbon (SOC) changes from a change in management from conventional till to no-till (NT) in Agroclimatic Zones 2 and 3 predicted by cumulative probability function from field studies (data points and white line) with CropSyst model estimates (blue arrow).

However, modeling results indicate that carbon sequestration values could be different depending which part of the soil profile is measured.

Using CropSyst, we explored carbon dynamics at several different depths (0-15 cm, 0-30 cm, or whole soil profile, and with or without crop residue on the surface). Results varied depending on the soil depth that was analyzed. Most modeled dryland scenarios simulated sequestration in the top 15 cm, loss of C in the 15-30 cm layer, and very small amounts of C sequestration between 30 cm and the bottom of the profile, though there were important exceptions to each of these general patterns (Figure 11). Given that no RT or NT field operations extend below 15 cm, it makes sense that the benefits of reduced tillage were concentrated in the upper soil layers, because residues are not mechanically delivered below 15 cm, as they are in CT. However, anecdotal evidence from a long-term no-till site in the region (Aeschliman farm) indicates that additional micro-faunal processes may emerge that challenge this perception. Thus, sampling deeper than the common protocol of 10-20 cm may be important for understanding how SOC levels change over long periods of time. Our field research ([described in Huggins and Uberuaga, this report](#)) and that of others (Angers et al., 1997; Baker et al., 2007; Deen and Kataki 2003) supports this conclusion.



Lnd = Lind (crop rotation winter wheat – summer fallow)
 SJ = St. John (crop rotation winter wheat – spring barley – summer fallow)
 Pul-b = Pullman (crop rotation winter wheat – spring barley – spring wheat)
 Pul-p = Pullman (crop rotation winter wheat – spring wheat – spring pea)
 Pat = Paterson (crop rotation sweet corn – sweet corn – potato)
 RT = reduced tillage; NT = no tillage (both as compared to conventional tillage)

Figure 11. Simulated average annual change in carbon over the first 12 years after converting from CT to either RT or NT for various crop rotations at four locations in eastern Washington State. Carbon values are for indicated soil depths and include soil organic C or soil organic C plus residue C as indicated. Changes in carbon were simulated with a low SOC oxidation rate (lower boundary, part A) and a high SOC oxidation rate (upper boundary, part B).

In addition, including residue C measurements showed that in general, a long term increase in the steady-state residue C under NT systems made a relatively large contribution to the total C sequestered, albeit a contribution that is potentially ephemeral if residue management practices change due to its fast decomposition.¹⁹ Not surprisingly, the increase in residues occurred only in cases where soil disturbance was minimal (thus, generally not for a conversion to RT) and where residue inputs were sizeable.

¹⁹ Residue contained within a given field varies considerably from season to season and year to year, but a long-term steady state amount can be counted as contribution toward C storage, providing some additional advantage to NT systems. We therefore took residue C to be the minimum amount of residue, both above- and below-ground, that existed 95% of the time for the last 27 years of the 30-year simulations.

Washington State is unlikely to become a major producer of first generation biofuels such as oilseeds. However, these crops may have an important role to play as an alternative to other rotational crops, if they can be incorporated into existing rotations. On the other hand, Washington is well positioned to produce second-generation biofuels from feedstocks such as perennial grasses and organic residuals (e.g. manure, food waste). An understanding of net GHG impacts and other environmental impacts will be important for developing a sustainable biofuels industry.

Our economic analysis and that of others (Yoder et al., 2008) suggests that canola, an oilseed crop that can be used as a feedstock for biofuel, is unlikely to ever become sufficiently profitable in Washington State to become a primary crop. Depending on the success of breeding efforts and future economics, canola may have a role to play within current crop rotations in some areas of the PNW. In dryland areas where wheat is the main crop, canola can provide several important rotational benefits, including erosion, disease and weed control. Some of these benefits can also be provided to varying extents by other rotational crops, to which canola must be compared ([for economic analysis, see Yorgey et al, this report](#)).

One farm in Adams County, Washington, in an 11" annual rainfall zone (C. Hennings, personal communication, 2006) contrasted a typical two-year rotation of winter wheat-summer fallow with a four-year rotation of fallow-winter canola-fallow-winter wheat. In the typical two-year rotation, average winter wheat yields (based on county averages) were 45 bushels per acre, with an economic return of \$101 each time that wheat is produced, or \$50 per year. Based on the grower's records over more than 20 years, winter wheat in his four-year rotation had a 22% yield advantage of 55 bushels per acre. If a conservative canola crop failure rate of 20% is assumed due to poor conditions at seeding, average returns were \$77 per acre per year, 54% higher than the two-year rotation returns. However, other economic analyses completed by our group showed that under some conditions net returns could be even greater by substituting other crops such as dry peas, lentils, or garbanzos for canola within the rotation.

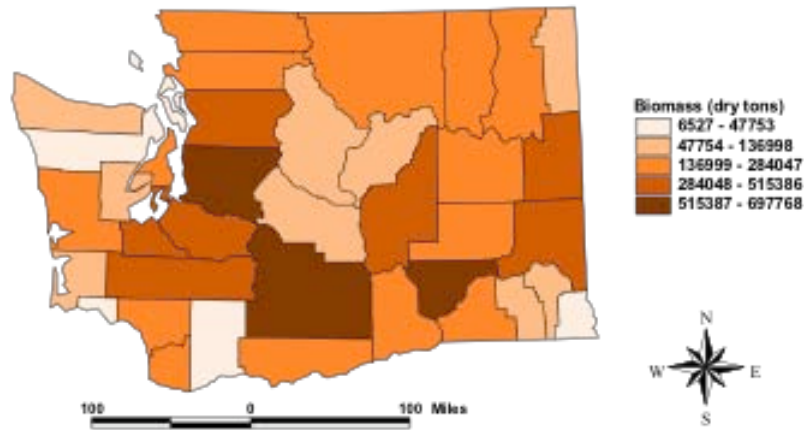
Switchgrass may be a promising bioenergy crop for some areas of the state, specifically if it is used to restore degraded soils or in more novel cropping systems, such as companion planting in early successional hybrid poplar plantations. Our irrigated switchgrass trials produced yields that surpass many areas where the grass is native (Table 4). Switchgrass can also sequester SOC; adjusting for C inputs from previous cropping, we estimated an average accrual rate of switchgrass-derived carbon of 1.6 MT CO₂e per acre per year for three-year old stands in our field trials. Analyses of the full environmental and GHG impact of switchgrass production for energy will need to consider whether the crop is really sustainable given the nutrient needs of the crop and the corresponding impacts on lifecycle GHG emissions. Economic analyses are also needed to determine whether or not the crop will provide a profitable option compared to other forage crops for growers in the PNW.

[Table 4.](#) Annual switchgrass yields in several states across the U. S. (Collins, et.al. 2007).

| State | Switchgrass variety | |
|-------------------|------------------------|--------------|
| | Kanlow | Cave In rock |
| | ----- T per acre ----- | |
| Texas | 4.5 | 2.4 |
| Upper South | 5.5 | 4.2 |
| Alabama | 8.3 | 4.2 |
| Iowa | 5.8 | -- |
| Nebraska | 9.2 | 7.3 |
| Washington | 14.8 | 9.4 |

Values presented are the sum of two cuttings per year.

Bioenergy generated from organic residuals tends to have a much more positive GHG balance than bioenergy from dedicated energy crops, because there is no “GHG cost” associated with producing the residuals (these costs are credited to the main product) and because in many cases a credit may be available for eliminating GHG emissions that would have occurred from current waste treatment systems, in addition to the GHG benefits from replacing fossil fuels. A [biomass inventory](#) conducted by our group in collaboration with the Department of Ecology showed that Washington State has significant biomass resources, with regional areas of concentration where forestry and municipal wastes or forestry and agricultural wastes intersect, such as the Puget Sound/Cascade and Yakima regions. Areas with concentrated wastes align geographically with areas of the state where development of new business opportunities and jobs is of vital interest (Figure 12). Extensive potential exists for developing bioenergy resources with anaerobic digestion technology and next generation biofuel technology from currently stored or discarded food processing wastes, manures, the Organic Fraction of Municipal Solid Waste (OFMSW) and forest thinnings.



^a Mill residue and MSW paper were excluded because these materials are already being used for energy and/or recycling in many areas. For analysis with mill residues, see Frear et al., 2005 or Yorgey et al., Chapter 22 this report.

Figure 12. Underutilized biomass (dry tons per year) by county, excluding mill residues and municipal solid waste (MSW) paper.^a






While this inventory included biomass estimates of crop residues, we believe that it is essential to recognize the important agronomic role of crop residues; these are not “wastes” available for energy production. Further analysis shows that removal of wheat straw residues for biofuel production exports valuable nutrients from the field, and may leave inadequate residues to build or even maintain soil organic matter over time with the common crop rotations used to grow wheat in the PNW. In-field variability in wheat (and therefore residue) production is typically high (3-fold to 4-fold) due to differences in soil carbon levels, erosion, slope and aspect. Thus, uniform residue removal for bioenergy applied across a field will have vastly different impacts in different parts of that field.

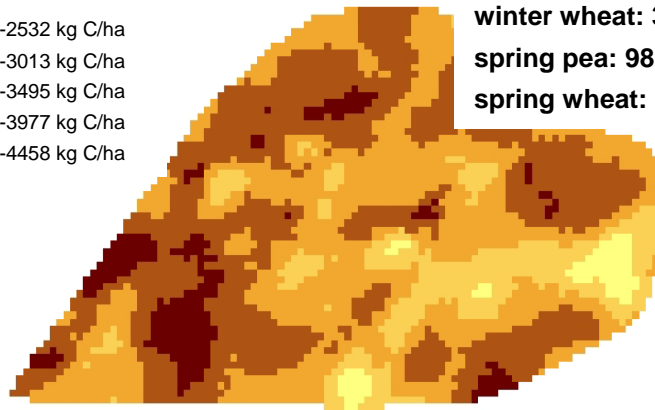
[Modeling by the CFF Project at the Cook Agronomy Farm in Pullman, WA](#) illustrates this point. Figure 13 shows the expected availability of wheat residue carbon from a three-year crop rotation (winter wheat – spring pea – spring wheat). From this image, the in-field variability of residue production is obvious. Note also that average residue production for the field during the spring pea year is only 981 kg C/ha, much lower than amount needed to maintain soil carbon levels over the rotation (2250 kg C/ha). The field average production of winter wheat residues (3431 kg C/ha) makes up for this on average, but only if no residues are harvested. And even without any residue harvest, there are areas of the field that likely are not receiving enough residues to maintain SOC levels over time.

Legend

ww_res_C

Value

| | |
|---|-------------------|
|  | 2050-2532 kg C/ha |
|  | 2532-3013 kg C/ha |
|  | 3013-3495 kg C/ha |
|  | 3495-3977 kg C/ha |
|  | 3977-4458 kg C/ha |

**Residue Production During Each Year of Rotation (field average):****winter wheat: 3431 kg C/ha****spring pea: 981 kg C/ha****spring wheat: 2345 kg C/ha**

**Annual C inputs needed to maintain organic matter:
2250-2750 kg/ha**

[Figure 13](#). Carbon (kg/ha) remaining in winter wheat residues produced in a winter wheat, spring pea, spring wheat rotation at WSU Cook Agronomy Farm, Pullman, WA.

[Pyrolysis](#) is another technology being considered for conversion of biomass to bioenergy. Biochar, essentially charcoal, is a co-product from pyrolysis that has been linked to significant soil improvement and long-term C sequestration in Amazonian soils (Lehmann et al., 2004). Biochar additions to soil are being proposed as a globally significant GHG mitigation strategy (Laird, 2008). However, the long-term effects of biochar amendment under field conditions in temperate region soils are not clear.

Preliminary laboratory research conducted by our group showed that biochars made from several in-state organic residuals increased carbon concentration in five soils from different parts of the state in proportion to increasing application rates. C-mineralization studies confirmed that the majority of the C added is biologically inert, and that the addition of biochar did not accelerate loss of indigenous organic matter through the 'priming effect'. An initial greenhouse study showed no benefit to wheat growth (Granatstein, et.al. 2009). Crop yield enhancements or nutrient conservation would be necessary to justify the cost of biochar purchase, transport, and spreading on a large scale, unless the price of carbon credits was very high.

Conclusion

The research carried out in this six-year effort points to promising strategies that should be pursued to make farming more climate friendly in the Pacific Northwest and beyond. While many challenges remain, we have identified a number of promising technologies that are either commercially ready, or are close. These technologies include the bio-conversion of organic wastes, precision agriculture, improved cropping systems (rotation and perennial crops), and reduced tillage. At

least as importantly, the tools developed through the project, including CropSyst, C-Farm, budgeting tools, and GISCOD, can be used broadly to investigate other promising strategies, both in the PNW and elsewhere. The CFF Project findings clearly illustrate opportunities for agriculture to reduce its carbon footprint while continuing to perform its primary role of feeding a large and growing human population. Many of these actions will also provide other environmental benefits, but may or may not currently be profitable choices for farmers. Policy-makers need to carefully balance these realities with the complexities of agricultural GHG mitigation strategies to achieve sustainable economic, environmental, and social outcomes from the agricultural sector.

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