Carbon sequestration potential in cropland soils in the inland Pacific Northwest: Knowledge and gaps

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Abstract

Cropland agricultural soils have the potential to either release (be a source of) or capture and sequester (be a sink for) carbon. We provide a summary of existing experimental and modeling evidence relating to the carbon sequestration potential of cropland soils in the Pacific Northwest. We also identify remaining knowledge gaps. The purpose of this summary is to provide context for regional discussions intent on fostering farming practices that show the best potential for carbon sequestration. We review regional research on the impacts of agricultural management strategies on carbon sequestration, including intensifying crop production, tillage, perennial crops, soil amendments, cover crops, crop rotation, reduced burning, and reduced erosion. Our summary suggests that a number of practices can provide real contributions to carbon sequestration, with the likelihood of substantial co-benefits in the form of soil conservation, improved water quality and soil water storage, increased microbial activity, and sustaining our soil's ability to grow food over future generations. Within cropland agriculture, the opportunities to build soil organic carbon are greater in annually cropped systems with higher productivity, though the benefits of particular management practices are variable and depend on multiple environmental and physical conditions. There is an ongoing need to establish credible estimates of carbon fluxes for Northwest agricultural systems. These estimates must also be accompanied by monitoring to determine whether cropland soils are achieving carbon sequestration goals. Thoughtful consideration of the environmental and production contexts surrounding Pacific Northwest agriculture, combined with targeted research to identify the most effective carbon sequestration practices, could lead to the development of strategies that can realize the contributions that croplands in the Pacific Northwest could make to climate change mitigation efforts.

Introduction

The science and the discussion about climate change—its causes, effects, and the paths forward to adapt to and mitigate its negative impacts—have progressed significantly over the last three decades, in many regions of the world, at different scales, and for different decision-making purposes. Agriculture is an economic sector that plays an important role in the global carbon cycle, and therefore has the potential to impact the progression of climate change. Cropland agriculture affects the carbon balance through production and use of nitrogen-based fertilizers that can generate greenhouse gases, photosynthesis that extracts carbon dioxide from the air and stores it in plants and soils, and management practices that impact how much carbon dioxide is released from soils back into the atmosphere versus captured in forms that can be sequestered in the soil (Smith et al., 2008). The consumption of agricultural products, either by livestock or people, eventually releases carbon back into the atmosphere as well.

Agriculture is also an important economic driver in the northwestern states of Washington, Idaho, and Oregon, with agricultural production valued at more than $22 billion in 2017, and farms comprising more than 42 million acres in the region (USDA, 2019). Agricultural soils have the potential to either release (that is, be a source of) or capture and sequester (that is, become a sink for) carbon (Smith et al., 2008). Some resources exist that can inform policy on management strategies that could increase the "sink" potential of agricultural soils. At a national scale, estimates exist of the carbon sequestration potential of croplands (e.g., Chambers et al., 2016; Fargione et al., 2018 and citations therein). Recently, the National Academies of Sciences, Engineering, and Medicine (2019) estimated that cropland agriculture could sequester 250 MMT CO_2e per year in the United States without compromising food security or biodiversity (i.e. without land conversion), using current technologies at direct costs of 0-$50 per ton of CO_2e. However, no similar assessments exist focused specifically on croplands in the Pacific Northwest. Those interested in this topic must search multiple sources of information, each providing conclusions for a particular set of conditions, many with specifics and caveats that make drawing overall conclusions to inform policies on climate change mitigation potential difficult.

The purpose of this document is to summarize existing experimental and modeling evidence for the potential that cropland soils in the Pacific Northwest have for sequestering organic carbon, and to identify where gaps in knowledge remain. Due to the lack of systematic and comprehensive data for the area west of the Cascade Mountains (a clear research need), the synthesis we provide focuses on the inland Pacific Northwest (more details below). Note that the focus in this white paper is on organic carbon; while some types of soils may also have high content of inorganic carbon, inorganic carbon has different dynamics, and is generally not as responsive to management.

This summary can help inform what farming practices or strategies show the best potential for carbon sequestration, providing context for regional discussions of policies that might lead to incentives for their adoption. It can also help identify what research tracks might be most informative for such policies, thereby providing incentive to fund such research in this region.

Why increase soil carbon?

There is a growing recognition that limiting future climate-related risks depends on stabilizing global warming at a lower temperatures, ideally close to 1.5°C, a pathway that would be possible if global emissions were reduced sharply by 2030, and reached net zero around 2050 (IPCC 2018). Within this context, there has been ongoing discussion of the role that "carbon drawdown" or "negative emissions technologies," which remove carbon from the atmosphere and sequester it, could play in stabilizing the Earth's climate (NASEM, 2019). While reducing emissions will *also* be required, carbon sequestration strategies may be less expensive than reducing some emissions, such as some transportation emissions and a substantial portion of agricultural and land-use emissions (NASEM, 2019). Sequestration can occur on a number of land types, including on agricultural lands, the focus of this publication.

Croplands' carbon sequestration potential is a reflection of the fact that, historically, converting land to agriculture resulted in large carbon losses, with total losses since conversion estimated to be roughly 20-70% in the US (Flach et al., 1997; Paustian et al., 1997; Lal, 2001; Franzleubbers and Follett, 2005). This is consistent with estimates of carbon loss for dryland (non-irrigated) cropland soils in the inland Pacific Northwest, where an analysis of existing experimental data indicated that that 75% of converted native land lost at least 0.14 to 0.70 Mg carbon ha^{-1} yr^{-1} (0.21 to 1.04 MT CO_2e $acre^{-1}$ yr^{-1}, see sidebar, *Units in this Publication*) over an average of 55 to 74 years, depending on the soils, rainfall, and cropping system (Brown and Huggins, 2012).

The loss of soil carbon when native vegetation is converted to agriculture results from factors such as export of carbon in harvested portions of the crop as food, fiber or fuel, lower inputs of organic matter in annual cropping systems compared to the native systems they replace, increases in the release of carbon as carbon dioxide due to increased decomposition, and, in some cases, erosion losses (note that losses from erosion may or may not contribute to greenhouse gas emissions; in some cases, erosion simply moves the soil carbon across the landscape, sequestering it elsewhere).

While this pattern of carbon loss in agricultural soils has occurred across much of the United States, there are important exceptions to this general rule. One key exception in our region is the irrigated Columbia Basin and other semi-arid irrigated croplands. In this semi-arid environment, where plant productivity and soil organic carbon levels are low in natural ecosystems, a comparison of agricultural and native soils in Grant County, Washington, showed that soil organic carbon levels were *higher* in fields that had been cultivated for two or three years than in soils under native shrub steppe (Figure 1, Cochran et al., 2007). Increases in organic soil carbon were influenced in this example by irrigation and the resulting increased plant productivity, cultivation, crop residue incorporation, and dairy manure compost amendments. These factors are discussed in more detail below, as they are the basis for practices by which producers can and do increase soil carbon sequestration in agricultural soils.

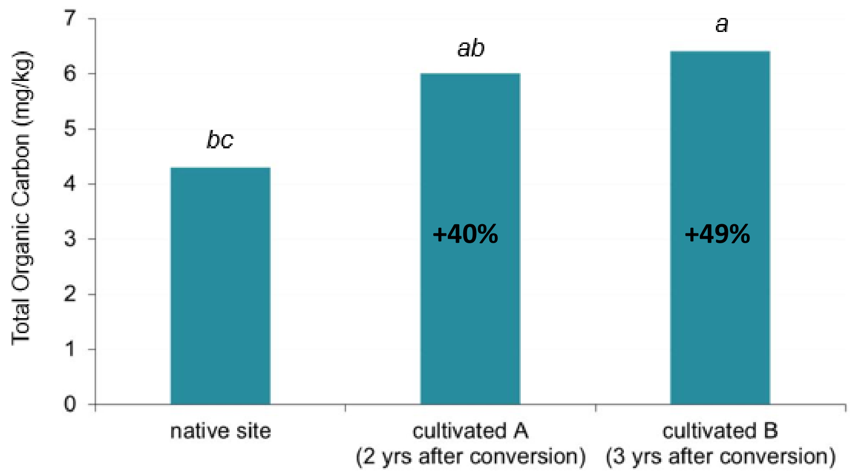

Figure 1. Total organic carbon in native shrub-steppe soils (native site) and two sites recently converted to irrigated agriculture in Grant County, Washington. Percent changes reflect increases relative to the organic carbon in the native site. Based on data published by Cochran et al. (2007).

Environmental co-benefits of carbon sequestration

It is also important to note that although climate impacts (e.g. soil carbon sequestration and greenhouse gas mitigation) are the focus of this publication, there are other critical benefits to be gained from management strategies focused on a broader objective of improving soil health. These benefits include reduced erosion, improved water retention in soils, improved water quality in nearby waterways, increased microbial activity, improved nutrient cycling, enhanced crop productivity (e.g., Johnston, 1986; Brown et al. 2011; Cogger et al. 2013; Reeve et al. 2012; Sharratt et al. 2018; Wuest et al. 2005), and even the potential for improvements in human health through more nutritious foods (Brevik and Burgess 2013). Approaching the challenge of agricultural climate mitigation through a more comprehensive strategy focused on improving soil health could help transform a climate mitigation strategy into a win-win opportunity for both consumers and producers, creating social signals and market incentives that help producers overcome the additional costs and operational hurdles necessary to increase soil organic carbon.

The fundamentals of the carbon cycle

Any climate mitigation policy intending to provide incentives for increased carbon sequestration in agricultural soils must be based on an understanding of some of the basics of soil carbon dynamics. This understanding is also critical for correctly interpreting experimental and modeling data relevant to soil carbon in the Pacific Northwest, which we summarize in later sections of this publication.

Soil scientists consider the amount of carbon stored in the soil as the carbon “stock,” and the additions and losses of carbon from each pool as “fluxes” or “flows.” It can therefore be helpful to visualize soil carbon as water flowing into and out of a bathtub (Figure 2). The size of the soil carbon "stock" or "pool" is analogous to the amount of water in the bathtub. The rate of water flowing into the tub—which can be changed by strategies that open or close the faucet—represents the rate of carbon additions to the soil carbon pool. The rate of carbon losses is represented by the size of the drain, which can be changed by placing a plug in the drain (though, unlike the bathtub analogy, it is not possible to completely eliminate losses of soil carbon).

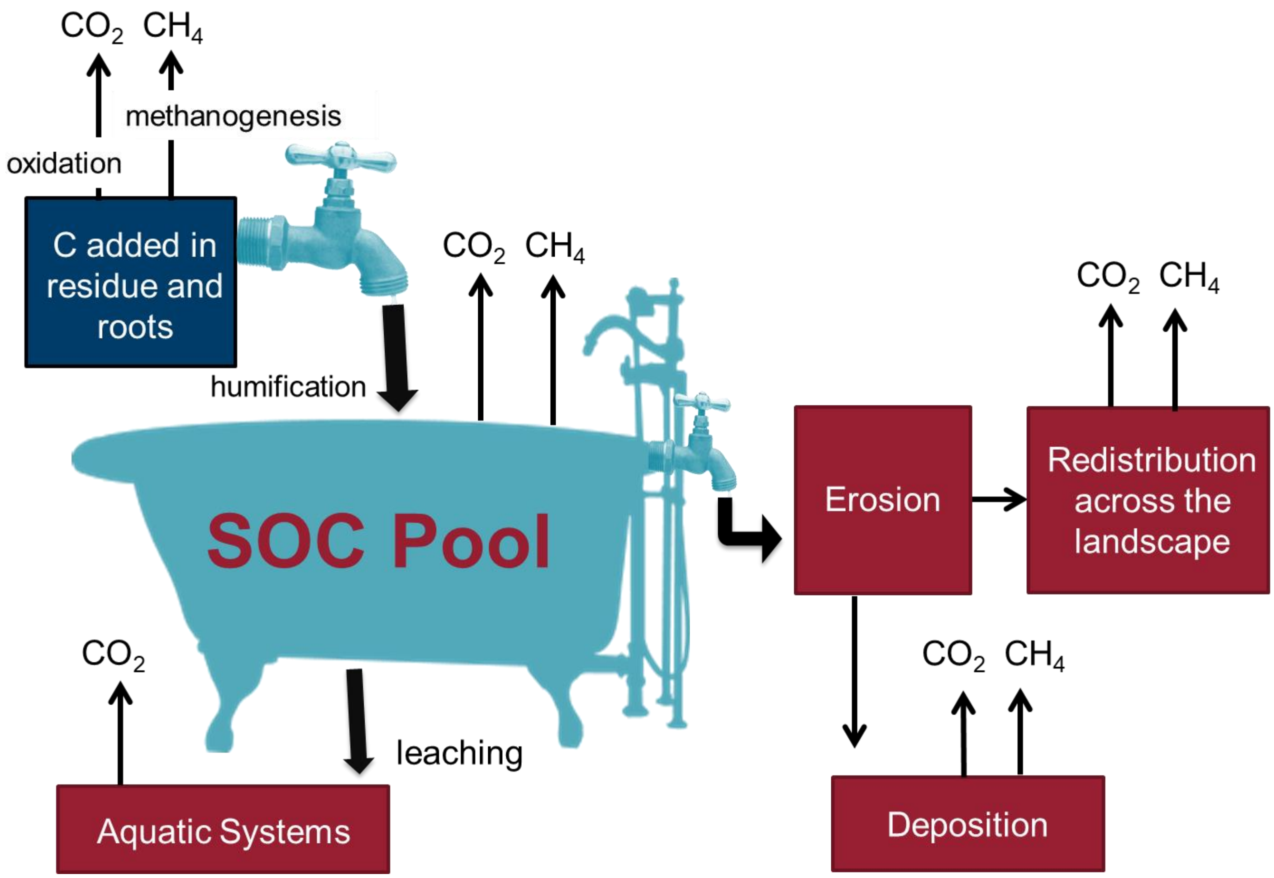

Figure 2. Conceptual schematic of the agricultural soil organic carbon pool, with the pool size at any given time reflecting the size of ongoing carbon additions and losses. The blue pool represents the soil organic carbon pool in a given agricultural system, dark blue box represents inputs of carbon to the agricultural system, and red boxes represent other carbon pools elsewhere across the landscape. Figure by Elizabeth Allen and Georgine Yorgey, adapted from Lal (2001).

For most agricultural soils, each soil type at a particular geographic location is considered to have a practical upper limit to the soil organic pool. This upper limit reflects the point at which all the available soil surface area is interacting with soil carbon, and can be conceptualized as the size of the bathtub itself. There is also a practical lower limit, where the remaining soil carbon is very hard to decompose. Peatlands are usually considered an exception to this, unlikely to saturate in the amount of carbon that can be stored (Smith et al. 2014).

Soil carbon levels at any given time are a reflection of ongoing gains and losses from the soil, the fluxes. If additions of carbon are greater than the losses, the soil carbon pool will increase in size—what researchers call a "carbon sink." If, on the other hand, losses are greater than additions, the soil carbon pool will shrink, becoming a "carbon source," mainly producing carbon-based greenhouse gases, like carbon dioxide or methane.

Additions of carbon to soils can come from a variety of sources, including:

- Unharvested plant residues and roots
- Root exudates
- Living and dead microorganisms or other soil biota
- Animal residues (e.g. manures), either from animals that graze the site, or added as soil amendments
- Other organic amendments. In addition to the animal residues described above, other amendments could include compost, biochar, etc.
- Erosional deposits

Meanwhile, losses of soil carbon result from a number of processes, including:

- Decomposition of organic materials, with oxidation and release of carbon dioxide
- Leaching of some types of soil organic carbon (i.e. soluble carbon)
- Soil erosion processes

What practices can be used to increase carbon in cropland soils?

Based on the understanding of soil carbon dynamics described above, mitigation occurs if agricultural management is changed in ways that increase carbon inputs, reduce carbon losses, or both. Strategies that focus on *increasing inputs* include (Lal, 2004a, 2004b; Smith et al., 2008; Lal, 2015):

- **Increasing crop residues**, which can be achieved through a number of practices. Varieties of the same crop that produce more residues can be chosen, for example. Or crop rotations can be changed to alternate the primary revenue-producing crop with crops that leave greater residues (e.g. Kirby et al., 2017). If nutrient availability is constraining crop growth, then fertilization generally increases carbon inputs to the soil by increasing biomass production, including the residues left after harvest.
- I**ntensifying production by eliminating fallow or double-cropping (growing two crops in a single year)**. If a crop is grown every year, total residues being added to the soil would be greater than if one crop is grown every other year, with a fallow in between. Double cropping also increases the amount of time when there are living plants in the soil. There are of course limitations to this strategy, as there are usually good reasons why producers grow a single crop or fallow their fields, such as lack of sufficient rainfall, or limits to the growing season.
- **Adding manure or other organic amendments**. Carbon-rich soil amendments, such as manure, compost, and biochar, provide an additional, new flux of carbon—a new faucet to the soil carbon pool bathtub. Under current agricultural patterns in the inland Pacific Northwest, livestock and cropping systems are frequently spatially isolated, so the costs of transporting and applying organic amendments is one important barrier to their more widespread use (Yorgey et al., 2017a).

- **Using cover crops**. Cover crops are crops grown specifically to add residues to the soil and are not harvested, though in some cases they may be grazed. If grazed, a portion of the organic carbon may be harvested by the grazing animals (though another portion may be cycled back to the soil as manure). Although there has been widespread interest in cover cropping in the Pacific Northwest, and although there are some viable irrigated cover cropping systems (e.g. Yorgey et al., 2017b), existing research to date with single- and multiple-species cover crops in dryland eastern Washington and semiarid eastern Colorado has not found reliable agronomic or economic benefits (Thompson and Carter, 2014; Nielsen et al., 2015; Roberts et al., 2016).
- **Replacing annual crops with perennial crops**. Perennial crops tend to produce more and deeper rooting systems, which add more organic carbon to the soil. They are also tilled less, even if under conventional tillage, as tillage only occurs, generally, when the crop is renewed, rather than every year as with annual crops. However, many of the most widespread cropping systems are based on annual crops. While efforts are underway to develop perennial varieties of some grain crops, these efforts are not guaranteed to be successful, or economically viable (e.g. Bell et al., 2008; NASEM, 2019).

Strategies that focus on *reducing losses* include (Lal, 2004a, 2004b; Smith et al., 2008; Lal, 2015):

- **Reducing the amount or intensity of tillage**. Tillage leads to aeration of the soil, accelerating decomposition, which releases carbon in inorganic forms, such as carbon dioxide. Soils are tilled for multiple reasons, including weed suppression, and accelerating the release of the nitrogen and other nutrients held in the organic matter where they are inaccessible to the crop. In the driest parts of the inland Pacific Northwest, tillage can also suppress loss of water during the fallow year, by severing capillaries in the soil that bring water to the surface, making it susceptible to evaporative losses (Hammel et al., 1981; Wuest 2010; Wuest and Schillinger, 2011).
- **Reducing the burning of residues**. Burning of the carbon-rich organic residues volatilizes that carbon, mostly as carbon dioxide. Burning has declined in the Pacific Northwest in recent years due to air quality concerns, but has been used in some cases to improve seedbed preparation and to reduce viable weed seed (Tao et al., 2017; Burke et al., 2017).
- **Strategies that reduce erosion** will reduce in-field losses of carbon, though the ultimate fate of the eroded and re-deposited soil carbon—and therefore its impact on regional-scale soil carbon—is not well understood (see, for example, Lal, 2003 and Van Oost et al., 2007).

Cropping systems are complex, and their different components are interrelated. One key consideration is how future climate itself may affect what strategies to increase carbon sequestration are feasible in particular locations. Kaur and colleagues, for example, studied which climatic variables best allow us to determine agro-ecological classes (an agricultural land use classification) in the inland Pacific Northwest. They then used climate model projections for those climatic variables to evaluate how those classes might shift in the future. They found that dynamic classes (i.e. those with different cropping systems during different time periods) and agro-ecological classes that included fallow increased in area, while stable classes (i.e. those consistently maintaining a particular cropping system) and annual cropping classes were expected to shrink (Kaur et al., 2017). These shifts would likely imply an increase in fallowing, increased erosion hazards, and potentially decreased flexibility in rotations (Kaur et al., 2017), with the potential for negative impacts on carbon storage in soils.

Independently Karimi et al. (2017) used a process-based cropping system model to evaluate likely future shifts in cropping systems in response to climate change. They found that the direct effects of increased carbon dioxide concentrations on wheat and the earlier start to the growing season would more than compensate for the negative effects of increased temperature and decreased water availability, leading to higher wheat yields. Higher crop yields are usually associated with higher residue yields, and therefore carbon inputs to soils. Karimi and colleagues concluded that overall shifts away from crop-fallow towards annual cropping systems would be expected (Karimi et al., 2017).

The contradictory results of these two studies highlight the uncertainties that remain and limitations of current research methodologies, with each emphasizing different key factors needed to accurately project changes in agro-ecological classes: the effects of increased atmospheric carbon dioxide (Karimi et al., 2017), and the impacts of socio-economic factors driving farmers' cropping decisions (Kaur et al., 2017).

A third study focused specifically on the soil organic matter content in soils, relating values from long-term experiments in the inland Pacific Northwest to mean annual precipitation and temperature (Morrow et al., 2017). They found that these climatic variables were more influential than either tillage or cropping intensity in determining organic matter levels, and that the direction these variables are expected to change as the climate changes would lead to declines in organic matter in surface soils (Morrow et al., 2017). However, one important limitation to this study is that it does not incorporate the impacts of higher carbon dioxide levels in the atmosphere on plant growth.

In sum, these studies point to a need for ongoing work to clarify how future climate may affect what strategies to increase carbon sequestration are feasible in particular locations and cropping systems in the future, as well as the likelihood that they will be adopted by farmers.

Constraints on the carbon sequestration potential of cropland soils

As previously mentioned, for most agricultural soils, there is a practical upper limit to the amount of carbon that can be stored in soils – thus, there is a maximum "bathtub size" in Figure 2. This is partly because of environmental constraints including temperature (more organic carbon is stored in cooler climates, as decomposition is slower), moisture (more organic carbon is stored in wetter soils, as there is less oxygen available to aid in its decomposition), soil texture (more organic carbon is stored in finely textured soils, as clay particles help stick to and protect organic material from microbes that decompose it) and soil structure (more organic carbon is stored in well-aggregated soils, as the aggregates protect the organic material inside from decomposition).

However, the practical upper limit also results from the point of equilibrium between inputs and outputs, and the fact that losses from the soil carbon pool depend on the size of the pool itself. The more carbon in the pool that can decompose, or erode, or leach, the more carbon that will generally be decomposed, eroded, or leached. Generally speaking, when a change in management increases carbon inputs, this leads to higher outputs. If the constraints and management stay constant, these will eventually balance out, and thus, the system may approach a new steady state over time. This is when carbon input rates equal carbon output rates, such that the size of the soil organic carbon pool remains steady over many years.

From a climate change perspective, mitigation only occurs if the size of the soil organic pool (the amount of water in the bathtub) *increases.* It is not sufficient to increase inputs, if losses also increase by the same magnitude. It is also not sufficient to maintain soil organic carbon levels in healthy soils. Maintaining soil carbon levels can avoid further outputs, and therefore be a worthy goal, and one that is important for ensuring that agricultural soils are not a source of greenhouse gases. However, it does not sequester additional carbon, and therefore cannot mitigate other greenhouse emissions.

What do the fundamentals of the carbon cycle mean for climate change mitigation efforts that focus on building soil organic carbon in cropland soils in the Pacific Northwest?

Implication #1: Most of the increase in carbon stocks resulting from a management change occur in the first decade or two after the change

The amount of new soil organic carbon stored due to a change in management will vary over time, because there is an upper limit to the amount of carbon that can be stored in soils. Tracking changes in soil organic carbon over time is best illustrated through long-term experiments, of which there are few anywhere in the world. One such long-term experiment in our region is in Pendleton, Oregon. Another, which will be used here to illustrate this point, is in Rothamsted, in the United Kingdom (Jenkinson et al., 1990). Plots were established in 1843 on previously cropped ground, and three different treatments were applied to continuous winter wheat:

- Annual additions of inorganic fertilizer
- Annual additions of farmyard manure
- No soil amendments

Plots receiving farmyard manure gained carbon over time, while the other two treatments maintained or lost carbon (Figure 3). The field data collected from these soils (the symbols in Figure 3) were complemented with modeling studies. The outputs of the models (the lines in Figure 3) allow a more detailed visualization of how soil organic carbon changed through time, and shows how the rate of change of plots where manure was added—the slope of the FYM line at different points in time—decreased through the life of the experiment, as soil organic carbon increased. Models can also help predict the steady state values that such soils might achieve.

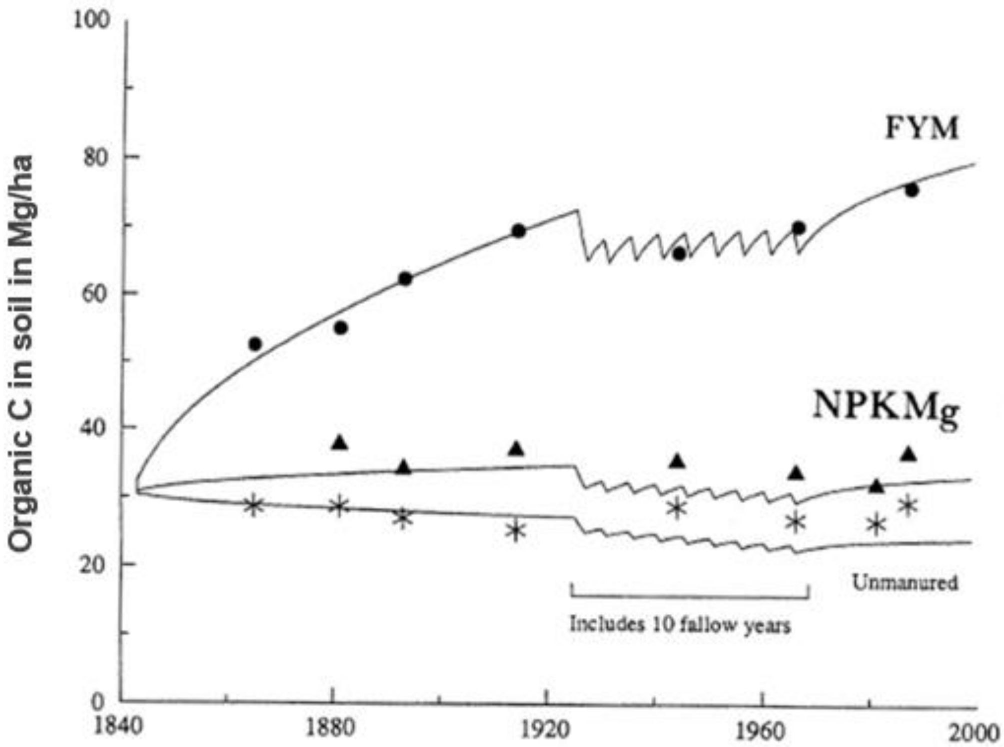

Figure 3. Results of a long-term experiment in winter wheat crops in Rothamsted, United Kingdom. Plots were untreated (stars), had annual additions of inorganic fertilizer (triangles), or annual additions of farmyard manure (circles). Symbols represent experimental data and lines represent modeled data. Figure reproduced from Jenkinson et al. (1990).

Implication #2: Soil organic carbon storage potential varies geographically

Soil organic carbon varies across the Pacific Northwest. For example, there is a strong environmental gradient from west to east across the region, with high annual precipitation on the west side of the Cascade Mountains, low precipitation just east, in the mountains' rain shadow, and then increasing amounts of precipitation further east-northeast across the region. These environmental gradients, which extend well beyond the Pacific Northwest, not only influence the maximum amounts of carbon that can be stored in soils, but also influence the management strategies that are appropriate at each location. Both of these factors affect the potential for sequestering carbon in cropland soils.

Beyond this, the carbon levels of soils at the time that particular practices are implemented can have very significant impacts on the ability of those soils to store *additional* carbon. Results of a modeling study that examined the carbon benefit over the first 50 years after adopting no-till management for two locations in the inland Pacific Northwest—Colfax, Washington and Heppner, Oregon—illustrate this variation (Kemanian and Stockle, 2010). Colfax is a relatively wet location for the inland Pacific Northwest, with higher residue production than the drier Heppner site. In addition to looking at the change in management from conversion to no-till, they added one additional important variable: how much organic carbon was in the soil before management changed to no-till; that is, how full the soil carbon pool (the bathtub) was to begin with. The team ran the model a number of different times for each location, using a range of different values for initial soil organic carbon. These values were selected based on the range of soil carbon values found in each area, as documented in SSURGO, a soils database developed by the US Department of Agriculture's Natural Resources Conservation Service (https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/?cid=nrcs142p2_053627).

When the same initial soil organic carbon levels were used at both locations (for example, 6 g kg^{-1} of soil, in Figure 4), there was a greater change in soil carbon in Colfax (approximately 14 Mg ha^{-1} in 50 years) than in Heppner (approximately 5 Mg ha^{-1} in 50 years), illustrating how soil carbon benefits vary based on regional-scale factors such as precipitation and the resulting differences in the amount of residue inputs to the soil. However, when they compared results across the range of initial soil carbon values, they found that within each location, soils would gain carbon over 50 years (positive change) if the initial soil carbon was low, but would lose carbon (negative change) if soil organic carbon was initially high (Figure 4). The magnitude of these differences led the researchers to conclude that initial soil organic carbon levels at each site were more important than the differences in residue inputs between the two sites—a reflection of the difference in precipitation—in terms of determining the changes in soil organic carbon levels (Kemanian and Stockle, 2010).

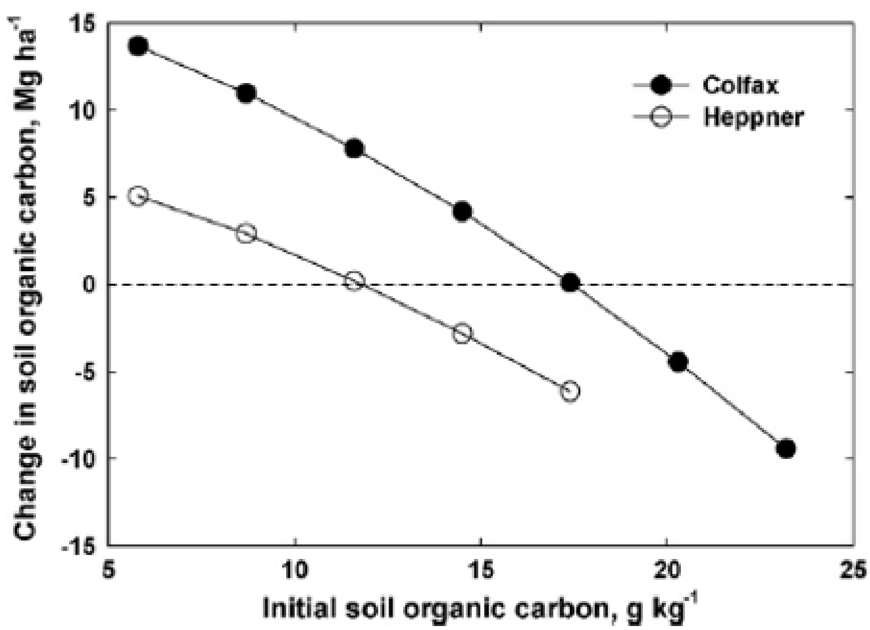

Figure 4. Modeled changes in soil organic carbon over 50 years after adoption of no-till at two inland Pacific Northwest locations. Changes in soil organic carbon were modeled for different values of initial soil organic carbon expected at each location. Figure reproduced from Kemanian and Stockle (2010).

Beyond carbon – Additional principles of cropland's greenhouse gas emissions

Management changes that impact soil carbon can also impact other agricultural greenhouse gases. For example, reducing the amount of tillage might reduce decomposition (carbon outputs), increasing the amount of soil organic carbon. But this reduction in tillage might also:

- Reduce fuel use, as fewer tractor passes are made over the field. This would further reduce greenhouse gas emissions.
- Change emissions of nitrous oxide from the soil, where the direction and magnitude of the change depends on soil conditions, which are influenced by tillage.
- Change emissions from fertilizer and pesticide production, as different fertilizer and pesticide application rates may be used.

Zaher and colleagues completed a life-cycle assessment study in an effort to evaluate the potential carbon benefits from no- and reduced-tillage when fertilizer production and use of machinery are considered as well (Zaher et al., 2013). They found that accounting for fuel consumption, emissions of nitrous oxide, and up-stream emissions from fertilizer production have significant impacts on the total carbon footprint of the agricultural system (Figure 5) – and in this particular study, strategies that tended to store more carbon in soils also tended to have lower net emissions compared to the baseline conventional tillage scenarios. Note that to examine the impact of a management change, it is necessary to compare the emissions between management scenarios at a *single* site – so, for example, comparing LCA scenario #3 (no till) to LCA scenario #1 (conventional tillage at the same high rainfall site). This study highlights the importance of understanding the full production system, and establishing the boundaries of the system or potential "leakage" factors based on the question that is being asked. However, this is one of just two regional studies to date to look at impacts of management changes on greenhouse gases from this comprehensive of a set of activities.

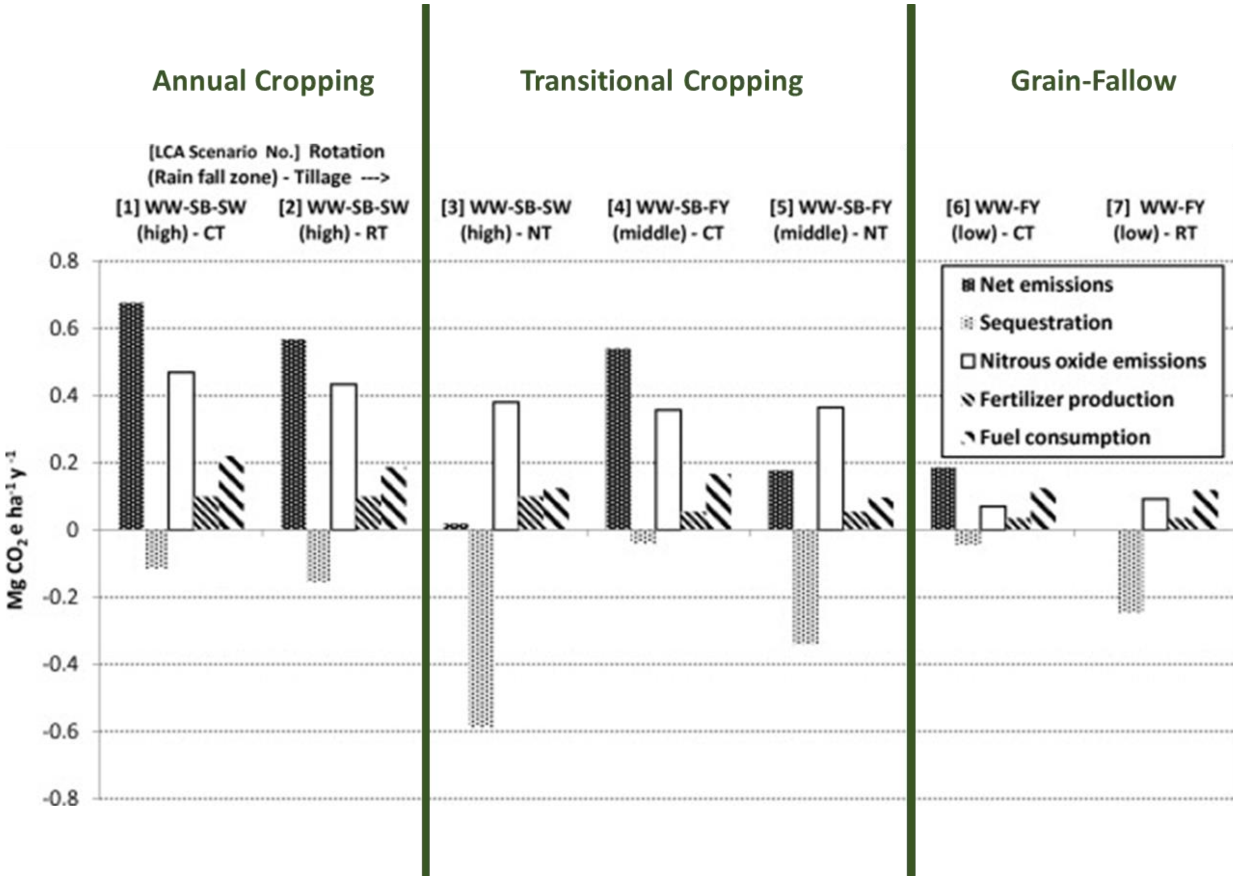

Figure 5. Greenhouse gas emissions from a life-cycle assessment study to evaluate the potential carbon benefits from no- and reduced-tillage when fertilizer production and use of machinery are considered as well. CT = conventional tillage, RT = reduced tillage, and MT = no tillage. Figure reproduced from Zaher et al. (2013) and modified slightly.

The potential for changes in management to result in changes in nitrous oxide emissions from soils presents a particular problem. Nitrous oxide's global warming potential is 298 times higher than carbon dioxide's, for equivalent mass (IPCC, 2007). Nitrous oxide emissions thus represent a significant challenge, as changes in nitrous oxide emissions that are negligible from an agronomic perspective can represent a substantial impact from a greenhouse gas perspective. The impacts of changes in management on nitrous oxide emissions are not well understood, and significant obstacles still exist for effectively quantifying such impacts of a management change. These include:

- An incomplete understanding of how nitrous oxide is produced in soils (Venterea et al., 2012).
- High variability in measurements of nitrous oxide fluxes from soils, with variability both from location to location and from one point in time to another (Henault et al., 2012; Nicolini et al., 2013).
- Difficulty in accurately measuring nitrous oxide emissions from soils (partly due to the variability identified above), which leads to difficulty in tracking changes in emissions due to changes in management (Henault et al., 2012). However, recent advances in

methodology (e.g. Waldo et al. 2016) continue to improve our ability to measure nitrous oxide.
- In the inland Pacific Northwest, there are very few direct measurements of nitrous oxide emissions from agricultural systems, and existing studies range in terms of the conclusions they reach as to how nitrous oxide emissions in this region compare to climate-change related benchmarks (e.g. Stockle et al., 2012; Waldo et al., 2016).

These challenges not only complicate an understanding of how changes in management focused on storing soil carbon may impact soil nitrous oxide emissions – but they also currently complicate efforts to quantify whether management practices focused on nitrogen management can predictably and consistently reduce nitrous oxide emissions. So, although it is an important topic, it remains an unresolved issue in the science-policy nexus.

Agriculture in the Pacific Northwest

Agriculture in the Pacific Northwest is diverse and productive, and varies across the region, following variations in climatic, soils, and geographic characteristics. Regional production includes over 300 agricultural products (WSDA, 2016), ranging from milk and livestock, fruits and vegetables, to grains and forage. In 2017, the market value of agricultural production was estimated at $9.6 billion in Washington, $7.6 billion in Idaho, and $5.0 billion in Oregon (USDA NASS, 2019), and is a fundamental part of the economy, communities, and culture of all three states.

Production systems also vary, including extensive and intensive dryland production, irrigated agriculture, organic production, livestock and dairy production, and more. The coastal portion of the region, west of the Cascade Range, is generally characterized by deep soils and a mild, maritime climate, and produce a wide range of fruits and vegetables. As mentioned earlier, few comprehensive studies exist on the carbon sequestration potential in the western portion of the Pacific Northwest.

East of the Cascades, and inland through the Snake River Plains and towards the Rocky Mountains, the climate is drier and more continental, with warmer summers and colder winters. Precipitation is lowest in the direct rain shadow of the Cascades and the high plains of southern Oregon and Idaho, and increases eastward and northward, respectively, towards the Rocky Mountains (Douglas et al., 1992). Soils also vary across the region, following patterns of glacial and glacial flood silty deposits transported to the region by prevailing westerly winds tens of thousands of years ago (Schillinger et al., 2010; Phillips, ND). Researchers have classified this part of the region—known as the inland Pacific Northwest—into agro-ecological classes where the climatic, topographic, and soils differences across the regions have led to distinct agricultural patterns. Dryland agricultural production has been classified into three major agro-ecological classes (Huggins et al., 2014b):

- Grain-fallow, defined as areas with greater than 40% fallow,
- Annual crop-fallow transition (transition), with 10 to 40% of the area using fallow, and
- Annual crop, with less than 10% fallow.

The differences among these classes—and the cropping systems and geographic regions they are currently associated with—affect the carbon sequestration potential of soils, so it is important to analyze studies within the context of each class. Some of the studies reviewed in this publication (e.g. Brown and Huggins 2012) use an older classification system. Readers who would like a more detailed description of this classification should see Brown and Huggins 2012 and citations therein.

Irrigation is another driving factor influencing agricultural systems in the region, leading to an additional agro-ecological class (Huggins et al., 2014b). However, overall, there are fewer data available for the irrigated region than for the dryland agro-ecological classes.

Types of scientific evidence used to understand the impacts of strategies on carbon sequestration in cropland soils in the inland Pacific Northwest

In general, there are three types of research that provide evidence about the impacts of various management strategies on carbon sequestration in agricultural soils in the Pacific Northwest. The first two approaches involve experimental measurements of conditions in the field (soil and gas exchange), while the third approach involves the use of simulation models to predict effects of management strategies:

- <u>Sampling of soil carbon stocks</u>, where data is collected from soil samples taken from research experiments or from cultivated fields on commercial farms. After sampling, carbon content is determined using one of a number of methods, and the relationship between carbon content and other factors is explored, to draw conclusions on the impact of different strategies on carbon sequestration. These other factors can include the biophysical context, management practices, cropping history, and other variables. Valuable studies that examine soil samples from our region include Brown and Huggins (2012), who summarized the results of existing field experiments on carbon sequestration in dryland systems in the inland Pacific Northwest (see sidebar, *Review of Experimental Data Sets*) where it was possible to identify the sampling methodology from the original study. A long-term experiment established in 1931 in Pendleton, Oregon (Rasmussen and Smiley, 1997; see sidebar, Pendleton Long-Term Experiments) is also particularly valuable, given that changes in soil carbon can take decades to occur and stabilize.
- <u>Measurement of trace gas exchange</u>, which can be accomplished by a number of methods, including chambers, eddy covariance, or other methods. Chambers are used to measure fluxes at the point or plot scale, over relatively small areas that can be covered and are usually utilized to compare fluxes across multiple treatments within an experiment. Eddy covariance systems measure fluxes at the field scale using an array of sensors deployed on towers above specific fields of interest. These sensors quantify small variations in the concentrations of gases in the air—such as carbon dioxide—above each field, and these variations allow researchers to quantify and infer changes in the relative fluxes of carbon

(and other elements) to and from the plants and soil. Both chambers and eddy covariance methods have been used in the Pacific Northwest.

Modeling studies, where biogeochemical models—which synthesize current empirical understanding of the processes that govern the flows of water, carbon and other nutrients between the atmosphere, the crops and the soil, and quantify the size of the pools and the rates of the flows—are used to quantify carbon inputs, outputs, and pool sizes under defined environmental and management conditions. Modeling studies depend on existing experimental measurements (e.g. the two approaches noted above) and can be used to look at carbon changes over wider spatial and time scales than can be measured experimentally. Though there are multiple models used across the United States for these purposes, in the inland Pacific Northwest, these studies have most often, but not always, used a cropping system model developed and rigorously tested in this region – CropSyst (Stockle et al., 1994, 2003; See sidebar, CropSyst – a simulation model).

Understanding something about each of the research approaches is important for understanding and interpreting the resulting data. Note that sampling and analytical methods both impact results, and therefore great care needs to be taken when comparing or aggregating results from studies using different methodologies. And it is almost certain that ongoing integration of these three approaches will be critical for establishing a comprehensive understanding of regional soil carbon dynamics and improving our capacity to predict soil organic carbon levels under future climate and management scenarios.

Carbon sequestration in Pacific Northwest cropland soils – Existing evidence

This section summarizes the existing studies that provide evidence about the effects of different practices or strategies on soil organic carbon, with the intent of supporting informed inferences about the potential for increasing carbon sequestration in agricultural soils in the inland Pacific Northwest. As discussed earlier, the difference between soil carbon in agricultural lands and those under native vegetation may be very different for dryland and irrigated agriculture. First, we discuss evidence in dryland systems of the inland Pacific Northwest, which has been a major focus of recent research, with practices that have more robust evidence discussed first. Then we synthesize what is known in inland Pacific Northwest irrigated systems, which have not yet been studied as extensively.

Dryland cropping systems – the role of cropping intensity and tillage across the dryland environmental gradient

Reducing Fallow: Cropping intensity and resulting residue inputs matter, so eliminating fallow where feasible can increase soil organic carbon.

Across much of the dryland region, cropping systems include fallow as a long term response to water limitations. In the grain-fallow region, land is normally cropped to winter wheat every other year, and maintained as fallow (no crop) in the alternate years. In such grain-fallow cropping systems, the land lies fallow every other year, and thus there are no carbon additions during those years. Meanwhile, microbial activity continues, and may increase due to the availability of additional water during the fallow year. Fallow ground is also normally tilled for weed control, contributing to carbon losses. Data from the Pendleton long-term experiments illustrate clearly the difficulty of maintaining soil carbon in a winter wheat-fallow system, as all but one treatment lost soil carbon (Table 2; Figure 7; Machado 2011). Within this experiment, note that adding nitrogen fertilizer increased yields and therefore carbon inputs but only slowed the loss of carbon.

Table 2. Crop residue management effects on soil organic carbon (SOC) in a winter wheat-summer fallow system, at the Pendleton long-term experiment. Reproduced, with modification, from Machado (2011).

N† kg ha^{-1}	Burn	SOC 1976‡ Mg ha^{-1}	SOC 2005‡ Mg ha^{-1}	Significance level§
0	No burn	64.8ab	56.1c	***
45	No burn	63.5ab	55.6c	**
90	No burn	64.3ab	57.6bc	*
Pea⁋	No burn	69.6a	61.3b	**
Manure#	No burn	69.4a	68.8a	ns††
0	Fall burn	59.1b	48.9d	**
se		2.3	1.5	

* Indicates that SOC means in 1976 and 2005 are significantly different at the 0.05 probability level.

** Indicates that SOC means in 1976 and 2005 are significantly different at the 0.01 probability level.

*** Indicates that SOC means in 1976 and 2005 are significantly different at the 0.001 probability level.

† N, nitrogen; SOC, soil organic carbon.

‡ Within columns, means followed by the same letter are not significantly different according to LSD (0.05).

§ This compares SOC across years.

⁋ Pea vines = 2.24 Mg ha^{-1} field weight; 87.8% dry matter; 0.82 Mg C ha^{-1} and 0.037 Mg N ha^{-1}; applied 1-3 days before plowing.

Manure = 22.4 Mg ha-1 wet wt; 47.5% dry matter; 1.69 Mg C ha-1 and 0.14 Mg N ha-1; applied 1-3 days before plowing in April or May of plow year.

††ns = not significant.

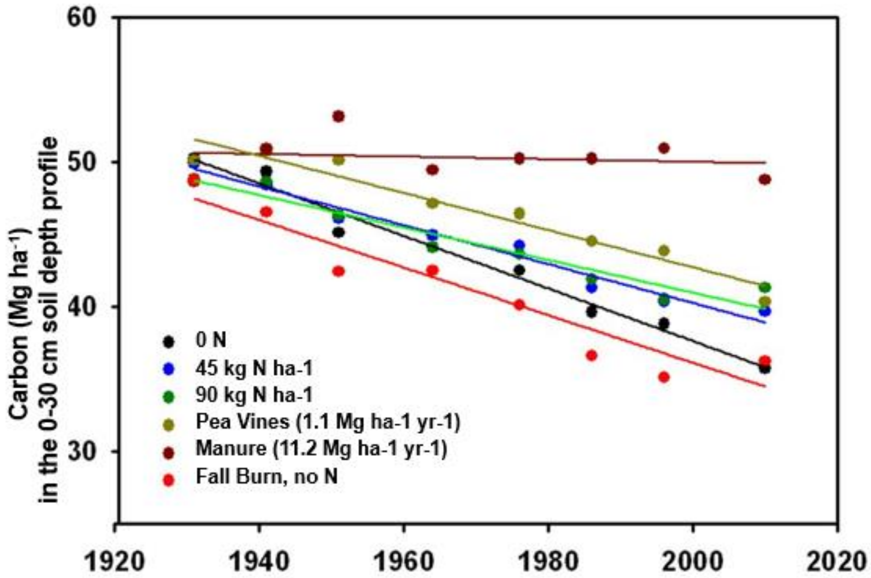

Figure 7. Soil carbon in the top 30 cm of soil in plots in the long-term residue management experiment at Pendleton, Oregon (1931-2010) (Machado et al., unpublished data)

Waldo and colleagues' eddy covariance study reached similar conclusions on the impact of fallow years on cropland soils' ability to sequester carbon. They concluded that during wheat-growing years both high and low rainfall sites (Pullman and Lind, respectively) were sinks for carbon, but that fallow years at Lind were close to neutral (Waldo et al., 2016). In theory this would mean that eliminating fallow would increase the carbon sequestration at Lind. However, it is important to highlight once again the interaction with climatic conditions: rainfall at Lind is generally not sufficient to support annual wheat crops, which is why fields are fallowed in alternate years. It is also important to note that these results, though based on frequent measurements during the year, are only from one or two seasons under each crop. The researchers point out the need for longer-term studies to fully understand the potential of soils as long-term carbon sinks (Waldo et al., 2016).

In both the grain-fallow and transition cropping areas, there may be limited but real opportunities for intensification through lengthening rotations to reduce fallow, or by opportunistically replacing fallow with a spring crop in years with adequate moisture and when other conditions are favorable (Kirby, 2017; Lutcher, 2009). Other avenues for cropping intensification, such as perennial cropping, in some cases cover cropping, are discussed below.

No-till and reduced tillage: Reducing or eliminating tillage can increase soil organic carbon, particularly in the higher rainfall, annual cropping agro-ecological class. Conversion from conventional tillage to no-till has greater impact than reduced tillage, and carbon gains occur mainly in the first decade after the change is implemented.

Brown and Huggins (2012), in their synthesis of existing soil carbon datasets on carbon sequestration in agricultural soils of the inland Pacific Northwest, analyzed the impact of conversion from conventional tillage to no tillage or to reduced tillage. Conversion to no-till generally resulted in soil organic carbon gains in the top 30 cm (11.8 in), where the majority of carbon accumulates when soils are not turned over.

The amount of carbon gained in response to an elimination of tillage varied across the dryland production region. In the annual cropping zone (ACZ 2 in Table 1), soil carbon stocks across the soil depth profile increased an average of 0.71 (± 0.63) Mg C ha^{-1} yr^{-1} (1.05 ±0.93 MT CO_2e $acre^{-1}$ yr^{-1}) over an average of 14 years following conversion from conventional tillage to no-till, with all changes occurring within the surface 20 cm (Table 1; Brown and Huggins, 2012). In the transition zone (ACZ 3 in Figure 8 and Table 1), increases in carbon across the soil profile were less, averaging 0.21 (± 0.10) Mg C ha^{-1} yr^{-1} (0.31 ±0.15 MT CO_2e $acre^{-1}$ yr^{-1}) in the surface 20 cm over an average of 10 years following conversion (Table 1; Brown and Huggins, 2012).

Given the relatively high standard deviation for these data, a cumulative probability analysis was used to define expectations for soil organic carbon changes. Cumulative probabilities allow for an evaluation of results based on the probability of their occurrence. For example, a change in soil organic carbon of 0.21 Mg C ha^{-1} yr^{-1} (0.31 MT CO_2e $acre^{-1}$ yr^{-1}) or more would be expected on 75% of annual cropping sites over the first 14 years following conversion (Table 1, see cumulative probability estimates for no-till).

Due to a lack of experimental data from the winter wheat-fallow class, they were not able to estimate an average impact or carry out a probability analysis in the grain-fallow class (ACZ 5). However, in the one study they located, soil organic carbon was observed to accumulate at a rate of less than 0.07 Mg C ha^{-1} yr^{-1} (0.10 MT CO_2e ac^{-1} yr^{-1}) with no appreciable gains observed below 5 cm following 14 years of no-till (Bezdicek et al., 1998, as cited in Brown and Huggins, 2012).

Table 1. Changes in soil organic carbon (SOC) across studies evaluating changes in tillage and inclusion of perennials in the crop rotation, in the inland Pacific Northwest. Reproduced from Table 2 in Brown and Huggins (2012).

Management	Agroclimatic Zone	Number of studies	Period covered by data (mean y)	Mean SOC change (Mg C ha^{-1} yr^{-1})[a]	Cumulative probability of SOC change (Mg C ha^{-1} yr^{-1})[b] 25th	50th	75th
No-tillage	2 (annual cropping)	12	14	0.71 (±0.63)	0.21	0.64	1.04
	3 (transition)	5	10	0.21 (±0.10)	0.12	0.19	0.25
Mixed perennial-annual	2 (annual)	8	12	1.03 (±0.41)	0.69	0.94	1.12

[a] Values in parenthesis indicate plus or minus one standard deviation from mean value.

[b] The 25th, 50th, and 75th percentiles of the cumulative probability function.

Brown and Huggins note that the 0.71 Mg C ha^{-1} yr^{-1} estimated for the annual cropping class is at the extreme high end or exceeds other national and global estimates for conversion from conventional tillage to no-till (Smith, 2004; West and Post, 2002; Follett, 2001; West and Marland, 2001; Paustian et al., 1997, all as cited in Brown and Huggins, 2012). In contrast, the 0.21 Mg C ha^{-1} yr^{-1} estimated for the annual crop fallow transition class is similar to lower rates reported in other studies (Liebig et al., 2005; Smith, 2004; Follett, 2001; West and Marland, 2001; Paustian et al., 1997, all as cited in Brown and Huggins, 2012). Brown and Huggins attribute the high estimates in the annual cropping class and the large range and standard deviation to the influence of sampling biases (e.g. sampling just after residue incorporation) and soil erosion processes.

Brown and Huggins (2012) found limited regional datasets addressing changes in soil organic carbon with the adoption of reduced tillage: five datasets, mostly in the transition zone. However, the available data did indicate that gains in soil carbon when converting to reduced tillage were less than for conversion to no-till. In the annual cropping area, the use of reduced tillage resulted in a 0.045 Mg C ha^{-1} yr^{-1} (0.07 MT CO_2e $acre^{-1}$ yr^{-1}) gain in soil organic carbon in the surface 15 cm (Brown and Huggins, 2012).

In the transitional cropping area, reduced tillage resulted in a relatively large increase in the surface 7.5-cm that declined to near zero between 7.5- to 22.5-cm. However, the change in SOC compared to CT increased at 22.5 to 45-cm depth and may indicate that RT is more efficient at increasing SOC stocks in the subsurface compared to NT (Brown and Huggins, 2012). At a national and global level, where carbon accumulates in the soil's depth profile is a key question that leads to uncertainty about whether no-till and reduced tillage lead to overall soil carbon accumulation, or simply a redistribution in depth, with increases in the top 20 to 30 cm, and decreases below that (Syswerda et al., 2011; Powlson et al., 2014).

Stockle and colleagues investigated the impact of converting cropland from conventional tillage to no-till and reduced tillage in the inland Pacific Northwest through a modeling approach using CropSyst (Stockle et al., 2012). They simulated the changes in soil organic carbon at three dryland locations with varying amounts of rainfall—Lind, in the grain-fallow class, St. John, in the annual crop-fallow transition class; and Pullman, in the annual cropping class. The team looked at average changes in soil carbon during the first 12 and 30 years after the change in tillage. To accommodate real-world variability, the modeling scenarios were completed twice, using a lower and upper boundary value to represent the range of possible impacts that tillage could have on oxidation rates of agricultural soil carbon. Together, the two sets of results can thus be interpreted as showing a range of possible expected changes in soil organic carbon.

In general, modeling results were consistent with the lower mid-range of results found by Brown and Huggins (2012) described above. Annual changes in soil organic carbon after conversion to no-till were all positive, with conversion to no-till in the annual cropping class (Pullman) producing the largest increase in soil carbon (Figure 6). Change in soil organic carbon was less with lower annual precipitation and greater fallow frequency. Again, in accordance with the field data summarized by Brown and Huggins (2012), Stockle et al. (2012) found that the rate of change in soil organic carbon decreased over time, with the increases almost all occurring within the first 12 years after conversion (Figure 6).

Annual changes in soil organic carbon after conversion to reduced tillage were also almost all positive (Figure 6). Reduced tillage includes a wide range of tillage strategies in the inland Pacific Northwest, which can result in a corresponding wide range of carbon gains estimates. As with conversion to no-till, the magnitude of values obtained by Stockle et al. (2012) was largely consistent with the values found by Brown and Huggins (2012).

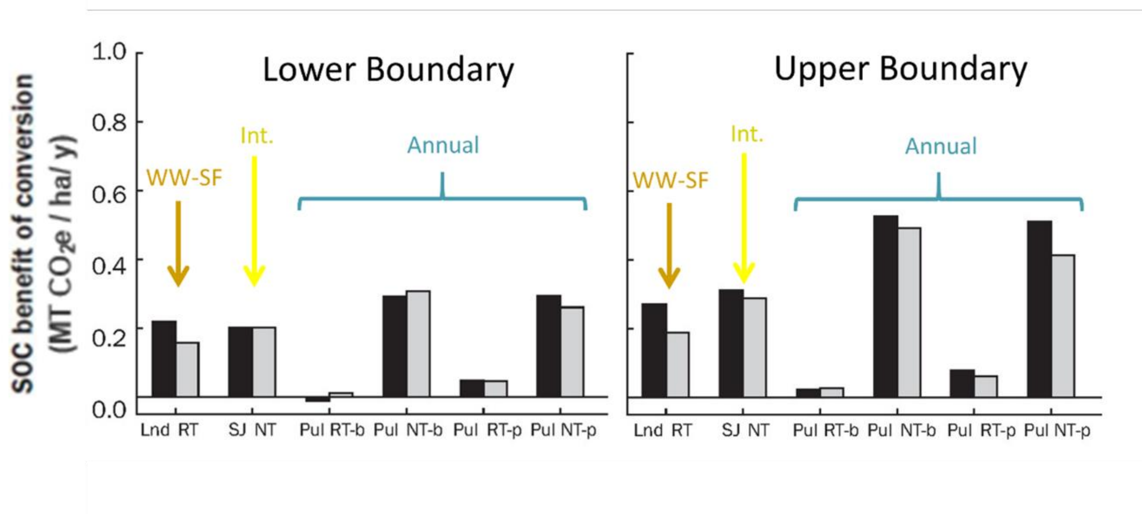

Figure 6. Modeled changes in soil organic carbon in the top 30 cm in soils 12 years (black bars) and 30 years (gray bars) after conversion to reduced tillage (RT) or no-till (NT), in different agro-ecological classes and with different rotations. The left panel shows the lower boundary of soil carbon changes, and the right panel shows the upper boundary. Reproduced from Stockle et al. (2012).

In results that illustrate the important interaction between tillage and fallow frequency, Albrecht and colleagues (2008) compared the effects of reducing fallow to those of reducing tillage. Several different experiments were established at the Columbia Basin Agricultural Research Center in Pendleton, Oregon between 1931 and 1981. Albrecht et al. (2008) compared data from these experiments, extrapolating the data as if all experiments had been initiated at the same time, in 1931 (though they were not) (Figure 8). They found that the winter wheat-fallow systems had lost carbon over time, and there was little difference between the sites that were tilled with a moldboard plow and those that utilized a less intense sweep plow operation. In contrast, the continuous wheat systems (in which wheat was planted every year) maintained or even increased (depending on tillage practices) their carbon content over time, as carbon-rich crop residues are added to the system each year.

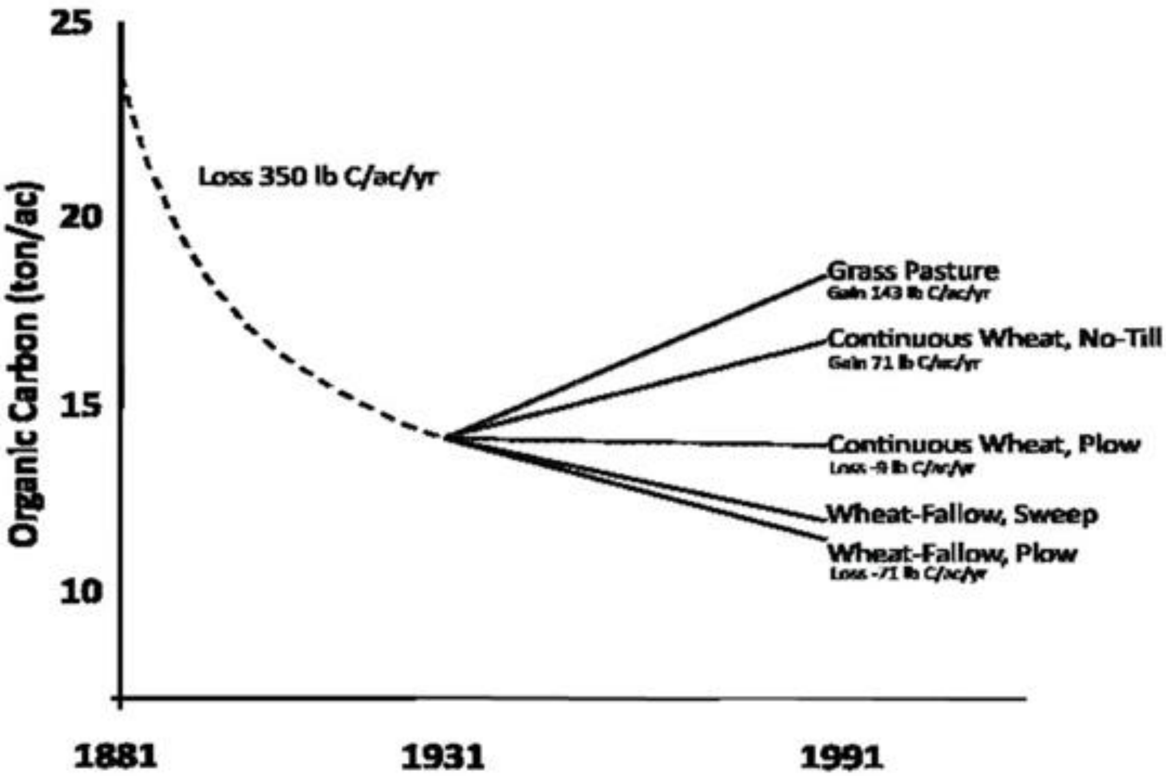

Figure 8. Extrapolated trends in soil organic carbon in the Pendleton, Oregon long-term experiments. In this figure, data are shown as if the experiments were all established in 2931, though they were not. Actual establishment dates are as follows: Plowed (conventional tillage) continuous wheat was established in 1931. The plowed and sweep tillage (reduced tillage) wheat-fallow was established in 1940. No till continuous wheat was established in 1981, and was in place for just 10 years prior to Rasmussen and Albrecht, so it's possible these numbers are somewhat inflated. Figure reproduced from Albrecht et al. (2008).

In summary, these data indicate that opportunities to build soil organic carbon seem to lie mostly in higher intensity annual cropping systems (where moisture is less limiting, and yields, residues and carbon inputs are greater and occur every year). Tillage appears to be one important factor affecting the magnitude of the carbon sequestration potential.

Dryland cropping systems – other strategies for increasing soil C sequestration

Perennial crops: Where economically and biologically feasible, mixed perennial-annual systems could increase soil organic carbon.

Perennial cropping is another form of increasing crop intensity, as it results in living plants growing continuously through the year, and root systems that are generally much more extensive. Brown and Huggins (2012) analyzed the impact of adding perennial crops to an otherwise continuous annual crop rotation (Table 1). These mixed annual-perennial rotations include crops such as alfalfa or grasses (bluegrass, wheatgrass, alfalfa-grass mix) in an otherwise annual wheat rotation. Compared to annual cropping systems, mixed perennial-annual systems in the annual cropping area increased mean soil organic carbon in the soil profile by 1.03 (± 0.41) Mg C ha^{-1} yr^{-1} (1.53 ± 0.61 MT CO_2e $acre^{-1}$ yr^{-1}). Though this value is somewhat higher than estimates made by other groups (e.g., Liebig et al., 2005; West and Post, 2002; Follett, 2001;

Paustian et al., 1997), those other estimates still suggest that shifting to a mixed annual-perennial rotation can provide soil carbon benefits that are equivalent to, or higher than, the benefit obtained from a conversion to no-till in annual systems. Such mixed annual-perennial systems are not common in the inland Pacific Northwest due to economic and market factors, as well as logistical barriers (for example, harvesting alfalfa requires different equipment, leading to greater investment needs). However, successful models used by some innovative growers include, for example, growing dryland alfalfa or other perennial crops in rotation with wheat (Yorgey et al., 2017c; Yorgey et al. 2019b).

It should be noted that changes in crop rotations in localized areas may not lead to global impacts on C sequestration, if reductions in production of a specific crop in one area are offset by increases elsewhere, due to economic forces. We are not aware of studies that examined the landscape-level impact of mixed perennial-annual cropping compared to isolating annual cropping and perennial cropping on specific acreage.

Soil amendments: Additions of carbon-rich materials can maintain, or potentially increase, soil organic carbon. The amount of carbon added is an important determinant of the amount of carbon stored in soils, and properties of both the amendment and the soils to which it is applied are likely important determinants of the amount of carbon stored over time. Continued research into different types of amendments is also important because these amendments can have other effects on the production system or on the environment, as well as the potential to help solve other management challenges (e.g., manure management challenges for some livestock operations).

One of the more promising insights arising from the Pendleton long-term experiments is that the treatment that amended soils with manure maintained (though did not build) soil carbon over the length of the experiment, while all other treatments in this winter wheat-fallow system lost soil carbon (Table 2; Figure 7; Machado et al. 2011). Further analysis (not shown here; Machado 2011) suggests that the organic carbon levels in the soils are related to the amount of carbon added in crop residues and amendments – meaning that the rate at which the manure or other amendment is applied is a key determinant of the impact on soil carbon.

Organic amendments can increase carbon content in soils directly as well as indirectly by increasing primary productivity. A number of different materials, including manures, municipal biosolids, compost, and biochar, have been investigated as potential amendments, with one important question being how much of the carbon in specific types of amendments results in additional carbon storage in soils.

Generally speaking, carbon stability is dependent on how the properties of the amendment interact with the biotic and abiotic characteristics of the soil environment (Schmidt et al. 2011). Research suggests that generally speaking, for a given amendment, soils with the lowest initial carbon levels have the highest rates of amendment carbon storage (Brown et al. 2011). Meanwhile, comparing different amendments, results from an experiment on a Walla Walla silt loam soil near Pendleton, Oregon showed that biosolids and cattle manure were more efficient at sequestering carbon than other amendments (Table 3). Gains in stable soil carbon may be related to amendment quality, such as phosphorus and sulfur content of an amendment (Wuest and Reardon, 2016) or the amount of microbial processing that amendment carbon has undergone (Wuest and Reardon, 2016; Wuest and Gollany 2012). The loss of soil organic carbon over the 28 months following tillage to grow annual wheat crops was not strongly related to initial soil organic carbon gain (Wuest and Reardon, 2016).

Table 3. Effect of amendment type on soil organic carbon (C) accumulation in the surface (0–9.8 inches) of a silt loam soil near Pendleton, Oregon. Amendments were applied at similar C rates for five years and sampled 7 years after final amendment application. (Adapted from Wuest and Reardon 2016 by Yorgey et al., 2017a)

Amendment 2.5 Mg C ha^{-1} yr^{-1} [a]	Sequestration Efficiency[b]
	%
Municipal biosolids	49
Cattle manure (no bedding)	21
Alfalfa feed pellets	14
Wood sawdust	11
Composted wheat residue	11
Brassica residue	10
Wheat residue	9
Sucrose	5
Cotton linters	3

[a] Over the five years of annual amendment, a total of 12.5 Mg C ha^{-1} was applied.

[b] Sequestration efficiency is calculated by increase in soil organic carbon compared with the treatment receiving no amendment, divided by the amount of C applied.

In Douglas County, Washington, application of anaerobically digested biosolids every 4 years at 3 rates (0, 4.8, 6.9 and 9.0 dry Mg ha^{-1}, or approximately 0, 1.7, 2.5, or 3.2 Mg C ha^{-1}), to a wheat-fallow system on Timentwa ashy fine sandy loam, was sufficient to maintain wheat yields and grain nitrogen at or above treatments receiving commercial fertilizer as anhydrous ammonia during the fallow year (Cogger et al. 2013). Pan et al. (2017a) evaluated levels of total C in soil from the experiment 20 years after it began (with continued application of biosolids every 4 years) and reported that 91% of biosolids C was effectively retained in the soil, though a smaller amount, 34%, of this was retained as stable (non-hydrolyzable) C. (Note that we have expressed carbon values on the basis of percentage retained over time, rather than on the basis of change per area, due to the fact that the application rate can be varied, and is an over-riding determinant of overall carbon drawdown resulting from this strategy.) The 91% of biosolid C retained in soil is higher than the value of 49% suggested by Wuest and Reardon, which was also similar to the value suggested for climate impacts modeling of biosolids application based on previous observations (Brown et al. 2010). Pan and colleagues suggested that the high soil C accumulation efficiency found in this experiment may be related to the semiarid climate, where surface soil drying during the summer may slow organic decomposition. In this experiment, the high biosolids application rate (9.0 dry MT ha^{-1} of anaerobically digested biosolids applied every 4 years) stored carbon at 0.77 Mg C ha^{-1} yr^{-1} (1.14 MT CO_2e $acre^{-1}$ yr^{-1}) over 20 years (Pan et al. 2017a).

There are few studies documenting carbon sequestration potential of compost application in the inland Pacific Northwest. Cox et al. (2001) found that high rates of compost application (110 t ha^{-1} annually) to hilltops in the Palouse raised soil carbon from 1.2% to 2.9% after the second year of application, but no information was available on longer-term carbon storage. Wuest and Reardon found that 11% of carbon from composted wheat residues was present in soil 7 years after application (Table 3). In a study from dryland wheat system in Utah, Reeves et al. (2012) looked at the residual effects of a one-time application of compost at 50 Mg ha^{-1} (dry weight) and found that sixteen years later, soils from compost-amended plots contained 1.6-fold higher total organic C (1.43% vs. 0.89%). Other organic amendments that are applied in some cropland in the region include livestock manure and fiber resulting from solids separation of dairy manure, though regional results showing long-term carbon sequestration potential of these amendments are limited (see Irrigated cropping systems). A global meta-analysis of 42 studies documenting change in soil organic carbon due to manure application, estimated that 12% (±4) of manure carbon was retained in soil for an average study duration of 18 years (Maillard and Angers, 2014). It should be noted that this meta-analysis included different manure types, applied in a variety of climates and agricultural systems.

Amending soil with biochar has the potential to sequester carbon, due to the recalcitrant nature of biochar carbon, and may also improve soil properties related to crop growth, particularly pH and water holding capacity – though it is important to note that feedstock type and processing conditions strongly influence the resulting biochar characteristics. While biochar carbon is generally much more recalcitrant than carbon in other potential amendments, the amount of biochar carbon stored in soils has also been shown to be variable based both on the characteristics of the feedstock, and of the soils to which it is applied. Laboratory research from outside of the Pacific Northwest examining a variety of potential feedstocks (including cow manure, sawdust, and wheat straw) estimated 21-47% of biochar carbon would be retained in soil long term (Zhao et al. 2013; Windeatt et al. 2014). In a 225-day laboratory incubation experiment, in which biochar from a variety of feedstocks (switchgrass straw, anaerobically digested dairy manure fiber, softwood bark, and wood pellets) was applied to five Washington State soil types, Streubel et al. (2011) found that soil type was more important than feedstock type in terms of increasing soil carbon content, with Quincy sandy loam soil showing a greater increase in carbon content than the four silt loam soil types tested. While there has been some discussion that biochar can, in some cases, hasten decomposition of indigenous soil organic carbon through a ‘priming effect’ (De Gryze et al., 2010), recent literature suggests a small enhancement of soil organic C by biochar amendments to agricultural soils (as further discussed in Amonette 2019).

Amonette (2019) estimated the net climate benefit of a biochar strategy using forestry residues and waste wood as primary feedstocks, land applied to 26 of Washington’s 39 counties, through development and application of a high-resolution scalable model. The algorithm used to perform the assessment is a modification of the Biochar Global Response Assessment Model (BGRAM) implemented in spreadsheet form by Woolf et al. (2010), and considers biomass composition, pyrolysis and combustion process parameters, energy production, C intensity of energy being offset, rate of technology adoption, biochar properties, biomass growth response, biomass and biochar transport, biochar decomposition rates, and greenhouse gas emissions at every stage of the cycle from biomass harvest to 100 years after biochar has been added to the soil. The Washington application also includes spatial integration of soil productivity and crop information at 1 hectare resolution, separate accounting for changes in soil organic C levels resulting from feedstock harvesting and biochar application, and tracking of biochar production and soil storage capacities over time. For each county, seven biomass feedstock and biochar process scenarios were developed including one for waste wood harvested from municipal solid waste (MSW) alone and processed at a central facility, and six for MSW waste wood combined with residual forest biomass from timber harvesting operations. Results find that across the 26 counties, over 100 years, 39 MT CO_2e could be offset by generating biochar from only municipal solid waste. Including forestry residuals resulted in substantially higher values were obtained: 414-1298 MT CO_2e depending on the level of timber harvest and the processing locations assumed.

Cover crops: The inputs of carbon from cover crops can increase carbon sequestration, but its ability to reverse losses depends on other management practices, such as fallowing and crop rotation.

No comprehensive studies exist on the effects of cover crops on carbon sequestration. Brown and Huggins (2012) discussed two studies (Horner et al., 1960; Rasmussen and Parton, 1994, as cited in Brown and Huggins, 2012) where cover crops (called "green manure") did not reverse the loss in soil organic carbon. They concluded that, though using green manure could lead to increases in soil organic carbon, the impact of this practice is dependent on other practices, such as whether there is fallowing or not (Brown and Huggins, 2012). Cover cropping in a dryland context has so far been economically and technically challenging, due to the fact that the cover crop utilizes water in a system where yields for the main cash crop are already water-limited. Ongoing cover crop research in the dryland areas of the Pacific Northwest may generate additional data in coming years about the ability of cover cropping to contribute to carbon sequestration, and insights into whether and how it could be incorporated into economically viable dryland cropping strategies.

It should also be noted that the question of whether cover cropping leads to carbon drawdown from the atmosphere hinges on the question of the role it plays within the larger rotation of crops. If cover cropping intensifies cropping by adding living plants at times that there would otherwise not be (such as by reducing fallow, or via double cropping, the practice of planting two crops in a single growing season), then it could result in overall carbon sequestration. If cover cropping instead replaces another crop in the rotation, then the carbon impacts will depend on the carbon impacts of the cover crop compared to the crop that has been replaced.

Crop rotation: Where possible, including crops that produce higher residues could increase soil organic carbon. The increase will depend on other factors, such as rainfall, cropping intensity, and tillage.

As part of a broader modeling study investigating the impacts of tillage on soil carbon in the inland Pacific Northwest, Stockle and colleagues (Stockle et al., 2012) addressed rotation to some extent. They carried out two simulations in Pullman (the annual cropping class), one with spring barley and one with spring pea as part of the winter and spring wheat rotation. They found that, when spring barley was included, more soil organic carbon was stored in the soil than when spring pea was included, likely because of the increased residue generated from barley compared to pea (Stockle et al., 2012). The differences were most noticeable in no-till sites when focused on the top 15 cm.

Eddy covariance results also suggest rotation may impact carbon sequestration potential. For example, Waldo and colleagues estimated that fields in high and low rainfall sites (Pullman and Lind, Washington) were carbon sinks—that is, were accumulating carbon in the soil—when grown with winter wheat, but were carbon neutral—that is, sequestered or emitted a little carbon, but close to zero—when growing garbanzos (Pullman) or fallowed (Lind) (Waldo et al., 2016). Chi (2016) reached similar conclusions through a combination of eddy covariance and modeling work, though her results also highlight that interactions are important.

It should be noted that if changes in crop rotations in localized areas may not lead to global impacts on C sequestration, if reductions in production of a specific crop in one area are offset by increases elsewhere, due to economic forces. However, based on an understanding of the critical nature of crop residues for building soil carbon, some scientists have also suggested that breeding efforts focused on enhancing crop residues without negative impacts on yields could generate crops that build soil carbon over time (NASEM 2019).

Reduced residue burning and harvest: Practices that incorporate greater amounts of residue into the soil, where they contribute to soil carbon, are preferable to residue burning, from a carbon sequestration perspective. Residue harvesting should also be considered with respect to the tradeoffs in soil carbon sequestration.

The Pendleton long-term experiments include residue burning as a treatment. As mentioned above, only the manure additions led to maintained soil carbon, with the burning treatment showing among the highest losses in soil carbon (Figure 7; Albrecht et al., 2008). Follow-up studies at Pendleton confirmed that soil organic carbon was lowest under a fall burning treatment (with additional losses of 0.05 Mg C ha^{-1} yr^{-1} [0.07 MT CO_2e $acre^{-1}$ yr^{-1}] compared to a similar treatment without burning [Machado, 2011]), and that eliminating burning can improve sustainability of winter wheat production (Ghimire et al., 2015). These results were not unexpected, given that burning the crop residues leads to release of carbon as carbon dioxide, reducing the inputs of carbon into the soil.

Brown and Huggins (2012) discuss two studies that looked at the effects of burning on soil organic carbon (the same ones as discussed green manure: Horner et al., 1960, and Rasmussen and Parton, 1994, as cited in Brown and Huggins, 2012). In both these studies the loss of carbon with burning was accelerated compared to without burning. The difference in carbon loss of 0.15 Mg C ha^{-1} yr^{-1} in the annual cropping class over 10 years but after that point there was no difference between burned and unburned treatments (Huggins and Brown 2012). A study in the annual crop-fallow transition class, meanwhile, found that carbon losses were halved in unburned plots compared to burned plots (Brown and Huggins 2012). Brown and Huggins attribute the differences in these two studies to different lengths of time under burning management and depth of soil sampling. Burning of residues was common until the 1990s, but is now limited in the inland Pacific Northwest (Machado, 2011).

Harvesting residues is a practice that may be used as an alternative to burning in areas where residue production is high. Residues can contribute to sustainable products such as pulp, or to biofuels production. However, removal of straw does remove the associated carbon as well as other nutrients, and thus may exacerbate carbon losses where current cropping systems are already experiencing declines. In contexts where soil carbon is being maintained over time, such as in no-till continuous cropping systems, some harvesting may be possible if harvest leaves enough residues to maintain soil carbon over time (Machado 2011, Huggins et al. 2014a).

<u>*Reduced erosion*</u>*: Though reducing erosion is critical for long term agricultural productivity, the impact of soil redistribution through erosion and deposition on carbon sequestration is not yet well understood.*

Modeling studies in other areas, such as in the United Kingdom and in Maryland, United States, suggest that erosion and deposition could, on balance, lead to a watershed-level increase in carbon sequestration. This increase could be due either to increased carbon sequestration in eroded areas, due to shifting the balance of carbon sequestration in the soil (McCarty and Ritchie, 2002), or to increased rates of carbon sequestration in wetland deposition areas (Quine and Van Oost, 2007). We are unaware of field studies in the Pacific Northwest that specifically investigate the impact of efforts to reduce erosion on the soil's ability to sequester carbon. In the inland Pacific Northwest, however, Stockle et al. (2012), using the CropSyst model, concluded that even if erosion-deposition increased the oxidation of soil organic carbon by an unlikely 50%, erosion would only contribute modestly to changes in soil organic carbon, even under conventional tillage, where erosion values are high.

Evidence relating to dryland cropping systems is summarized in Table 4.

Table 4. Summary of research relevant to estimating the potential for changes in management to increase carbon storage in dryland soils in the Pacific Northwest. To support estimates of the regional carbon sequestration from agricultural croplands, estimates of dryland production in the inland Pacific Northwest were as follows for 2007-2014 (Kirby et al., 2017): 1.44 million acres of annual cropping, 1.85 million acres of transitional, and 2.52 million acres of grain-fallow.

Strategy	Per Hectare Estimates of Carbon Sequestration Potential[1]	Current Technical and Financial Considerations	Additional References Supporting Quantitative Estimates of Sequestration Potential in the Pacific Northwest
Eliminating tillage	**Annual cropping area:** Experimental results indicate 1.05 ±0.93 MT CO_2e ac^{-1} yr^{-1}over an average of 14 years following conversion from conventional tillage to no-till. (Brown and Huggins 2012) (See Table 1 for cumulative probabilities.) Modeling results indicate gains of 0.13 to 0.24 MT CO_2e $ac^{-1}yr^{-1}$; for a rotation in Pullman with barley (a relatively high residue crop) in the rotation over 12 years after conversion. **Transitional cropping area:** Experimental results indicate 0.31 ±0.15 MT CO_2e $acre^{-1}$ yr^{-1}) in the surface 20 cm over an average of 10 years following conversion (Brown and Huggins 2012). (See Table 1 for cumulative	In some parts of the grain-fallow area, no till fallow can lead to more limited near-surface moisture available for seeding winter wheat in late summer (compared to conservation tillage with undercutter), and this can create a substantial yield loss in the following wheat crop (Schillinger et al., 2016). Eliminating tillage may make other carbon-beneficial strategies more practical, by conserving moisture (for intensification), or reducing the time needed to prepare and plant fields. Concerns around herbicide resistance may make the complete elimination of tillage impractical in the future. Shifting tillage strategies generally requires a completely new compliment of equipment and represents an expensive transition. Making the shift may require significant other operational changes (e.g. adding new leases, changing crop rotations, managing new diseases, changing markets etc.) to be profitable.	Estimates of 2012-2013 adoption across the region (percentage of producers, note that this is not an estimate by acreage; Grey et al. 2015): • 30% use no-till • 39% use conservation tillage (either reduced or no-till) • 25% use conventional tillage For additional information on the role of tillage in dryland cropping systems, see Bista et al. 2017.

[1] Some estimates are based on relatively robust data, while others are based on more limited datasets; please see text for further detail.

	probabilities.) Modeling results indicated 0.09 to 0.14 MT CO_2e $ac^{-1} yr^{-1}$. over 12 years after conversion. **Grain-fallow cropping area**: Limited experimental data suggests gains are small. (One dataset indicated accumulations of less than 0.10 MT CO_2e $acre^{-1}$ yr^{-1}) (Bezdicek et al., 1998, as cited in Brown and Huggins, 2012)		
Reducing tillage	**Annual cropping area**: One experimental study indicates an average of 0.07 MT CO_2e $acre^{-1}$ yr^{-1} (Brown and Huggins, 2012). Modeling results indicate an average gain near zero (Stockle et al. 2012). **Transitional cropping area:** Summary data not available. **Grain-Fallow cropping area:** Modeling results indicate gains of 0.1 to 0.12 MT CO_2e $ac^{-1} yr^{-1}$. over 12 years after conversion (Stockle et al. 2012).	See above for technical and financial considerations for changing tillage.	See above for estimates of current adoption of conservation tillage in the inland Pacific Northwest. For additional information on the role of tillage in dryland cropping systems, see Bista et al. 2017.
Amendments (manures, biosolids, composts, biochar)	**Biosolids**: 47-91% of applied carbon retained as stable C long term (Brown et al. 2010; Pan et al. 2017; Wuest and Reardon 2016). At the	While research has suggested that transportation does not have a substantial impact on GHG balance for amendments (Brown et al. 2010), transportation costs are a significant financial constraint to the availability of amendments for many dryland	Washington State Biomass Inventory (Ecology 2011) provides estimates of a wide variety of potential amendment materials, following the methodology of Frear et al. 2005.

	high rate of application (9.0 dry MT ha^{-1} [4.0 t $acre^{-1}$ every 4 years), carbon was stored at a rate of 1.14 MT CO2e $acre^{-1}$ yr^{-1} over 20 years (Pan et al. 2017a). **Manure:** 21% of carbon stored long-term in soils 7 years after application every year for 5 years (manure with no bedding) (Wuest and Reardon, 2016). **Compost:** 11% of carbon stored long-term in soils 7 years after application every year for 5 years (composted wheat residues) (Wuest and Reardon, 2016). **Biochar:** Laboratory studies suggest that 21-47% of biochar carbon would be retained in soil long term (Zhao et al. 2013; Windeatt et al. 2014).	crops in the PNW. The fact that these crops are relatively low value on a per-acre basis further increases the economic barrier. There is relatively little regional field research investigating long term carbon storage from amendments and very little field research on application of biochar to regionally relevant crop and soil combinations. Food safety concerns limit timing for manure applications to some potential receiving crops (especially produce generally consumed raw).	For additional information on amendments and their potential role in dryland cropping systems, see Yorgey et al. 2017a. Amonette et al. estimated total potential impacts from biochar application across 26 of Washington State's 39 counties, over 100 years, could offset 40.3 MT CO_2e by generating biochar from only municipal solid waste. Including biochar from forestry residuals resulted in 414-1298 MT CO_2e depending on the level of timber harvest and the processing locations assumed.
Intensification of cropping / reducing or eliminating fallow	No systematic quantitative estimate. Based on data across experiments in the intermediate cropping zone, at the Pendleton Columbia Basin Agricultural Research Center, Albrecht et al.	Fallow typically only occurs in the drier parts of dryland cropping systems, and are a response to water limitations, which may limit opportunities to intensify cropping or make intensification more risky. Some innovative producers continue to develop appropriate strategies (Yorgey et al. 2016a; Yorgey et al. 2016b). Intensification of cropping (e.g. through double-cropping) may also provide opportunities in irrigated annual cropping systems.	For additional information on this strategy and its role in dryland cropping systems, see Kirby et al. 2017.

	estimate difference of 0.10 MT CO_2e ac^{-1} yr^{-1} difference in the carbon accumulation between two conventionally plowed cropping systems (annual grain or grain-fallow)	The lack of historical yield information can be an impediment to insuring new crops in some cases.	
Reducing burning	Limited data found that burning accelerated carbon losses compared to unburned treatments by 0.07 MT CO2e $acre^{-1}$ yr^{-1}	The number of acres burned has been greatly reduced, thus there may be limited opportunities for further reductions.	
Increasing perennial crops in rotation with annual crops	**Annual cropping area**: Experimental data suggest gains of 1.53 ± 0.61 MT CO2e ac^{-1} yr^{-1} (Brown and Huggins, 2012)	Profitability of some perennial crops are a barrier. Different equipment needed or infrastructure and markets for current perennial crops (e.g. alfalfa) may present a barrier. Landscape level impacts need to be better understood. Perennial crop development to replace existing annual crops (e.g. perennial grain crops) remains a long-term aspirational research effort.	
Cover crops	No systematic quantitative estimates yet possible.	Focus is on cover cropping that integrates with existing food production systems, rather than cover crops that replace food production. To date, cover cropping has more successfully been integrated into irrigated cropping systems (e.g. as green manures prior to potatoes). Viable cover cropping systems have not yet been developed for dryland systems, though this is an area of ongoing work (Pan et al. 2017b)	

		The profitablility of incorporating cover crops can be a barrier, especially where cover crops use soil water that would otherwise be available to cash crops.	

Irrigated cropping systems

While similar management strategies can be used in irrigated cropping systems to build soil organic carbon, the context in which those strategies are implemented is quite different – and there have been many fewer studies in the Pacific Northwest to evaluate the potential for changes in agricultural management to lead to long-term changes in soil organic carbon. Therefore, we present the limited evidence in a single section here.

As previously mentioned, existing evidence suggests that semi-arid irrigated crop systems in the Pacific Northwest tend to have greater soil organic carbon than nearby native ecosystems (Entry et al. 2002; Cochran et al. 2007). Entry et al. (2002) estimated soil organic carbon changes using soil sampling of sites maintained in native vegetation and a number of differently-managed irrigated systems in the Snake River Plain of southern Idaho. In that location, more carbon was stored in the soils of all irrigated cropping systems than in the native sagebrush site (Figure 11). However, note that accounting for carbon losses resulting from fertilizer production, farm operations, and carbon dioxide lost via dissolved carbonate in irrigation water over a 30-year period on the agricultural sites (green portion of the bars in Figure 11) led to a greater overall net carbon gain from native sagebrush compared to the cropped system with moldboard plow.

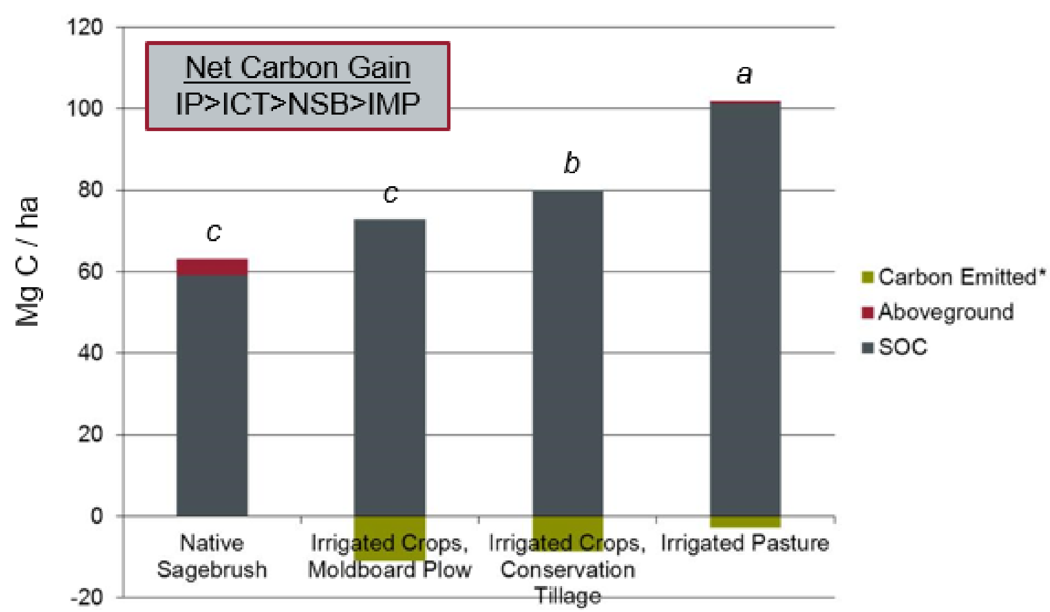

Figure 11. Organic carbon in soils and aboveground biomass in native sagebrush steppe and sites under three contrasting types of agricultural management in southern Idaho (management consistent for 8-30 years, three sites per treatment, five soil cores per site), and estimated carbon emitted during fertilizer production, farm operations, and dissolved in irrigation water over a 30-year period. Data from Entry et al. (2002).

Entry et al. estimated a net gain of 9.5 Mg C ha^{-1} (14.1 MT CO_2e $acre^{-1}$ yr^{-1}) over 30 years for conversion of irrigated moldboard plow systems to irrigated conservation tillage, and a net gain of 37.1 Mg C ha^{-1} (55.05 MT CO_2e $acre^{-1}$ yr^{-1}) over 30 years from conversion of irrigated moldboard plow to irrigated pasture (Figure 11). They estimated that conversion of irrigated cropland in the Pacific Northwest to conservation tillage would result in a gain of 311 million metric tons (MMT) CO_2e (Entry et al. 2002).

Stockle et al. (2012) modeled the soil organic carbon change resulting from conversion to reduced tillage in an irrigated system in Paterson, Washington. They used a representative rotation of sweet corn-sweet corn-potato (rather than a rotation chosen for its ability to generate residue or to store carbon), and found that the magnitude of overall carbon gains in this irrigated system were close to the values for reduced tillage in Lind (dryland winter wheat-fallow) (Figure 12). Note that true no-till may be limited as an option for some irrigated crops. For example, potatoes require tillage for harvest. High residue farming systems are continuing to be developed in the Pacific Northwest.

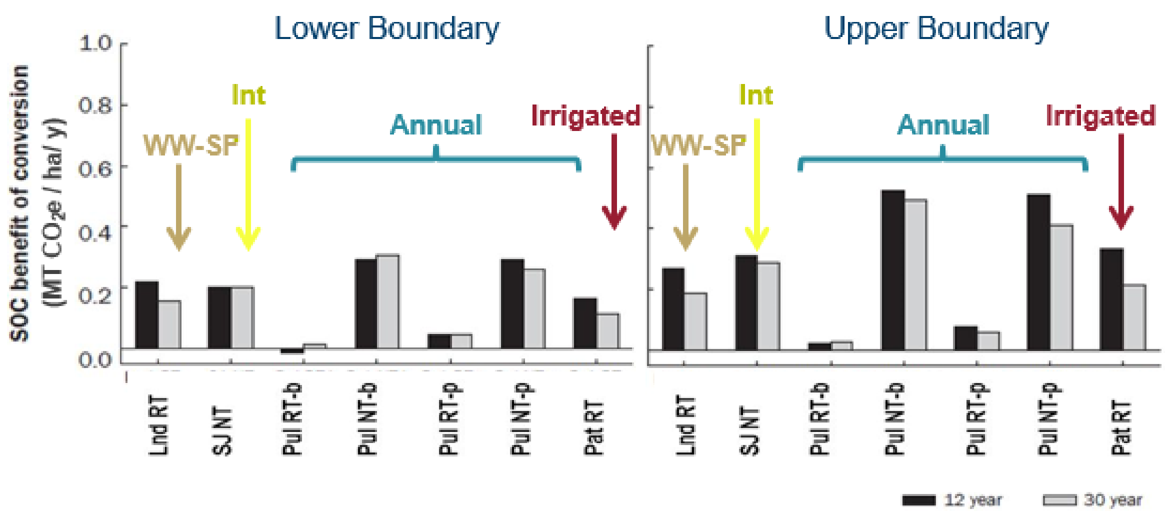

Figure 12. Modeled changes in soil organic carbon in the top 30 cm in soils 12 years (black bars) and 30 years (gray bars) after conversion to reduced tillage (RT) or no-till (NT), in different agro-ecological classes and with different rotations. The left panel shows the lower boundary of soil carbon changes, and the right panel shows the upper boundary. All bars except those marked "irrigated" were in Figure 9, above. Reproduced from Stockle et al. (2012).

Some experimental studies in irrigated systems have examined the effects of manure applied alone or in combination with biochar in silage corn systems, but few of these studies have determined long term C storage potential. Brown and colleagues (2011) found carbon storage of 6-67% of carbon applied for irrigated orchard crops (grape, cherry, and hops) in Yakima County, Washington on Warden silt loam and Esquatzel silt loam. For most crops, amendment was carried out annually for four to six years; in one case a single application was made and sampled five years later. A higher C storage rate of 89% was estimated for pear in Chelan County, Washington on a Cashmont sandy loam where applications were made annually for fifteen years. Moore and collaborators (2016) report an increase in soil organic matter of 0.02% for each ton of organic matter applied in the form of dairy manure in a three-year study near Kimberly ID.

Conclusions

The Pacific Northwest—particularly the inland Pacific Northwest—has benefited from field experiments – sampling carbon stocks and measurement of trace gas exchange – and modeling work that provide insights into the potential for carbon sequestration in cropland soils. Summarizing existing studies and their resulting conclusions highlights several key points that can provide the basis for discussions around policies and incentives for efforts to mitigate climate change through soil carbon sequestration:

- Eliminating tillage (and to a lesser extent reducing tillage), adding carbon-rich soil amendments, and increasing productivity and residue inputs through strategies such as incorporating perennial crops into rotations, can provide real contributions to carbon sequestration in the Pacific Northwest.
    - Benefits from conversion to no-till are smaller in annual crop transition and grain fallow agroclimatic zones. Benefits may be higher (on a per acre basis) in irrigated systems – but data are limited, so should be interpreted with caution.
    - In dryland systems, shifting to a mixed annual-perennial rotation can provide soil carbon benefits that are equivalent to, or higher than, the benefit obtained from a conversion to no-till in annual systems.
    - Integrating livestock manures or other organic amendments in both dryland and irrigated cropping systems, where this is economically viable, could significantly increase soil carbon sequestration on a per acre basis.
- Within dryland systems, greater opportunities exist to build soil organic carbon in annually cropped systems, where higher annual rainfall allows more frequent crop production, and higher yields. These opportunities are mirrored, to some extent, in irrigated systems. It is difficult to maintain, let alone build, soil organic carbon in areas with lower annual precipitation, due to lower yields and the greater frequency of fallow.
- The soil organic carbon benefits of particular management practices can vary depending on the initial soil organic carbon levels, the environmental and physical constraints of a site, and the amount of time that has elapsed since the management changes were made.
    - Soils with high initial organic carbon levels will accumulate further carbon at lower rates.
    - Precipitation is a key environmental condition, with greater carbon sequestration potential at higher rainfall sites.
    - It is likely that most increases in soil organic carbon due to management changes occur in the first decade or so after the change is implemented.

Interactions occur between management changes and the environment, so that the potential to sequester carbon depends on the specific combination of crop, management practices, and climatic conditions. Any policy targeting particular changes in management practices must consider the variations in climatic conditions across the region—and variations to come as the climate changes—as well as the agro-ecological classes and production systems within which such practices may or may not be implemented. Policies targeting management practices can directly impact individual farmers, who on a day-to-day basis consider and evaluate opportunities, risks and trade-offs posed by a variety of options. Farmers are subject to financial, operational and market conditions and constraints. Policy discussions should therefore engage with producers or producer groups to consider these same conditions and constraints, as they may impact the feasibility of adoption of particular practices in particular locations or under particular market conditions.

In addition to climate benefits, a variety of other benefits can result from management changes that increase soil carbon in the inland Pacific Northwest, including increased water retention and infiltration, decreased wind erosion, increased nutrient availability, and overall improvement in soil health and plant productivity. For farmer considering adopting these management strategies, it is likely that these co-benefits may be more important considerations than climate mitigation, especially in the absence of financial incentives to sequester carbon.

It is clear from the emphasis this article gives to the inland Pacific Northwest, as well as to certain management practices, that there are gaps in the available evidence supporting different carbon sequestration strategies in different areas. Similarly, the range of values—and researchers' emphasis in evaluating a range of values—highlights the variability in existing data (for more discussion of this, see Brown and Huggins, 2012). Interactions between factors are also hard to comprehensively capture and understand. These gaps, variability, and interactions all support the need to establish credible estimates of carbon fluxes for Northwest agricultural systems so that further innovation in and adoption of greenhouse gas reduction strategies can occur. This has been highlighted as a top priority for research and extension in the region (Yorgey et al., 2017d).

Credible estimates must also be accompanied by monitoring to determine whether cropland soils are achieving carbon sequestration goals. Such monitoring requires methods that are sensitive to short-term changes in soil organic carbon (e.g. Awale et al., 2017). It also requires an understanding of the impact the management practices have on other greenhouse gas emissions, from other parts of the life-cycle of inputs.

Thoughtful consideration of the environmental and production contexts surrounding Pacific Northwest crop production, combined with targeted research to facilitate the adoption of the most effective carbon sequestration practices, could lead to the development of policies that can realize the contributions that croplands in the Pacific Northwest can make to climate change mitigation efforts in the region.

Sidebar: Units in this publication

In the literature on agricultural soils, changes in carbon over time are often expressed as Mg carbon (C) ha^{-1} yr^{-1}. However, from a climate policy perspective, mitigation targets and strategies are usually discussed in terms of metric tons of carbon dioxide equivalents (MT CO_2e) – and strategies deployed across the landscape, are often discussed on a per acre basis (despite the fact that this mixes English and metric units). We have therefore chosen in this publication to express values in terms of Mg carbon (C) ha^{-1} yr^{-1}, with MT CO_2e $acre^{-1}$ yr^{-1} provided as well.

(1 Mg C/hectare year) x (44 units CO_2/12 units C) x (1 MT/1 Mg) x (1 hectare/2.47105 acres) = 1.48 MT CO_2e/acre year

Sidebar: Review of Regional Experimental Datasets

Brown and Huggins (2012) surveyed existing datasets on soil organic carbon changes in the inland Pacific Northwest. The authors found 131 datasets in the peer-reviewed and non-peer-reviewed literature (e.g. bulletins and project reports), with data concentrated in annual cropping and annual crop-fallow transition classes and, to a lesser extent, the grain-fallow class (Table 5, reproduced from Brown and Huggins, 2012). These authors used the agro-climatic zones that were defined by Douglas et al. (1992), where ACZ1 is mountain/forest (wet-cold), ACZ2 is annual cropping (wet-cold), ACZ3 is annual cropping (fallow-transition), ACZ4 is annual crop (dry), ACZ5 is grain-fallow, and ACZ6 is dryland (in the irrigated/very dry region). Zones 2, 3, and 5, which contain most of the data, correlate with the annual cropping, annual crop-fallow transition, and grain-fallow classes used in this document.

Table 5. Summary of collected soil organic content literature by management practice (native conversion [NC], no-tillage management [NT], reduced tillage management [RT], mixed perennial-annual system [Mixed P-A], Conservation Reserve Program planting [CRP], annual cropping, fallow cropping, residue burning, no residue burning, barnyard manure application, and green manure application) and Pacific Northwest agroclimatic zone (ACZ). Numbers of studies listed are by location rather than publication. Modified from Brown and Huggins (2012).

ACZ*	NC	NT	RT	Mixed P-A	CRP	Annual cropping	Fallow cropping	Residue burning	No residue burning	Barnyard manure	Green manure
1	1	—	—	—	—	—	—	—	—	—	—
2	8	12	3	9	2	16	9	1	1	6	14
3	4	13	4	1	1	2	4	3	3	2	2
4	—	—	—	—	—	—	—	—	—	—	—
5	3	1	—	—	1	—	—	1	2	1	—
6	1	—	—	—	—	—	—	—	—	—	—

*The agro-climatic zones (ACZ) used here are those that were defined by Douglas et al. (1992), where ACZ1 is mountain/forest (wet-cold), ACZ2 is annual cropping (wet-cold), ACZ3 is annual cropping (fallow-transition), ACZ4 is annual crop (dry), ACZ5 is grain-fallow, and ACZ6 is dryland (in the irrigated/very dry region).

The authors converted each dataset from its original units to mass per unit volume of soil per year, allowing them to (a) compare the values of different studies, (b) assess the depth-distribution of soil organic carbon change, and (c) estimate the total profile changes in soil organic carbon. Where they found adequate data with particular combinations of management changes and agro-ecological conditions, they carried out a comprehensive analysis to estimate mean soil organic carbon change, as well as the cumulative probability of that change. Their methods are described in detail in Brown and Huggins (2012). For other management practices where only limited data existed, they provided summaries of experimental results.

Sidebar: Pendleton Long-Term Experiments

One of the best resources for understanding long-term soil carbon dynamics in the dryland soils of the Pacific Northwest comes from a set of long-term experiments established at the Columbia Basin Agricultural Research Center (http://cbarc.aes.oregonstate.edu/) in Pendleton, Oregon. The longest-standing of these experiments, investigating residue management in a winter wheat-summer fallow cropping system, was established in 1931. Prior to the establishment of the trial, all plots were cultivated for about 50 years, and presumably were losing carbon over that time period. This experiment considers several methods for managing residues, including the following:

- No nitrogen added, no special management (the control)
- Fall burn, no nitrogen added (burning increases losses of carbon from the system)
- Two different rates of nitrogen addition, 45 kg ha^{-1} and 90 kg ha^{-1} (added nitrogen will result in higher yields and more residues, with the residues, and their incorporated carbon, being incorporated back into the soil during tillage operations)
- Pea vines added (at a rate of 1.1 Mg ha^{-1} yr^{-1})
- Manure added (at a rate of 11.2 Mg ha^{-1} yr^{-1})

Sidebar: CropSyst – a simulation model

CropSyst is a crop growth simulation model, developed with an emphasis on a friendly user interface (Stockle, 1996). The model is basically an analytical tool which can be used to study the effect of cropping systems management on crop productivity and the environment—for the purposes of this white paper, we focus on soils. CropSyst simulates—and quantifies—a variety of processes that determine how quickly crops grow and accumulate biomass, and move through different stages of development, what happens with water in the soil; what happens to nitrogen that is added to the cropping system, absorbed by the crop, and then recycled into the soil or released, how much residue is left after harvest, and what happens to that residue and the carbon it contains. These processes are affected by weather, soil characteristics, crop characteristics, and cropping system management options such as crop rotation, cultivar selection, irrigation, nitrogen fertilization, soil and irrigation water salinity, tillage operations, and residue management. Many of these management options are strategies that could increase the ability of agricultural soils to sequester carbon, helping mitigate climate change impacts. CropSyst is therefore a useful modeling tool to explore the carbon sequestration potential of agricultural soils in the Pacific Northwest, and has been used extensively in the region. For more details on CropSyst, please visit: http://modeling.bsyse.wsu.edu/CS_Suite/CropSyst/index.html

Version Note

As of the publication date of this document (November 2019), this document is currently undergoing peer review within the Washington State University Extension publications system.

After review, an updated version of this document will be available as a formal Extension publication.

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