CHAPTER 5

Agricultural Management and Soil Carbon Dynamics: Western U.S. Croplands

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CHAPTER OUTLINE

INTRODUCTION

Soils are the largest pool of carbon (C) in the terrestrial environment (Jobbagy and Jackson, 2000; Schlesinger, 1995). The amount of C stored in soils is twice the amount of C in the atmosphere and three times the amount of C stored in living plants (Kimble and Stewart, 1995). Therefore, a change in the size of the soil C pool could significantly alter current atmospheric CO_2 concentrations (Wang et al., 1999). Carbon stored in soils is derived from litter, root inputs, sediment deposits, and exogenous applications of manures/mulches, while losses result from microbial degradation of soil organic matter, eluviation, and erosion (Entry and Emmingham, 1998). As an ecosystem approaches maturity, C sequestration potential is controlled by climate, topography, soil type, and vegetation (Harmon et al., 1990;

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Dewar, 1991; Van Cleve et al., 1993). At equilibrium, the rate and amount of C added to the soil from plant residues and roots, organic amendments as well as erosion deposits, are equal to the rate and amount of C lost through organic matter decomposition and soil erosion processes (Henderson, 1995; Paustian et al., 1997).

Within limits, C in soil increases with increasing soil water and decreases with temperature (Wang et al., 1999). The effect of soil water is much greater than the effect of soil temperature (Liski et al., 1999). Increasing water within a temperature zone can increase plant production and, thus, C input to soils via increased plant litter and root production (Liski et al., 1999), but increasing water also can reduce soil C through enhanced decomposition.

Land-use changes can impact the amount of C stored in the soil by altering C inputs and losses. Conversion of native vegetation to agricultural cropping has resulted in both substantial C transfer to the atmosphere and loss of soil C (Lal et al., 1999; Wang et al., 1999).

Agricultural practices that can partially restore depleted soil organic carbon (SOC) include: (1) adoption of conservation tillage including no-tillage; (2) intensification of cropping by eliminating fallow, increasing cover crops and including more perennial vegetation (Sperow et al., 2003); and (3) improving biomass production through the use of soil amendments (manures), fertilizers and high yielding crop varieties (Lal et al., 1998; Follett, 2001).

CLIMATE, CROP, AND CROPPING PRACTICE CHARACTERISTICS

Climate

The climate of the western U.S. is characterized mainly as semiarid, with the southwest portion of the region classified as arid; each of these areas exhibits a diverse range in annual precipitation and temperature that significantly influences crop production and C storage. The following information was obtained from the Western Regional Climate Center (http://www.wrcc.dri.edu/). In general, precipitation decreases from east to west across the region. The average midwinter temperature ranges in the extreme north (ND) are −8 to −13°C, while summer temperatures range from 18 to 23°C. In the southern prairie (OK, TX) winter temperatures average 10 to 13°C, with little variation in summer temperatures from 27 to 29°C. Precipitation along the western border of the prairie region ranges from 46 cm yr^{-1} in the north to 64 cm yr^{-1} in the south. Most precipitation occurs in the early summer. Precipitation averages about 50 cm yr^{-1} in the southern plains and 25 cm yr^{-1} in the northern plains, with extreme year-to-year variations.

The intermountain arid region of the west shows considerable climatic variation between its northern and southern boundaries. In NM, AZ, and southeastern CA the greatest precipitation occurs in the summer months, with an average annual precipitation from 8 cm in the valleys to 76 cm yr^{-1} in the mountains of AZ, NM, NV, and UT. North of the UT–AZ line, the summer months usually are very dry; maximum precipitation occurs in the winter and early spring. In the desert valleys west of Great Salt Lake, mean annual precipitation averages 10 cm.

The Pacific Northwest states of WA, OR, and ID are bounded on the east by the Rocky Mountains and on the west by the Pacific Ocean. The climate is influenced by the region's mountain ranges. In WA and OR, the north to south Cascade Range delineates a wetter coastal climate from a drier inland continental climate. Precipitation across the region ranges from 15 to 500 cm yr^{-1} with most occurring between October and April. The valleys west of the Cascade Range receive $\geq$75 cm yr^{-1} of precipitation, whereas east of the Cascades precipitation is $\leq$50 cm yr^{-1}. Temperature in the coastal zone is mild, but east of the Cascades and in southern Idaho, temperature is warmer in the summer and has larger annual ranges. Many areas of the Pacific Northwest could be classified as Mediterranean if it were not for the cold winter temperatures. Although the northern plateaus are generally arid, some of the mountainous areas of central Washington and Idaho receive ≥150 cm of precipitation annually. Throughout the western U.S., the irregular occurrence of precipitation is the principal factor for the development of irrigation to support agriculture.

Crop Production

Agriculture in the western U.S. is as diverse as the climate and topography. The type of crops produced varies depending on rainfall, irrigation, soil, elevation, and temperature extremes. The total harvested cropland in the western U.S. was estimated at 57 Mha in 2007 (NASS, 2007). In the most arid regions (AZ, NM, NV, TX, and UT) without irrigation, production agriculture generally supports only livestock grazing. Texas is a major cattle and sheep raising area, as well as the nation's largest producer of cotton (*Gossypium hirsutum* L.). Climate is a major factor influencing the distribution of cropping systems within the dryland (i.e. rain-fed) cropping regions of the PNW (Papendick, 1996; Paustian et al., 1997). Areas that receive <375 mm of water annually are typically managed using a 2-year winter wheat (*Triticum aestivium* L.)—summer fallow rotation (Rasmussen et al., 1998; McCool and Roe, 2005). Areas with rainfall of 375 to 450 mm are typically managed using a 2- to 5-year rotation that often includes winter wheat–spring barley (*Hordeum vulgare* L.) or spring wheat–summer fallow (Papendick, 1996; Rasmussen et al., 1998; McCool and Roe, 2005). Fallow is discontinued and annual cropping is practiced in areas that receive >450 mm of precipitation. Approximately 80–90% of cropland in the Northern and Central Plains region is farmed under dryland conditions and <20% has irrigation. The traditional dryland rotation is winter wheat–fallow under conventional tillage (CT). In higher rainfall areas, crops grown in rotation with winter wheat are corn (*Zea maize* L.), spring wheat, proso millet (*Panicum miliaceum* L.), sorghum (*Sorghum bicolor* L.), sunflower (*Helianthus annuus* L.), barley, oat (*Avena sativa* L.), and pea (*Pisum sativum* L.). In eastern Montana and North Dakota the dominant dryland cropping system is spring wheat–fallow under conventional tillage. This rotation is being slowly replaced by continuous cropping of spring wheat, or in 1- to 4-year crop rotations with winter wheat, barley, pea, lentil (*Lens culinaris* L.), canola (*Brassica napus* L.), and yellow mustard (*Brassica juncea* L.). Annual forages, such as barley hay, Austrian winter pea hay, foxtail millet hay (*Setaria italic* L.), and perennial forages, such as alfalfa (*Medicago sativa* L.), are also included in rotation with cereals.

Irrigation dominates (>90%) crop production in AZ, CA, and NV, with ~64% in ID, NM, UT, and WY, ~36% in CO, NE, OR, and WA, and <25% in KS, MT, ND, OK, SD, and TX (NASS, 2007). CA has the largest area under irrigated production with >3.5 M ha (NASS, 2007). Irrigation in the areas of the PNW, CA, and the southwest (AZ, NM, NV, UT) allows for the production of perennial fruits, nuts, and annual vegetables as well as grain, alfalfa, and grass hay. California and Arizona are also major producers of citrus crops. Irrigated cropping systems in the Great Plains region incorporate sugar beet (*Beta vulgaris* L.), malt barley, potato (*Solanum tuberosum* L.), corn, soybean (*Glycine max* L.), and dry bean (*Phaseolus vulgaris* L.) in rotation.

Tillage

Tillage in the western U.S. is highly variable due to the diversity of crops. Grain crops are significant in all states and are mostly grown under conventional tillage by plowing. Area under no-tilled grains is increasing in the Pacific Northwest, although it is <10%. Area under full-width inversion tillage is similar to that under winter grain crop production but area under reduced tillage (e.g. chisel, sweeps, field cultivators) tends to follow that under spring crops (grain or legume) in the higher rainfall areas where annual dryland cropping is practiced. Overall, conservation tillage regime has been widely used for most crop rotations (Kok et al., 2009). Conservation tillage practices leave at least 30% of the soil surface covered by crop residues and reduce soil disturbance and SOC decomposition (Jian et al., 2005). In the drier regions, grain production with wheat–fallow systems is common and tillage includes several rod-weeding operations and disking in the fallow year. In the production of hay and grass seed, tillage is a minor component. However, most of the other crops produced in the Pacific Northwest use some form of tillage to prepare seed beds and control weeds.

MANAGEMENT EFFECTS ON SOIL C STORAGE: DRYLAND SYSTEMS

Conversion of Native Ecosystems to Agriculture

Soil disturbances resulting from land-use changes have been shown to modify the turnover of C and the formation of soil organic matter (SOM) (Huggins et al., 1998Collins; et al., 2000; Dinesh et al., 2003). Conversion of native ecosystems to agricultural production usually results in a loss of SOM. Such losses are well documented for the Great Plains and Corn Belt (Paustian et al., 1997), with few examples in the Pacific Northwest (Rasmussen et al., 1980; Rasmussen and Collins, 1991).

Conversion of native lands to agricultural production in the Pacific Northwest resulted in SOC loss of 35% during the first 40 years of cultivation (Sievers and Holtz, 1926). Purakayastha et al. (2008) reported a 56% reduction in SOC during 100 years of conventional tillage. The rate of profile (0 to 125 cm depth) SOC decline following conversion of native vegetation to cropland ranged from 0.53 to 1.07 Mg C ha^{-1} yr^{-1} in the higher and 0.33 to 0.76 Mg C ha^{-1} yr^{-1} in the intermediate precipitation zones of the PNW (Brown and Huggins, 2011). In the low precipitation region of the Pacific Northwest, profile SOC declined by 0.16 to 1.15 Mg C ha^{-1} yr^{-1}, but these data represent a more recent history of cultivation (3 to 9 years) compared to the high and intermediate precipitation regions (35 to 100 years). The 0.24 Mg C ha^{-1} yr^{-1} SOC gain observed by Rodman (1988) in a footslope position profile is indicative of the role erosion processes play in SOC content and distribution. The large range in SOC changes reflect not only the differences in biomass production and agricultural management, but also various soil depths sampled, time under cultivation, erosion history, and climatic conditions that exist within the Pacific Northwest. This emphasizes the need for a more standardized protocol as most of the data ranges reported here were not sampled specifically for evaluating SOC stocks.

Conversion of Conventional Tillage (CT) to No-Tillage (NT) or Reduced Tillage (RT)

Increasing concern about the sustainability of crop production systems and environmental quality has emphasized the need to develop and implement management strategies that maintain and protect soil, water, and air resources. Tillage affects the amount of SOM buildup in two fundamental ways: (1) through the physical disturbance and mixing of soil and the exposure of soil aggregates to disruptive forces and (2) through controlling the incorporation and distribution of plant residues in the soil profile. Adoption of NT following CT generally results in SOC increases, but in some instances little or no increases are observed. In the Pacific Northwest, increases in profile SOC (20 cm) ranged from 0.03 to 1.95 Mg C ha^{-1} yr^{-1} and averages for different agroclimatic zones ranged from 0.21 to 0.71 Mg C ha^{-1} yr^{-1} (Brown and Huggins, 2011). These values are within the range of 0.3 to 0.8 Mg C ha^{-1} yr^{-1} reported previously (Follett, 2001; West and Post, 2002; Liebig et al., 2005) from comparison of NT and CT cropland. Purakayastha et al. (2008) found that SOC (0 to 20 cm) from an unreplicated field survey was significantly higher under NT compared to CT, but no significant differences between 4 and 28 years of NT were observed. However, particulate organic C (POC) in the surface 5 cm was significantly higher in the 28-year NT site (8.1 Mg C ha^{-1}) compared to the 4-year NT site (6.3 Mg C ha^{-1}). In the surface 15 to 20 cm, changes in SOC following change from CT to RT ranged from gains of 0.02 to 0.07 Mg C ha^{-1} yr^{-1} to losses of 0.08 Mg C ha^{-1} yr^{-1}. Increased SOC storage in the fine organic matter (FOM) fraction of RT soils (i.e. sweep) compared to moldboard plowed soils ranged from 0.16 to

0.18 Mg C ha^{-1} yr^{-1} at N fertilizer rates of 45 and 180 kg N ha^{-1}, respectively, in a 60 cm profile under long-term (44 years) wheat–fallow system (Gollany et al., 2005). However, tillage had a larger effect on FOM than N fertilization (Gollany et al., 2005). On a profile SOC basis, the 0.05 Mg C ha^{-1} yr^{-1} rate of SOC increase under RT estimated by Liebig et al. (2005) for the northwestern U.S. appears reasonable for the dryland Pacific Northwest. It is also important to note that in a few instances NT was reported to have less (Fuentes et al., 2004; Kennedy and Schillinger, 2006) and RT more SOC than the CT counterpart (Rasmussen and Rhode, 1988; Machado et al., 2006). This can be explained by residue burial to a greater depth due to tillage and shows that limited depth of soil sampling could result in over- or under-estimation of SOC sequestration potential from changes in tillage management. Machado et al. (2006) in eastern Oregon found that tillage influenced the amount and distribution of SOC in the soil profile. Compared to historical records (1931), CT reduced SOC in all layers except the 10–20 cm depth increment, where plowing deposited the majority of crop residue. Under NT, SOC concentration closely mirrored a grass pasture system, with the highest SOC in the 0–10 cm layer and SOC decreasing with soil depth. Kennedy and Schillinger (2006) reported depth and landscape differences in SOC response to NT compared to traditional tillage (TT). The SOC content of NT soils was similar to TT at summit positions, but greater than TT in the surface 0 to 5 cm at toe and side slope landscape positions. This trend was not always observed in the 5 to 10 cm depth increment (Kennedy and Schillinger, 2006).

In central Montana, Sainju et al. (2006a) reported that NT increased SOC at the 0–5 cm depth by 0.2 Mg ha^{-1} yr^{-1} compared with CT after 6 years. They found that SOC at 0 to 20 cm increased by 0.6 Mg ha^{-1} in NT, but was reduced by 0.8 Mg ha^{-1} in CT from 1998 to 2003. In eastern Montana, Aase and Pikul (1995) observed that SOC declined at a rate of 0.4 Mg ha^{-1} yr^{-1} in CT spring wheat–fallow compared with a decline of 0.1 Mg ha^{-1} yr^{-1} in NT continuous spring wheat from 1983 to 1993. Sainju et al. (2007b) found that SOC at 0–20 cm was 8.4 Mg ha^{-1} greater in NT continuous spring wheat than in CT spring wheat–fallow after 21 years. They reported that SOC declined more in CT spring wheat–fallow than in NT continuous spring wheat from 1983 to 2004. When total (organic + inorganic) C was considered, C storage increased by 0.19 Mg ha^{-1} yr^{-1} in NT continuous spring wheat, but decreased by 0.27 Mg ha^{-1} yr^{-1} in CT spring wheat–fallow during the same period. They observed greater increases in POC, potential C mineralization, and microbial biomass C relative to SOC in NT continuous spring wheat than in CT spring wheat–fallow. However, SOC was not altered by tillage and cropping sequence after 3 years of study (Sainju et al., 2008, 2010).

In the central Great Plains (Akron, CO), on Weld silt loam soil (Aridic Paleustolls), NT increased SOC at the 0–5 cm depth by 0.2 Mg ha^{-1} compared with CT after 15 years of winter wheat–fallow rotation (Mikha et al., 2010). In NT, fallow elimination significantly increased SOC by 0.3 Mg ha^{-1} at 0–5 cm and by 0.24 Mg ha^{-1} at 0–20 cm, with wheat–corn–millet rotation compared with wheat–fallow rotation. The retention of SOC declined as the fallow frequency increased in rotation. Fabrizzi et al. (2007) evaluated four long-term sites (Table 5.1) in Kansas (Tribune, Hays, Manhattan, and Parsons) managed under variable fertilizer rates and tillage regimes including CT, RT and NT. They found the greatest increase in SOC at 0 to 5 cm for N-fertilized NT treatment. At 0 to 15 and 0 to 30 cm, no significant differences in SOC were

TABLE 5.1 Description of the Experimental Sites Reported by Fabrizzi et al. (2007)

Site	Location	Soil series/type	Year after initiation
Tribune	Southwest	Richfield silt loam (Aridic Argiustolls)	16
Hays	North central	Harney silt loam soil (Typic Argiustoll)	37
Manhattan	Northeast	Muir silt loam (Cumulic Haplustoll)	29
Parsons	Southeast	Parsons silt loam (Mollic Albaqualfs)	20

observed between tillage systems, except at the Manhattan site, although in most cases NT tended to result in greater SOC content. The SOC levels in CT and RT at Hays indicated a loss of C from the system and a gain in NT (0.02 Mg C ha^{-1} yr^{-1}). Previous land use was important when evaluating the effect of management on soil C. For example at Tribune, all tillage systems had a negative effect on C sequestration, but C loss was lower in NT (4%) than in either RT (7%) or CT (9%). This experiment was initiated on a site broken from native prairie sod, and had higher initial soil C concentrations than the other sites, which had been under cultivation before the establishment of the tillage systems. At the Parsons site, 20 years of N addition at 140 kg N ha^{-1} increased C sequestration by 0.11 Mg C ha^{-1} yr^{-1} relative to 0 N rate (Fabrizzi et al., 2007).

Previous research reported that NT practices improve soil aggregation, C sequestration, and aggregate stability (Mikha and Rice, 2004; McVay et al., 2006; Zhang et al., 2007) which results in increased water infiltration and resistance to wind and water erosion (Zhang et al., 2007). Macroaggregate stability (>250 μm diam.) is particularly responsive to management practices (Jiao et al., 2006; Zibilske and Bradford, 2007). The loss of macroaggregate-occluded organic matter is a primary source of carbon lost as a result of changes in management practices (Six et al., 2002; Mikha and Rice 2004; Jiao et al., 2006; Zibilske and Bradford, 2007). Continuous cropping with reduced fallow frequency and NT exhibits a positive effect on macroaggregate formation and stabilization as well as particulate organic matter (POM) and SOC (Mikha et al., 2010).

Fallow and Crop Rotations

Historical SOC losses from fallow cropping every other year have ranged from 0.09 to 0.65 Mg C ha^{-1} yr^{-1} within the surface 30 cm (Horner et al., 1960) but more recent estimates exhibit a narrower range of SOC loss (0.09 to 0.12 Mg C ha^{-1} yr^{-1}) (Rasmussen and Albrecht, 1998). Schillinger et al. (2007) observed that conversion from CT wheat–fallow to annual NT continuous wheat increased SOC to near native values in the top 5 cm but was not different in the surface 10 cm layer among NT cropping systems with various annual crop rotations in wheat–fallow regions of Washington state. Horner et al. (1960) reported surface SOC changes under annual wheat cropping with tillage to range from a loss of 0.25 Mg C ha^{-1} yr^{-1} to a gain of 0.04 Mg C ha^{-1} yr^{-1} and wheat in rotation with spring pea or oats to exhibit a relatively larger potential for SOC losses (0.07 to 0.47 Mg C ha^{-1} yr^{-1}). Greater SOC loss has been observed in unfertilized compared to N fertilized conditions but the rate and direction of change were site specific depending also on duration of residue and crop rotation management (Horner et al., 1960; Gollany et al., 2006). Gollany et al. (2006) and Wuest et al. (2005) reported that long-term (75-year) application of organic amendments (manure management and pea vine residue) on a Walla Walla silt loam (Typic Haploxeroll) was more effective in increasing the surface 60 cm SOC stocks than N fertilization or residue management for wheat–fallow plots under moldboard plowing. This was mostly due to the significant increases in FOM in the surface Ap horizon (0 to 26 cm).

Recent profile SOC data addressing the impact of current Pacific Northwest crop rotations, alternative cropping system practices and fertilization are lacking for many of the dryland cropping systems. In central Montana, Sainju et al. (2006b) found that cropping sequence did not influence SOC but POC at 0 to 5 cm was 0.6 to 0.8 Mg ha^{-1} greater in spring wheat–spring wheat–fallow and spring wheat–pea–fallow than in spring wheat–fallow after 6 years. Similarly, microbial biomass C (MBC) at 0 to 5 cm was 26% greater in spring wheat–spring wheat–fallow than in spring wheat–fallow (Sainju et al., 2007a). In eastern Montana, Sainju et al. (2011) reported that SOC at 0–5 cm was greater in continuous spring wheat than in spring wheat–barley hay–pea in both NT and CT after 4 years, probably a result of lower biomass of pea than spring wheat and removal of aboveground biomass of barley for hay.

In the central Great Plains, Akron, CO, on Weld silt loam soil (Aridic Paleustolls), Mikha et al. (2010) found that continuous cropping with reduced fallow significantly increased SOC compared with rotations that included fallow every second and third year. Including a legume in rotation (wheat–corn–millet–pea) did not increase SOC relative to summer fallow rotations in NT (Mikha et al., 2010). The low SOC accumulation with pea in rotation could be due to fast decomposition rate of pea residue, relative to other crops in rotation. After 15 years, MBC at 0 to 5 cm was 42% greater in continuous cropping (wheat–corn–millet) than in wheat–corn–fallow and 72% greater than wheat–fallow (Acosta-Martínez et al., 2007). Continuous cropping (wheat–corn–millet) significantly increased soil particulate organic matter (POM) at 0 to 5 cm depth by 29% compared to wheat–corn–fallow and by 21% compared with wheat–corn–millet–fallow. This indicates that the differences in soil POM among cropping sequences increased as fallow frequency increased in rotations (Mikha et al., 2010). This observation is consistent with the regional study, eight long-term sites throughout the Great Plains and the western Corn Belt (Table 5.2), reported by Mikha et al. (2006), who observed an increase in soil POM level with the reduction in tillage intensity and fallow frequency.

At the study site in Mandan, ND, on a Temvik-Wilton silt loam soil (Typic and Pachic Haplustolls), Halvorson et al. (2002a) reported that 12 years of fallow occurrence in cropping systems, even with NT management, could result in losses of SOC. Similarly, Mikha et al. (2006) also reported that decreased fallow frequency causes an increase in SOC, within the 0 to 7.5 cm depth. Overall, SOC loss is likely to occur during the fallow period (Cihacek and Ulmer, 1995) because fallow period signifies a phase of continued microbial activity and residue decomposition with no crop residue input. In addition, soil may be susceptible to SOC loss by wind erosion during the fallow period (Haas et al., 1975). The relative difference in SOC between the continuous cropping (wheat–corn–millet) and wheat–fallow cropping systems, Weld silt loam soil at Akron, CO, increased during the years between 1999 and 2005. Bowman et al. (1999) reported approximately 13% greater SOC, in the 0–15 cm depth, was associated with continuous rotation (wheat–corn–millet) compared with the wheat–fallow cropping systems and no differences in SOC among different cropping intensities below the 15 cm soil depth. However, Benjamin et al. (2008) reported a 30% increase of SOC with continuous cropping system (wheat–corn–millet) compared with the wheat–fallow rotation averaged over the 37 cm soil depth. They also found differences in SOC between wheat–corn–millet and wheat–fallow rotation at the 20 to 28 cm depth. The changes in SOC over time indicate that a fallow elimination continues to add SOC to the soil system and that the SOC is translocated deeper in the soil (Benjamin et al., 2008).

Compared to annual cropping systems, the inclusion of a perennial crop into an otherwise annual crop rotation (mixed perennial–annual rotation) was estimated to increase profile SOC stocks (0 to 92 cm layer) by 0.66 to 1.97 Mg C ha^{-1} yr^{-1} in higher precipitation regions of

TABLE 5.2 Site location and soil series from the regional study, throughout the Great Plains and the western Corn Belt, reported by Mikha et al. (2006)

Site location	Soil series
Akron, CO	Weld silt loam
Brookings, SD	Barnes sandy clay loam
Bushland, TX	Pullman silty clay loam
Fargo, ND	Fargo silty clay
Mandan, ND	Wilton silt loam
Mead, NE	Sharpsburg silty clay loam
Sidney, MT	Willams loam
Swift Current, SK	Swinton silt loam

the dryland Pacific Northwest (Brown and Huggins, 2011). Liebig et al. (2005) previously estimated a 0.94 ± 0.86 Mg C ha^{-1} yr^{-1} regional rate of SOC gain for conversion of cropland or reclaimed mining land to grass. Sainju and Lenssen (2011) reported that perennial legume forages, such as alfalfa, increased SOC by 2.2 to 5.2 Mg ha^{-1} and POC by 1.1 to 1.2 Mg ha^{-1} at 0–15 cm compared with durum–annual forage sequences after 4 years in eastern Montana. The corresponding increase in potential C mineralization at 0 to 120 cm was 46 to 84 kg ha^{-1} and MBC at 0 to 15 cm was 6 to 80 kg ha^{-1}. They attributed these to be a result of increased belowground biomass (root + rhizodeposit) C and a relatively undisturbed soil condition in alfalfa compared with durum–annual forages.

Potter (2006) compared soils collected from several perennial and annual cropping systems in central Texas on Houston Black Series (Udic Haplusterts) to archived (1949) soil samples. Tilled soils had lost nearly 60% of SOC in the surface 30 cm due to long-term agricultural practices compared to the adjacent native grasslands. Restoring the sites to a perennial coastal Bermuda grass (*Cynondon dactylon* (L.) Pers.) sequestered 13 to 20 Mg ha^{-1} after 39 and 55 years of cropping, respectively, averaging 0.36 Mg C ha^{-1} yr^{-1}. Two other sites evaluated had been in continuous row cropping of small grains, sorghum, corn and cotton since 1949. After 55 years of continuous cropping C sequestered in the surface 30 cm averaged 0.14 Mg C ha^{-1} yr^{-1}, roughly 40% of the C sequestered in the perennial Bermuda grass. In a related study, Potter and Derner (2006) showed that SOC in agricultural soils had a 40 to 60% reduction compared to native grassland soils. The native grassland soils contained 13.7 to 18.4 Mg C ha^{-1} in the surface 5 cm, while the agricultural soils ranged from 7.3 to 8.6 Mg C ha^{-1}. Particulate organic carbon comprised 15 to 18% of SOC in the 10 to 40 cm depth increment in the native grassland soils and 7 to 16% of SOC in the agricultural soils.

Conservation Reserve Program

The Conservation Reserve Program (CRP) is a voluntary USDA program of the U.S. Farm Bill that encourages farmers to convert highly erodible cropland or other environmentally sensitive areas to conservation vegetation, such as introduced or native grasses, trees, filter strips, or riparian buffers. Follett et al. (2001) reported C sequestration under CRP in areas of summer rainfall to range between 0.6 and 1.0 Mg C ha^{-1} yr^{-1}. In NE, KS, and TX, Gebhart et al. (1994) found that SOC accumulated at a rate of 1.1 Mg C ha^{-1} yr^{-1} in 2 m profile. Purakayastha et al. (2008) noted SOC gains of 0.35 and 0.03 Mg C ha^{-1} yr^{-1} at 0–5 cm and 5–10 cm, respectively, after 11 years in CRP management. In addition, Sanchez de-Leon (2007) observed that after 23 years in CRP, the SOC content of CRP ground remained 2.4, 4.1, and 3.4 Mg C ha^{-1} below those of native Palouse prairie in the 0 to 10, 10 to 20, and 20 to 30 cm depth increments, respectively. Lee et al. (2007) found that the amount of C sequestered was dependent upon the type of N fertilizer applied. They reported a C sequestration potential of 2.4 Mg C ha^{-1} yr^{-1} with NH_4NO_3 fertilizers and 4.0 Mg C ha^{-1} yr^{-1} for manure N in a 0.9 m profile under CRP lands in South Dakota.

Machado et al. (2006) in eastern Oregon found that after 73 years, grassland pasture in NT with large annual additions of grass residue had higher (82 Mg ha^{-1}) SOC concentrations than CT winter wheat fallow (50 Mg ha^{-1}) or for fertilized CT continuous crop winter wheat systems (75 Mg ha^{-1}). Bronson et al. (2004) in soils of the Southern High Plains found that SOC and total N content in CRP land was greater than that of nearby cropland soils in the 0 to 5 cm layer. They reported SOC in CRP soils averaged 5.7 Mg ha^{-1} compared to 3.4 Mg ha^{-1} for irrigated cotton soil. Below 5 cm, they found no differences in SOC storage.

In Wyoming, Reeder et al. (1998) estimated a 60–75% decrease in SOC of the 0 to 20 cm depth after 6 years following cultivation of a native grass land due to tillage. They also observed a significant increase in SOC (~0.4 Mg C ha^{-1} yr^{-1}) after 4 years of reestablishing a fertilized grassland (34 kg N ha^{-1}) after 60 years cropping. In central Kansas, Huang et al. (2002) reported an increase in SOC (~0.14 Mg C ha^{-1} yr^{-1}) at the 0–5 cm depth after 10 years of grass on CRP lands compared with adjacent cropland. However, as the CRP lands were converted to CT croplands, the surface 0–5 cm depth lost about 1.5 Mg C ha^{-1} where the 5–10 cm depth gained about 1.1 Mg C ha^{-1}. In eastern Washington, after 4 to 7 years of CRP establishment, Staben et al. (1997) observed no differences in SOC between CRP land and a wheat–fallow rotation at the 0 to 7.5 cm depth. However, the potential mineralizable C pool was 55% greater with CRP than wheat–fallow soils, which was probably due to buildup of higher quality SOM that increased the potential for microbial activity.

MANAGEMENT EFFECTS ON SOIL C STORAGE: IRRIGATED SYSTEMS

In arid and semiarid environments of the western U.S., crop production requires irrigation to increase plant production to the point where cropping becomes economically viable (Lal et al., 1998). Irrigated crops produce twice as much plant biomass as rain-fed crops (Bucks et al., 1990; Howell, 2000). Irrigation increases C input to soils by producing higher crop yields that contribute to stable SOC through greater inputs from plant residues and root systems and increased aggregate formation (De Gryze et al., 2005; Kong et al., 2005; Gillabel et al., 2007). Estimates of SOC accumulation resulting from irrigation in the western U.S. range between 0.25 and 0.52 Mg C ha^{-1} yr^{-1} (Eve et al., 2002). The increase in SOC in irrigated agricultural soils over native soils is contrary to responses in rain-fed agricultural soils (Paustian et al., 1997; Entry et al., 2002; Cochran et al., 2007). Semiarid shrub–steppe ecosystems differ from other native systems (forests, permanent grasslands or native prairies) because of the relatively small amount of annual precipitation and historically lower levels of C inputs and storage in soil (Entry et al., 2002). The effect of irrigation on soil C and N dynamics has been quantified for only a few agricultural systems (Leuking and Schepers, 1985; Entry et al., 2002; Gillabel et al., 2007).

Intensively managed irrigated crop or pasture lands have the potential for C gain through the use of improved grazing regimes, improved fertilization practices and irrigation management (Follett, 2001; Bruce et al., 1999). Entry et al. (2002) found that irrigated soils in southern Idaho under different agricultural practices (pasture and conservation tillage) exhibited significant increases in SOC over the native sagebrush ecosystem. They estimated a gain of 9.5 Mg C ha^{-1} over 30 years if the land currently managed with conventional tillage adopted conservation approaches. Carbon sequestration is expected to increase if efficient water use allows the expansion of irrigated agriculture. Land-use shifts from arid native vegetation could sequester 8.0 Mg C ha^{-1} and assuming 10% expansion of irrigated agriculture, 7.2×10^6 Mg C (0.01% of the total C emitted in the next 30 years) could potentially be sequestered in Pacific Northwestern soils.

Cochran et al. (2007) reported that converting the native shrub–steppe to irrigated organic vegetable production (sweet corn–pea rotation) in the Columbia Basin of eastern Washington increased SOC 0.7 g C kg^{-1} soil above the native soil on an Adkins silt loam (Xerollic Camborthid) after the first year of cropping. After 3 years of cropping and the incorporation of crop residues (10 Mg ha^{-1}) and stable compost (34 Mg ha^{-1}) SOC increased to 6.4 g g^{-1}, 2.1 g C kg^{-1} soil above the native soil in the surface 20 cm layer. The increase in SOC was attributed to the resistant nature of the added compost. Acid hydrolysis of the compost showed that 73% of compost C comprised the resistant fraction. This fraction was composed of aromatic humics and lignin which are slow to decompose. Laboratory incubation of the compost showed that 4.3% of the total C was mineralized, suggesting that 0.4 Mg C ha^{-1} could be lost during the growing season though decomposition. Collins et al., (2010) found that 18 years of cultivation of an irrigated Quincy sand (Xeric Torripsamment) in eastern WA consisting of potato-based rotations that included corn and wheat increased soil C 30% above the native SOC (3.0 g kg^{-1}).

Soils developed under native vegetation in the Sandhills of Nebraska with low values of SOC showed increased C concentrations after 15 years of irrigation while soils with higher initial C and N concentrations decreased only slightly (Lueking and Schepers, 1985). Gillabel et al. (2007) found that irrigation increased C inputs compared with no irrigation in semiarid southwestern Nebraska. They found that SOC content in the top 20 cm was almost 25% higher under irrigation (33.0 Mg C ha^{-1}) than under dryland production (26.6 Mg C ha^{-1}). Further, they reported an average C accumulation rate of 0.19 Mg ha^{-1} yr^{-1} over 33 years for the 0 to 20 cm layer under irrigation compared with no irrigation. This was at the lower end of estimates by Eve et al. (2002) but 20% higher than the values estimated by Lal et al. (1998).

King et al. (2009) reported that furrow irrigation in a California field had a positive effect on SOC storage, showing a net increase of 0.03 Mg C ha^{-1} yr^{-1}. This increase was in line with current estimates of C sequestration in global agricultural systems that range between 0.03 and 0.65 Mg C ha^{-1} yr^{-1} (Christopher and Lal, 2007; Hutchinson et al., 2007; Izaurralde et al., 2007). Poch et al. (2006) found similar results on SOC storage (0.02 Mg C ha^{-1} yr^{-1}) for a furrow-irrigated corn field in the central valley of CA. These C gains were 20% lower than the estimated yearly C sequestration rates for other long-term experiments (Eve et al., 2002). Wu et al. (2008) found that SOC concentrations were nearly 40% greater in an irrigated San Joaquin Valley soil above the C stored in native soil after 55 years of irrigated farming. After 90 years of irrigation, SOC increased 70% above the native SOC. The soil inorganic C (SIC) concentrations to a depth of 1 m showed an opposite trend: a decrease after 55 years of irrigation in the San Joaquin Valley but an increase in Imperial Valley after 85 years.

Major shifts in crop production will occur as farmers gear up to supply the demand for biomass feedstocks supporting biofuel production. These shifts will change agroecosystem services related to water use, carbon use and storage, nutrient cycling and greenhouse gas (GHG) emissions that have direct consequences on air, water, and soil quality. Perennial herbaceous crops such as switchgrass (*Panicum virgatum* L.) are important sources of cellulosic biomass for the bioenergy industry. Removal of this biomass may adversely affect C dynamics in soil. Collins et al. (2010) used the natural ^{13}C abundance of soils to calculate the quantity and turnover of C inputs in irrigated fields cropped to switchgrass. Three years of irrigated switchgrass on a Quincy sand soil (Xerollic Torripsamment) showed a 20% increase in SOC in the 0 to 15 cm depth increment with no significant change below 15 cm. The average accrual rate of switchgrass-derived SOC was estimated at 0.5 Mg ha^{-1} yr^{-1}. Mean residence time (turnover) for the active (C_a) and slow (C_s) C pools averaged 27 days and 2.4 years, respectively. Estimates of the mean residence time of the native C_3–C under the irrigated C_4 monocultures of switchgrass were greater than 60 years in the 0–15 cm and 30–55 years in the 15 to 30 cm depth increments. Liebig et al. (2008) also found accrual rates of 1.1 Mg C ha^{-1} yr^{-1} with much of that increase occurring in the surface 30 cm. Zan et al. (2001) found that switchgrass increased SOC by 3 Mg ha^{-1} yr^{-1} when compared to a corn field after 4 years of production. They further reported that corn biomass C contributed 0.8 Mg ha^{-1} yr^{-1} to SOC.

GAPS IN KNOWLEDGE

Long-term research designed to quantify soil organic C content under different management practices and agroecological conditions is limited. Rates of SOC sequestration may peak within 5 to 10 years and approach a new steady state 20 to 50 years following a management change (West and Post, 2002) or until the soil storage capacity is reached (Lal, 2004). Establishment of long-term sites with major agricultural systems and alternative management to business as usual for the diverse cropping systems of the western U.S. is needed. Overall it appears that (1) SOC databases are lacking for many areas of western U.S. dryland cropping systems; (2) baseline sampling of SOC prior to management practice is largely nonexistent; (3) soil erosion processes having large impact on SOC have not been quantified; and (4) inconsistent sampling methodologies and analyses have been used, thereby contributing to large variability in potential SOC sequestration rates (Brown and Huggins, 2011).

Sampling Methodology

Sampling biases can arise if disparate soil sampling occurs, such as sampling after a recent addition of biomass from residues after harvest. Crop residues can mix with soil sample that might overestimate SOC concentration. We suggest that soil sampling should take place prior to C inputs from crop residues. In the northern Great Plains, spring soil sampling, however, may not be feasible due to limited time available for planting crops. If a fall sampling is used for C analysis, it is suggested that soil sampling be done consistently after crop harvest in the fall every year to reduce any seasonal variation.

Depth of Sampling

Changes in SOC following a shift in agricultural management practice are often more pronounced at the surface than the subsurface layer (Follett, 2001; West and Post, 2002). However, changes occurring in the subsurface layer and the whole soil profile (minimum of 150 cm) need to be accounted for. The depth of soil sampling can have a major effect on changes in soil C stock among treatments (Franzluebbers, 2010). VandenBygaart et al. (2011) reported that soils should be sampled at different depths for evaluating the effects of management practices on SOC in agroecosystem experiments. They recommended that for evaluating the effect of fallow on SOC, soils should be sampled to a depth of 15 cm, for tillage to a depth of 30 cm, and for perennial crops to a depth of 45 cm. This is because of the differences in statistical significance observed among various management practices at various depths. For example, incorporation of crop residue to a depth below 15 cm due to tillage results in greater C storage in the subsoil layer. Similarly, greater root biomass of perennial than annual crops results in greater SOC below 30 cm. Kravchenko and Robertson (2011) and Syswerda et al. (2011) found that most of the significant differences in SOC among management practices occurred at the surface compared with subsurface soil layers or the whole soil profile due to high variability in the subsurface layers. As a result, they suggested that treatment effects on SOC should be evaluated at the individual soil depth rather than the whole soil profile.

Labile Carbon

Carbon stored in undecomposed and partially decomposed crop residues has largely been ignored when assessing soil C sequestration. In fact, attempts have been usually made to remove labile C, as it is assumed that these residues will quickly decompose and not contribute to stored SOC. However, observations of long-term NT fields suggest a different reality. Annual additions of surface residue and roots in NT management can result in the accumulation of a mulch layer that might influence SOC. The CropSyst model was used to simulate differences in C storage between tillage and cropping practices (C. Stockle, Washington State University, Personal Communication). Model results showed that SOC increased in 5 to 6 years in NT due to addition of crop residue and roots. For example, the model predicted an increase of 1.0 Mg C ha^{-1} in NT compared to CT in St. John and Pullman, WA. The size of this residue C was influenced by tillage crop rotations. For example, near Pullman, increases in residue C were predicted to be 1.2 Mg C ha^{-1} under continuous cereals compared to 0.7 Mg C ha^{-1} under peas every 3 years. Although labile C pools are sensitive to annual changes in C inputs, they represent a new steady state of C stocks that were absent under tillage.

Some of the parameters used to measure changes in SOC and biological soil quality are particulate organic C (POC), microbial biomass C (MBC), and potential C mineralization (PCM). Since SOC has a large pool size and inherent spatial variability, it changes slowly with management practices (Franzluebbers et al., 1995). As a result, measurement of SOC alone does not adequately reflect changes in soil quality and nutrient status (Franzluebbers et al., 1995; Bezdicek et al., 1996). Measurement of biologically active fractions of SOC, such as MBC and PCM that change rapidly with time, could better reflect changes in soil quality and productivity, because they alter nutrient dynamics due to immobilization–mineralization (Saffigna et al., 1989; Bremner and Van Kissel, 1992). These fractions can provide an assessment of soil organic matter changes induced by management practices, such as tillage and cropping systems (Campbell et al., 1989). Similarly, POC has been considered as an intermediate fraction of SOC between active and slow fractions that changes due to management practices (Cambardella and Elliott, 1992). The POC also provides substrates for microorganisms and influences soil aggregation (Six et al., 1999).

Inorganic Carbon

Studies on soil C sequestration have been focused mostly on changes in SOC. However, in dryland cropping systems, soil inorganic C (SIC) can dominate C storage more than SOC at the subsoil layers. In eastern Montana, observations have been noted about increasing SIC and decreasing SOC with depth (to 120 cm) under dryland cropping systems (unpublished data). Little is known about the effects of management practices on SIC storage.

Sainju et al. (2007a) reported greater SIC at 0 to 20 cm depth in NT continuous spring wheat than in CT spring wheat–fallow in eastern Montana. They speculated that surface placement of fertilizer and lower soil water content probably increased SIC in NT continuous spring wheat compared with CT spring wheat–fallow. Cropping can increase the formation of $CaCO_3$ compared with fallow by increasing plant and microbial activities, thereby increasing SIC (Cerling, 1984; Monger, 2002). Addition of soil amendments, such as fertilizer containing Ca that could lead to increased formation of $CaCO_3$, can also increase SIC (Amundson and Lund, 1987; Mikhailova and Post, 2006). In contrast, Cihacek and Ulmer (2002) found that SIC at the surface soil was greater in CT than in NT soils in the northern Great Plains.

Because of the higher concentration of SIC, especially at lower depths, under dryland cropping systems, soil total (organic + inorganic) C instead of SOC alone may be needed to evaluate the effects of management practices on soil C storage. Further studies are needed to examine if soil total C instead of SOC could be used for measuring changes in soil C storage as affected by management practices, especially under dryland cropping systems.

Erosion

Most comparative studies of management impacts (e.g. tillage, crop rotation) on SOC budgets emphasize differences in biological processes (e.g. primary production, decomposition) that operate at the pedon scale. In many situations, however, historic and current soil erosion processes have likely increased variability in profile and landscape-scale SOC distribution (Pennock et al., 1994). These are increasingly recognized as major determinants of field-scale SOC budgets (De Jong and Kachanoski, 1988; Starr et al., 1999; Fang et al., 2006; Chaplot et al., 2009). Erosion processes may be as important as other ecosystem processes controlling C flux, such as tillage and microbial decomposition (Christopher and Lal 2007; Izaurralde et al., 2007). Van Oost et al. (2007) also suggested that erosion may also lead to the stabilization of C in depositional sites through burial. Erosion-induced deposition and buried C may amount from 0.4 to 0.6 Gt C yr^{-1} (Lal, 2004).

Conversion of native vegetation to cropland in the Palouse region of the Pacific Northwest has resulted in historical soil erosion rates of over 25 Mg ha^{-1} yr^{-1}. This may have removed 100% of the topsoil from 10% of the landscape, with another 25 to 75% loss from 60% of agricultural land (USDA, 1978). Purakayastha et al. (2008) estimated that 50 to 70% of SOC had been lost from upland soils. Not often recognized or measured, however, is the effect of soil erosion processes on within-field soil deposition and likely site-specific SOC levels.

Soil erosion is a selective process and transported sediments are enriched in clay-sized particles (Ongley et al., 1981) and particulate and dissolved organic C (Lal, 1995). Erosion-related processes laterally redistribute SOC, thereby removing and/or depositing SOC from different field locations. This can be as important as biological processes for determining SOC stocks at a specific site. Huggins et al. (2011) concluded that historical soil erosion processes in dryland farming regions of the inland Pacific Northwest challenge the common expectation that SOC will be linearly related to C inputs (Larson et al., 1972; Rasmussen and Collins, 1991; Huggins et al., 1998). In this study, the close proximity of areas with low and high surface to subsoil SOC ratios indicated a coupling of soil detachment, transport, and deposition within the field that contributed to the large within-field variability of SOC stocks.

Redistribution of SOC due to erosion processes (historical or contemporary) has important implications for methods used to assess changes in SOC due to management practice. Baseline sampling at the initiation of the experiment is required to evaluate treatment-induced changes in SOC over time at a given location. Soil sampling must occur throughout the soil profile at targeted depth increments to capture all SOC changes, along with associated bulk density to express SOC on a mass per unit volume basis. Furthermore, the importance of sampling at various depths that extend beyond the surface is important for evaluating changes in SOC when shifting tillage from CT to NT in eroded soils (Huggins et al., 2007; Baker et al., 2007; Huggins et al., 2011). Coupling baseline and depth-increment sampling representing landscapes over time will enable field-scale assessment of treatment effects on SOC that include biological as well as physical processes (VandenBygaart et al., 2002, 2006; Huggins et al., 2011). The sampling methodologies described will lead to greater understanding of field-scale variations of SOC that arise from the interaction of biophysical processes (e.g. C inputs from crop residues and roots; decomposition; soil erosion driven by water, wind and tillage).This will be important to quantifying SOC sequestration, for developing sophisticated SOC models, and for promoting improved land use and management decisions for precision conservation practices.

SYNTHESIS

For decades, predominant dryland cropping systems in the Great Plains of western U.S. have been winter wheat–summer fallow management in the CT system. The system promoted SOC decomposition and soil losses through erosion. It has been well documented that tillage (1) enhances residue decomposition by incorporating the residue into the soil, (2) exposes previously protected SOC to soil fauna by destroying soil aggregates, and (3) increases soil losses due to wind and water erosion. Adopting NT increases residue accumulation and surface SOC due to less soil disturbance, less residue incorporation and oxidation, and decreased risks of soil erosion, in addition to improving soil water content during the fallow period compared with the CT system. Although only partly adopted in the western U.S., NT has shown improved soil water conservation and increased SOC storage. Increased cropping intensity and crop rotation that includes perennial crops combined with reduced tillage and fallow periods can increase the amount of crop residue returned to the soil, increase SOC content, and reduce the potential for soil erosion.

Irrigation can increase crop yields and economic viability of agriculture in arid and semiarid environments where plant growth is limited by available water. Irrigation also increases C input to soils via increased litter and root production. However, the potential of irrigation to cause a net increase of C storage is tempered by C loss as CO_2 emitted to the atmosphere as a result of (1) fertilizer manufacture, storage, transport, and application, (2) pumping irrigation water, (3) farm operations such as tillage and planting, (4) dissolved carbonate in irrigation water, and (5) increased C mineralization from soil organic matter and crop residue due to increased microbial activity as a result of greater soil water content. The SOC levels, however, will be determined by a balance between C inputs from crop residues and soil amendments and rate of C mineralization. Intensively managed irrigated crop or pasture lands have the potential for C gain through the use of improved grazing regimes, improved fertilization practices, and irrigation management. Irrigated lands produce approximately twice as much plant biomass as rain-fed agricultural production systems. A substantial reduction of atmospheric CO_2 could be attained if policy makers and agricultural experts recognize the potential benefit of land and water management strategies. Lands could be more purposely used for their greatest good, be that food production, carbon storage, native habitat, or other uses.

We recommend that future research directions to improve C storage in western agricultural systems over the next 5–10 years include:

- Increased precision farming of crops
- Greater use of buffer zones
- Increased use of no-till
- Increase in fertilizer use efficiency
- Precision irrigation and fertilizer application
- Development of organic amendments and fertilizers
- Better organic farming practices (BMPs)
- Increase in tree fruit fertilizer use efficiency

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