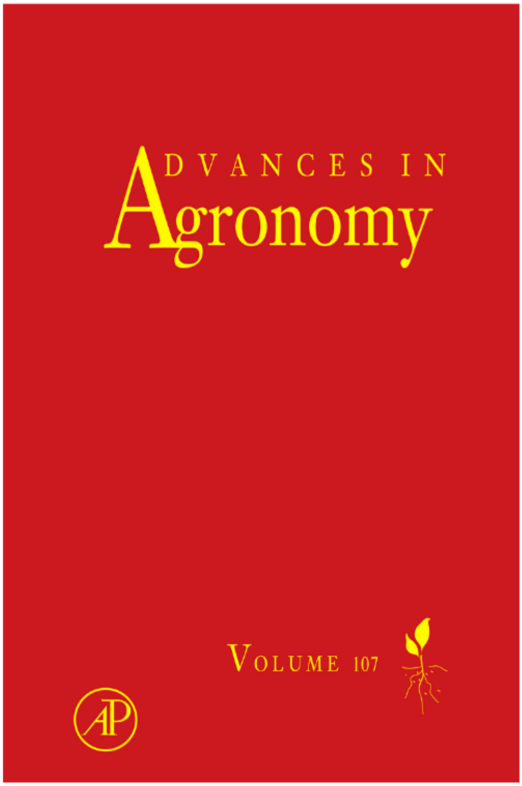

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CHAPTER FOUR

The Potential for California Agricultural Crop Soils to Reduce Greenhouse Gas Emissions: A Holistic Evaluation

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Contents

Abstract

Climate change predictions for California indicate that agriculture will need to substantially adapt to reduced water availability, changing crops, and changes in temperatures, in order to sustain the level and diversity of crop production in

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California. California legislators recently passed the California Global Warming Solutions Act of 2006 (AB 32) that requires all industries to reduce the three major greenhouse gases (GHGs) (CO_2, N_2O, and CH_4) to 1990 levels by 2020. The great diversity of cropping systems and management practices in California agriculture leads, however, to greater uncertainties in estimates of GHG budgets compared to Midwest agriculture. In light of AB 32, we, here, synthesize all the available information on the potentials for California agriculture to sequester C and reduce GHG emissions through various alternative management practices: minimum or no tillage, organic, cover cropping, manuring, and reduced chemical fertilizer management. Our review indicates that C sequestration and GHG emission reductions are possible, but there is no single land management practice or change in inputs that could mitigate the C released from agricultural practices (e.g., fossil fuel usage, land-use changes, soil erosion, biomass burning, and N fertilizer associated emissions) and meet climate change commitments set out in AB 32. Therefore, it is only the integration of different management strategies that shows considerable potential for C mitigation as well as provides important cobenefits to ensure the future sustainability of California agriculture.

1. Introduction

It is well documented that increases in atmospheric carbon dioxide (CO_2) and other radiatively active trace gases, such as nitrous oxide (N_2O) and methane (CH_4), released from anthropogenic activities, including fossil fuel combustion, energy generation, transportation, land-use change, and agricultural practices, lead to climate change (IPCC, 2007; Kerr, 2005). Increases in greenhouse gases (GHGs) have prompted the current generation of policy acts, frameworks, and intergovernmental reporting, such as the Kyoto Protocol (1997), The United Nations Framework Convention on Climate Change (UNFCCC), and the Intergovernmental Panel on Climate Change assessments (IPCC, 1990–2007). The procedures set out by the Kyoto Protocol enable participating nations to offset any CO_2 emissions by implementing strategies and guidelines for all contributing sectors in which net carbon (C) enhancement and storage may be achieved with an overall ensuing reduction in GHG emissions.

Despite the fact that the United States is not a party of the Kyoto Protocol agreement, certain states, such as California, have been compelled to pass legislation that mandates the reduction of GHG emissions with the potential for a carbon cap and trade/taxation system. In California, more specifically, predictions of future climate change have indicated that climate change could seriously impact natural and agricultural ecosystems (Hayhoe *et al.*, 2004), therefore, the State of California passed the California Global Warming Solutions Act of 2006 (AB 32). This directive requires the reduction of the three major GHGs (CO_2, N_2O, and CH_4) to 1990 levels by 2020 by the utilities and manufacturing industries. Currently, for the agricultural industry, AB 32 and GHG reductions are on a voluntary basis. The voluntary reduction of GHG from agriculture would, however, provide agriculture the opportunity to engage in the carbon market.

Agricultural land use, land-use changes, and forestry systems contribute to the GHG balance by playing an important role in the production and consumption of GHGs. Conventional agricultural management practices (e.g., conventional tillage, high synthetic nitrogen fertilizer inputs) can lead to the loss of soil organic carbon (SOC) and/or greater fuel consumption, leading to increased CO_2 and other GHG emissions (Lal, 2002; Post and Mann, 1990; Schlesinger, 2000). Nevertheless, agricultural soils also have the ability to store C and may potentially offset GHG losses to the atmosphere (Lal, 2004a,b, 2007; Paustian *et al.*, 1997).

Both conventional and sustainable management strategies (e.g., minimal or no tillage (NT) practices, residue management, cover cropping) need to be further investigated in order to understand the processes affecting soil C and to establish the possibility of agricultural soils in California to sequester C and therefore offset and reduce GHG emissions. If this possibility exists, it would lead to the adaptation of mitigation strategies, and thereby reduce adverse effects of global climate change and improve soil quality for sustainable crop and food production.

1.1. The importance of California agriculture

California has a wide range of climate regions and ecosystems (e.g., croplands, forests, coastal margins, mountainous areas, and desert). California agriculture is incredibly diverse because of its varied microclimates that allow for a wide variety of annual (vegetables and cereal) and high-value specialty perennial crops, such as citrus, nuts, stone fruits, and grapes. California accounts for approximately 43% of the vegetable and fruit production and 42% of nut production in the United States (CFBF, 2002). Agricultural production is also especially important within California as it generates a great percentage of jobs (7.3%) and global commerce (CFBF, 2002). In California, approximately 8.5 million acres are devoted to harvested crop land, where approximately 34% of that area is planted to orchards and vineyards, 23% is devoted to hay, with another 14% of the total being under vegetable crops (CFBF, 2002). Despite the apparent importance of agriculture within California there is little data on GHG emissions occurring from agricultural land (e.g., Bemis and Allen, 2005). There is even less data on potentials of C sequestration and GHG reduction in Californian agriculture (e.g., De Gryze *et al.*, 2009).

California is the 16th largest emitter of GHGs in the world, accounting for approximately 500 million metric tons of CO_2 equivalents per annum

(Bemis and Allen, 2005). It is well documented that, globally, agriculture has been a substantial source of GHG emissions but that a potential for GHG reductions currently exists within agriculture (Cole *et al.*, 1993; Follett, 2001; Jawson *et al.*, 2005). Within California, it is estimated that agricultural practices contribute to approximately 8% of the total GHG emissions (Bemis and Allen, 2005). The climate within California is characterized as Mediterranean, with long hot dry summers and wet cooler winters. Recent model estimates for the potential effect of climate change in California predict that there will be a substantial increase in temperature, especially significantly higher summer temperatures (Hayhoe *et al.*, 2004; Snyder *et al.*, 2002). These changes will ultimately affect agricultural ecosystems, water, and energy demands within the state (Cayan *et al.*, 2006). For example, rising temperatures can decrease yields for many of California's crops such as lettuce, rice, and tomatoes (Lee *et al.*, 2008). Furthermore, a shift in precipitation patterns and quantity within the state are expected; however, no clear trend in how precipitation will change has emerged from model scenarios and is, therefore, still an area of high uncertainty (Hayhoe *et al.*, 2004; Snyder *et al.*, 2002). Nevertheless, it is clear that climate change will have an impact on water resources through reduction in mountain snowpack and earlier runoff (Cayan *et al.*, 2006). Consequently, addressing GHG emissions reductions and C sequestration potentials within California agriculture is vital to maintaining the industry.

Agricultural management practices within California are highly intensive because the vast majority of crop acreage is cultivated using standard tillage (ST) operations, high inputs of synthetic nitrogen fertilizers, and intensive furrow irrigation. However, implementing alternative management practices and reducing inputs would likely provide an opportunity to reduce GHG emissions and partly address climate change issues both locally and globally. Furthermore, developing alternative management strategies, such as conservation tillage (CT) practices, cover cropping, organic management, residue management strategies, reduced synthetic nitrogen fertilizer input, and improved irrigation systems, have obvious cobenefits (see Section 5). Previous research and reviews of C sequestration potentials in agriculture within the United States have largely focused on the Great Plains and the Corn Belt (Follett, 2001; Lal, 2002, 2004a,b; Swift, 2001). None of the previous summary studies thoroughly includes California (Lal, 2003; Six *et al.*, 2002; West and Marland, 2002), and until recently, there was very little ground-based field data available to quantify the interactions and impacts of irrigated farming with alternative practices on soil C dynamics and GHG emissions (Follett, 2001). The fundamental differences in agricultural management and climate (i.e., Mediterranean) found within California compared to the rest of the United States prevents a direct extrapolation of Midwest findings to California. Therefore, the objectives of this paper are to

(1) highlight current research on GHG reduction within California agriculture, (2) discuss the principles regulating GHG fluxes in irrigated agriculture under the Mediterranean climate of California, (3) critically review the capability of Californian soils to sequester C and reduce GHG emissions through the adoption of recommended alternative practices, and (4) identify areas where research is still lacking.

2. Basic Concepts of Carbon Sequestration

The soils of the world contain approximately 1550 Pg SOC and approximately 750 Pg soil inorganic carbon (SIC) (Lal, 2002; Schlesinger, 1997). The soil pools of C are greater than the atmospheric pool (750 Pg C) and the terrestrial vegetation biotic pool (600 Pg C) put together. Estimates of the SOC pool within the United States (contiguous 48 states only) range from 59.4(Waltman and Bliss, 1997) to 78–84.5 Pg C (Kern, 1994) or about 5% of the global terrestrial C stocks.

The levels of SOC currently in soils reflect the long-term equilibrium between any C input and output. Farming practices influence both the input of C to the soil through organic amendments and the output of C through influencing decomposition and, therefore, the release of SOC to the atmosphere as CO_2. In short, when the output of C (CO_2 emissions) is greater than the input (e.g., crop residues) to the soil system, SOC will decline. Agricultural soils are generally lower in SOC (Haas *et al.*, 1957) because cultivation leads to SOC losses through (1) disturbance of the soil via extensive tillage practices, (2) reduced diversity and inputs of plant residues, and (3) intensive use of nitrogen fertilizers and irrigation water (Khan *et al.*, 2007; Lal, 2002; Paustian *et al.*, 1997; Schlesinger, 2000; Six *et al.*, 1999; West and Post, 2002). Farming practices contribute to the accelerated mineralization of SOC by changing soil physical and chemical characteristics, such as soil moisture (e.g., through irrigation), temperature, nutrient availability (e.g., addition of synthetic nitrogen fertilizers) (Paustian *et al.*, 2000), oxygen content (Rovira and Greacen, 1957), and structural stability (e.g., Kong *et al.*, 2005). This significant depletion of SOC will eventually reduce the quality of the soil, which in turn will contribute to a reduction in crop production, increased emissions of GHGs from the soils, and negative impact on the surrounding environment (e.g., eutrophication of water ways) (Lal, 2008). Furthermore, under current standard management practices, SOC is predicted to be further depleted because of the projected increases in global temperatures (Lal, 2004a,b, 2008), whereby a strong correlation exists between SOC losses and climate change driven increases in soil temperatures (Kirschbaum, 1994, 2000).

There are numerous physical, chemical, and biological processes affecting the stabilization (and destabilization) of SOC (see Fig. 1). These processes are dependent upon climate, soil type, vegetation, and previous management practices and the principle soil properties affected by management are temperature, moisture, aeration, and aggregation (Lal *et al.*, 1995). The stabilization of soil C is dependent upon soil structure which is the organization of both aggregates and the pore spaces surrounding them (Tisdall, 1996). Stabilization of soil C by aggregates is dependent on the interaction between soil particles, soil microbial activity, and above- and belowground C inputs. The formation of a stable soil matrix decreases the accessibility of substrates to microorganisms and thereby reduces the decay rate of substrates. Hence, from a management standpoint, conservation or reduced tillage practices can lead to a stabilization of soil C because the decreased mechanical disturbance of the soil decreases the disruption of soil aggregates (Beare *et al.*, 1994; Elliott, 1986; Six *et al.*, 1998). The reduction in tillage intensity also allows for the reformation of soil aggregates and thus compensates for the increased turnover of aggregates induced by tillage (Kong *et al.*, 2005; Six *et al.*, 2000). Additionally, Kong *et al.* (2005) showed a strong relationship between the stability of soil aggregates, C input, and SOC at the Long-Term Research on Agricultural Systems (LTRAS) site in Davis, CA. This relationship between C input and SOC sequestration

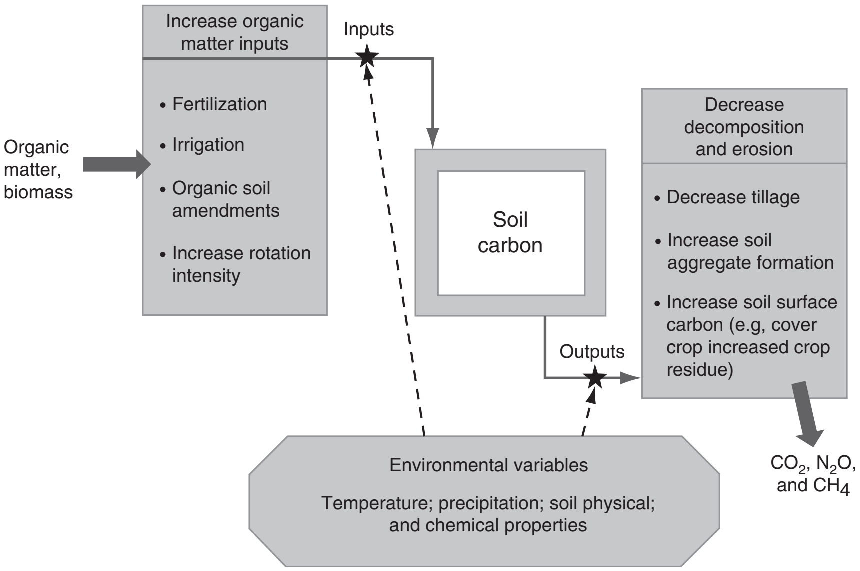

Figure 1 Environmental and management inputs and outputs that affects soil carbon sequestration. Adapted from West and Six (2007).

across different management treatments has also been shown in the Sustainable Agriculture Farming Systems (SAFS) site in Davis, CA (Poudel *et al.*, 2001).

The basic technologies by which C can be sequestered in agricultural soils have been reported extensively (e.g., Cole *et al.*, 1993) and include agroecosystem management strategies such as CT or NT, organic farming, the conversion of fallow land to permanent vegetation, winter cover cropping, application of manures or sewage, greater inclusion of hay crops in the rotation, reduced nitrogen fertilizer application, and improved irrigation methods (e.g., drip irrigation) (Follett, 2001). By restoring degraded soils and by adopting sustainable practices it has been estimated that the C sequestration potential of the world's cropland soils is approximately 0.73–0.87 Pg yr^{-1} (Lal and Bruce, 1999). This estimate is based largely on studies conducted in non-Mediterranean ecosystems; further research is needed to establish the actual amount of C that California agricultural soils can sequester over time under the same conservation management practices as studied elsewhere.

3. Land Conversion and the Effect on Carbon Sequestration

The current worldwide coverage of irrigated agriculture following conversion of native natural soils is greater than 271 Mha. It is expected to increase with the need to convert marginal lands to address the increasing food demand of the growing world population (Bruinsma, 2003). The initial conversion of native soils to cropland in the United States released approximately 5 Pg C with more soil C released with the expansion of agricultural land (Lal *et al.*, 1998). In the recent past, California has experienced huge shifts in land use for two reasons. Firstly, increases in population (e.g., a rise of 50% in the last 30 years), where this trend is predicted to rise from 37 million to between 42 and 48 million by the year 2025 (PPIC, 2008), has led to rapid urbanization rates. Secondly, the increased relative profitability and demand for specialty commodity crops such as grapes and almonds, both of which can be grown on marginal lands, has led to an increased conversion of native grasslands and oak woodlands to agriculture. Between 1969 and 1998, approximately 45,675 ha of timberland (i.e., predominantly oak woodland), rangeland, and abandoned agricultural land was converted to other land uses. Approximately 4% of the total converted land was for agricultural purposes (Shih, 2002). For example, from 1991 until 1999, an estimated 480 ha of the 4% total was converted into vineyards (Shih, 2002), and today there are approximately 360,000 ha of grape vineyards in California.

A study by Carlisle *et al.* (2006) investigated the effect of the conversion of oak woodland to a vineyard on soil CO_2 respiration and soil C. They hypothesized that the conversion of oak woodlands to vineyards would increase the amount of C respired from recalcitrant sources within the vineyard soils compared to the woodland soils and that the amount of soil respiration would decline under the cultivation of a perennial crop like grapevines, because the long-term cultivation of such a crop would reduce the quantity of the nonrecalcitrant carbon within the plough layer. They observed higher seasonal rates of soil CO_2 respired from the oak woodland; however, during the dry season, fluxes of CO_2 were comparable to the vineyard. Annual emissions of C from the vineyard soils was 7.02 ± 0.58 Mg C ha^{-1} yr^{-1}, whereas the oak woodland lost approximately 15.67 ± 1.44 Mg C ha^{-1} yr^{-1}, indicating that the oak woodland before conversion emitted significantly more soil CO_2 compared to the vineyard (Carlisle *et al.*, 2006). The comparatively lower rates of CO_2 emissions from the vineyard soil may have been a result of the soil having lower total C and thus less available labile C to be respired (Carlisle *et al.*, 2006).

It is well documented that the conversion of natural ecosystems to agriculture will lead to an immediate loss in SOC and temporarily enhances evolution of CO_2 to the atmosphere. However, this study measured lower emissions of CO_2 from the vineyard soils compared to the oak woodland after 30 years following conversion. These lower emissions, however, could be attributed to both changes in physical soil properties through conversion and cultural practices, which could affect CO_2 diffusion in the upper soil layer, as well as to the deeper root distribution in vineyards than oak woodlands (Smart *et al.*, 2005). It was also noticed that cultural practices such as tillage and the overall preparation of the vineyard had significant impacts upon the soil C pools (Carlisle *et al.*, 2006). Therefore, further research is needed on the effects of such practices on perennial crops and how conversion of timberland to agricultural perennial crops affects the flow and storage of C in such ecosystems.

3.1. Conservation tillage and carbon sequestration

In the Midwestern United States, standard conventional tillage regimes are known to cause a decrease in soil C stocks through the enhanced mineralization and erosion of C rich soil material (Lal, 2004a,b; Swift, 2001). The general objective of all types of conservation practices is to enhance soil quality and reduce soil loss. A major indicator for improved soil quality is increased SOM levels because SOM influences plant yield and environmental impact by providing and recycling nutrients, improving water infiltration and retention as well as stabilizing soil structure (Lal, 1997). Several CT systems have been developed in the past few decades and include ridge tilling, mulch tillage, strip tillage, and NT (Blevins *et al.*, 1998).

Within the United States, a decrease in ST practices for planted cropland areas and a rise in CT usage has occurred over the past 20 years. For example, NT practices have tripled in acreage (from 6.8 to 21.1 Mha) in the last decade alone (Post *et al.*, 2004). Numerous studies within the Midwestern United States have reported the benefits of reduced or CT (Kern and Johnson, 1993; Puget and Lal, 2005; Rasmussen and Collins, 1991; Sperow *et al.*, 2003; West and Marland, 2002). Reducing tillage has been cited as a sustainable practice because it reduces fossil fuel usage, labor needs, and also improves aspects of soil quality by decreasing soil erosion and improving water conservation (Lal, 2003).

Follett (2001) estimated that CT management of cropping systems in the United States has the potential to sequester 30–105 million metric tons carbon (MMTC) yr^{-1}. Lal and Kimble (1997) and Paustian *et al.* (1997) suggested that the conversion to CT practices can potentially sequester 0.1% ha^{-1} yr^{-1} CO_2 from the atmosphere in the top 5 cm of the soil layer and could potentially sequester approximately 10 tons C ha^{-1} over the next 30 years. Furthermore, a meta-analysis conducted by Ogle *et al.* (2005) showed that the conversion of ST to NT increased SOC storage in tropical moist climates by a factor of 1.23 ± 0.05 and by a factor of 1.17 ± 0.05, 1.16 ± 0.02, and 1.10 ± 0.03 for tropical dry, temperate moist, and temperate dry climates, respectively. Six *et al.* (2004) also observed an increase in SOC between humid and dry climates following conversion from ST to NT. Stocks of SOC increased 10 years following conversion for both climate regimes; however, a significantly higher increase of 222 kg ha^{-1} yr^{-1} in the humid regime as compared to 97 kg ha^{-1} yr^{-1} in the dry regime. However, recent studies have indicated that C sequestration drastically varies among different CT practices and with soil depth (West and Post, 2002). Lal *et al.* (1998) found that within the top 20 cm, NT practices had the potential to sequester approximately 500 kg C ha^{-1} yr^{-1}, while mulch and ridge till systems within the United States could sequester 600 kg C ha^{-1} yr^{-1}. At shallower soil depths of 7.5 cm in the Midwest, Robertson *et al.* (2000) measured a potential sequestration rate of 300 kg C ha^{-1} yr^{-1} for NT systems. A similar value was reported by West and Marland (2002) in NT systems. Also within the Great Plains region of the United States, Follett and McConkey (2000) observed a C sequestration range of 300–600 kg C ha^{-1} yr^{-1} for NT, mulch and ridge tillage practices, falling within the ranges also noted by the above cited studies.

In California, there is far less information and research on CT practices than is available for the Midwestern United States. This lack of research is in part due to the challenges in adoption of CT in irrigated systems and the heterogeneity of California agriculture. Currently, only about 10% of total crop acreage in California is under CT, which is up from 3% in 2006 (Horwath *et al.*, 2006). Evapotranspiration demands are extremely high in the Central Valley of California and many row crops use furrow or flood irrigation throughout the growing season. One potential difficulty in assessing the practicality of converting such cropping systems to CT may be due to the likely increases in SOC mineralization which may occur due to a reduction in surface evaporation and accelerated microbial activity when crop residues are left on the soil surface through CT (Unger *et al.*, 1997). Few studies in the past decade have addressed CT within the arid, irrigated agriculture of California where very intensive ST methods are prevalent. One of the main reasons for intensive tillage is to maintain the furrows to convey irrigation water. Generally, long-term and frequent tillage has been associated with increased GHG emissions (Lal, 2002; Reicosky *et al.*, 1997). For instance increased soil surface emissions of CO_2 have been measured shortly after a tilling event (Reicosky and Lindstrom, 1993; Reicosky *et al.*, 1997; Rochette and Angers, 1999). More importantly, tilling also deteriorates and modifies the soil structure (Blanco-Canqui and Lal, 2004; Six *et al.*, 1999) and microbial community, thereby affecting the ability to store SOC and N (Jackson *et al.*, 2003). Reduced crop diversity and fallow associated with intensive cropping systems also reduces significantly SOC and soil N (Seiter and Horwath, 2004).

In accordance with the Midwestern United States studies, a significant release of CO_2 following tillage has also been reported for annual row crops in California (Calderon and Jackson, 2002; Calderon *et al.*, 2000; Jackson *et al.*, 2003). In intensively managed vegetable systems in Northern California, CO_2 emissions increased substantially after the first day of tillage but soil respiration declined shortly after, thus indicating a physical degassing process of CO_2 due to soil disturbance (Calderon and Jackson, 2002; Jackson *et al.*, 2003). Furthermore, they noticed increases in net mineralization and the accumulation of nitrate several days following a tillage event, which in turn could be associated with higher rates of denitrification, the potential for NO_3 leaching and increases in N_2O emissions (Jackson *et al.*, 2003).

Within California's San Joaquin Valley, a 5-year study by Veenstra *et al.* (2007) investigated how CT and cover cropping affected SOC dynamics in an intensively managed cotton (*Gossypium hirsutum* L.) and tomato (*Lycopersicon esculentum* Mill.) rotation. The four treatments included standard tillage and cover crop (STCC), standard tillage and no cover crop (STNO), conservation tillage and cover crop (CTCC), and conservation tillage and no cover crop (CTNO). After the 5 years, they observed an overall lack of C accumulation within the CT systems (CTCC and CTNO) but they did see a total increase of soil C within the top 30 cm of the soil layer in both of the cover crop treatments (STCC and CTCC). A significant proportion of the accumulation of soil C at 0–15 cm depth occurred in the CTCC treatment (4504 kg C ha^{-1}) and within both the 0–15 cm depth (2035 kg C ha^{-1}) and 15–30 cm depth (1799 kg C ha^{-1}) for the STCC treatment. In the two treatments without cover crops (CTNO and STNO), there was no net change in soil C observed.

Puget and Lal (2005) recognized that any increases in soil C relating to CT practices often occurred within the top few centimeters of the soil layer and that, after taking into account the bulk density, no significant changes or accumulation in SOC occurred. Veenstra *et al.* (2007) also found that despite an increase in aboveground crop biomass in the CT treatments, the reduced soil disturbance contributed merely to a redistribution of the soil C rather than an overall accumulation as compared to the ST treatments. A study in central Spain, which has a climate similar to California, observed small increases in C within CT and NT cropping systems (Hernanz *et al.*, 2002, 2009), however, in another study under similar climatic conditions accumulation of SOC was reported to be negligible or nonexistent (Buschiazzo *et al.*, 1998).

No- or reduced-tillage crop production in California is increasing, however, the number of studies are still insufficient to demonstrate changes in soil C sequestration. Most of these investigations have not been carried out over an appropriate time interval of 5–10 years. For example, in a short-term study (i.e., 1 year) in a vegetable crop rotation in California with NT under continuous cropping and ST under fallow, Minoshima *et al.* (2007a) found that NT under continuous cropping resulted in a significant increase in microbial biomass C (MBC) and fungi biomass as indicated by ergosterol and phospholipid fatty acid biomarkers in the soil surface layer compared to the ST under fallow, which is the principal production method for vegetable crops within California. The fungal activity and increases in soil MBC may contribute to greater C accumulation in the soil surface layer compared with the ST with fallow treatment (Minoshima *et al.*, 2007a). Furthermore, in another short-term decomposition experiment, Minoshima *et al.* (2007b) observed similar emissions of CO_2 from both NT and ST systems fallow and continuous cropping of tomato/sorghum/garbanzo/cow pea rotations. However, results showed lower assimilation of newly added C shoot residues into active SOM pools and soil organisms (i.e., nematodes and microbial biomass) within the NT compared to the ST soils. Reasons for the lower assimilation of shoot C from the NT system were likely due to there being less shoot residue contact with the soil surface which along with minimizing soil disturbances in the longer term would enhance assimilation of residue soil C by soil organisms. Therefore, in the long term, NT could be effective at storing SOC in California soils if shoot residues had increased soil contact and could increase the diversity of specific groups of soil biota (i.e., opportunistic colonizer-persistent bacterial feeders) (Minoshima *et al.*, 2007a), which are also associated with the increase in soil C, particularly within the surface layer where the accumulation of C occurs more rapidly in systems under NT (Six *et al.*, 2006).

Moreover CT practices including NT have the potential to reduce CO_2 emissions through a decrease in energy and fossil fuel usage via the reduction in tractor passes made during cultivation and production. Mitchell *et al.* (2006) compared CT and ST production systems for cotton in the San Joaquin Valley and showed a reduction of between 41% and 53% in tractor passes, as well as reducing fuel usage by 48–62% (Table 1) (Mitchell *et al.*, 2006). A reduction in CO_2 emissions via fuel usage reduction calculated according to the IPCC assessment (IPCC, 2006) showed an emission reduction for CT practices such as NT and ridge-till with or without chopped cover crops of greater than 50% compared to ST.

3.2. Cover cropping and carbon sequestration

Cover cropping is a conservation management option which has many benefits (Hartwig and Ammon, 2002; Snapp *et al.*, 2005). Cover crops are generally grown to protect soil from erosion, enhance N supply, and diminish nutrient loss through assimilation. Cover crops may also benefit and protect cash crops by acting as biological control agents, suppressing weed species through competition for light, nutrients, and water as well as providing potential habitats for beneficial insect species which predate upon any pest species detrimental to the cash crop (Bugg *et al.*, 2007). They can also improve soil quality and enhance soil C stocks by adding OM (Sainju *et al.*, 2002).

Table 1 Tractor passes, fuel usage, operating costs, and calculated CO_2 emissions per acre in both standard and conservation systems including cover cropping, tillage systems in cotton crops in the San Joaquin Valley, California (adapted from Mitchell *et al.*, 2006)

Cover crop/tillage system	Standard	NT/Chop	NT	RT/Chop	RT	ST/Chop	ST
Tractor passes over field	17	9	8	9	8	10	9
Gallons of fuel used	19.5	8.5	7.5	8.5	7.5	10.2	9.2
*CO_2 emissions per gallons of fuel used (lbs/gallon)	378	165	146	165	146	198	178
Total operating costs	$237	$199	$195	$199	$195	$204	$200

NT/Chop, no-till with chopped cover crop; NT, no-till; RT/Chop, ridge-till with chopped cover crop; RT, ridge-till; ST/Chop, strip-till with chopped cover crop; ST, strip-till.
*CO_2 emissions calculated according to IPCC (2006) and are based upon the gasoline carbon content per gallon (2421 g), which is equal to 19.4 lbs of CO_2 emissions.

In California, cover cropping has been practiced over the past several decades (Blake, 1991). For instance, in the Californian grape industry the most important use of cover crops have been to improve soil water infiltration, reduce dust emissions, provide an extra source of nutrients, provide a habitat for beneficial organisms, and weed suppression (Auburn and Bugg, 1991). In general, cover crops have a great potential to restore, maintain, and sustain the productivity, fertility, and quality of soils (Bruce *et al.*, 1991; Reicosky and Forcella, 1998; Tilman *et al.*, 2002) by increasing both plant biomass and soil nutrients, while increasing SOM levels and biological activity, as well as reducing soil erosion and compaction through the improvement of aggregate stability (Luna, 1998; Seiter and Horwath, 2004).

Recent work within annual cropping systems of California has shown the ability of cover crops to provide additional inputs of OM, benefiting soil quality and increasing the labile C pool (Veenstra *et al.*, 2007). Following the conversion of standard management practices to winter cover cropping under organically managed systems, SOC increased up to 36% after 12 years under California conditions (Horwath *et al.*, 2002). Despite the fact that tillage is increased with some of the alternative practices used in organic production, SOC still increased due to the inputs of animal derived and green manures (Horwath *et al.*, 2002). Increases in SOC were observed during an on farm assessment of a vegetable cropping system in the Salinas Valley, California; Jackson *et al.* (2004) showed that the addition of cover crops along with composted materials resulted in an increase in MBC and microbial biomass nitrogen (MBN), a reduction in soil bulk density and also a reduction in leachable nitrate (NO_3^-–N) in the soil profile from 0 to 90 cm depth compared to treatments that received no organic matter inputs from cover cropping. Furthermore, another study in California by Poudel *et al.* (2002) showed that the use of cover crops reduced fertilizer N losses by approximately 2% as compared to nonwinter cover cropped soils.

Most data on cover cropping in California is focused primarily on annual row cropping systems (Jackson *et al.*, 2003; Mitchell *et al.*, 2008; Veenstra *et al.*, 2007; Wyland *et al.*, 1996). California, however, has a vast expanse of perennial crops, where approximately 1.0×10^6 ha are devoted to perennials such as vineyards (~360,000 ha) and orchards (~650,000 ha). Perennial cropping systems are also important because of their potential to sequester C by cover cropping between row alleys and because their management practices are less disruptive of soil than those used in annual crop systems (Kroodsma and Field, 2006). Perennials also have potential to store C in their woody biomass (Kroodsma and Field, 2006) and extensive deep root systems (e.g., Smart *et al.*, 2005). Despite this potential, far less consideration has been given to these crops in relation to climate change mitigation and C sequestration.

Roberson *et al.* (1991) observed a significant improvement in soil quality, aggregate stability, and water infiltration when a grass cover crop was present within a prune orchard in Butte County, CA. In contrast, a study within a vineyard in the Napa Valley region of California, observed no detectable increases in soil C content with the use of annual grass cover crops in comparison to bare soil (Steenwerth and Belina, 2008a,b). This may have been due to the fact that soil C mineralization rates were significantly high enough to preclude any detectable belowground C increases within the soil (Steenwerth and Belina, 2008a,b) and the short-term nature of the study. There was, however, a notable release of CO_2 from both the ST and cover crop treatments following a tillage tractor pass (Steenwerth and Belina, 2008a,b), most likely due to the oxidation of SOM released from aggregates. Another vineyard study in California did show that after 5 consecutive years of cover cropping total soil C was approximately 40–50% higher (Steenwerth and Belina, 2008a,b), thus indicating that SOC content may increase following cover crop management in perennial systems.

3.3. Organic residue management and nitrogen fertilizer usage

After California Legislature passed the 1989 Integrated Waste Management Act, which required the diversion of green wastes (e.g., trees, twigs, leaves) as one of the single most components in California's waste stream from landfills (Warnert, 1996), organic farming has become more popular with conventional vegetable farmers within the State along with the knowledge that intensive ST practices are resulting in a loss of SOC (Hartz *et al.*, 2000). Green manuring, however, is a common commercial practice within the rice industry in California since the 1950s and is used as an alternative to burning straw residues (Williams *et al.*, 1957). Studies have shown that the use of a winter leguminous cover crop as a green manure can provide a cost effective and efficient source of N to rice (Williams *et al.*, 1957). Organic farming relies heavily upon inputs of organic residues in the form of green manures (i.e., cover crops), plant compost, and composted animal manures added to the soil along with integrated biological pest and weed management, crop rotation, and mechanical cultivation to sustain and enhance soil productivity and fertility without the use of synthetic N fertilizers and pesticides (Rigby and Caceres, 2001).

As mentioned earlier conservation techniques can increase SOC and subsequently can improve soil structure and aggregate stability. By stabilizing soil aggregates, SOM is more protected from microbial decay (Six *et al.*, 1999) where further soil C increases can be signified by the enhanced accumulation of C within various types of soil aggregates (Six *et al.*, 2001). The use of organic amendments, cover crops and manures can lead to SOC accumulation by improving aggregation as well as reducing the need for synthetic fertilizer additions while still being able to provide crops with equally adequate amounts of nutrients. The principal aim of alternative cropping systems including organic farming in order to achieve a sustainable agroecosystem both in terms of economic value and soil productivity from the addition of organic inputs to the soil are to reduce environmental pollution potentials (e.g., in the form of nitrate leaching) while maximizing N use efficiency and providing the crop with sufficient N. Therefore, growers must adjust their nitrogen needs according to amendments added to the soil and their subsequent mineralization rates. Hartz *et al.* (2000) investigated the N and C mineralization rates of 31 manure and compost samples representative of those typically used as soil amendments within vegetable production in California. They observed that the N and C mineralization rates of both manures and composts were relatively low where the recovery of N ranged from 5% to 18% of total N for manures and 8% for composts (Hartz *et al.*, 2000). The relatively low N and C mineralization rates measured within the study suggest that application rates of the organic amendments would need to be sizeable in order to considerably increase short-term N supply and would typically need a long-term regime of repeated applications at lower application rates (Hartz *et al.*, 2000) compared to the growth and incorporation of a leguminous cover crop (Kuo *et al.*, 1997b). Kramer *et al.* (2002) observed that the effectiveness of a winter leguminous cover crop as an N source for the successive crop depends greatly upon temporal N release from the residue during the growing season. They found during a 1-year period maize crop study conducted at the SAFS site, that fertilizer N uptake was higher in the conventional system (additions of synthetic fertilizer only) at 4.3 kg N ha^{-1} day^{-1} compared to 0.6 kg N ha^{-1} day^{-1} from the vetch cover crop in the low input (i.e., synthetic fertilizer and organic N inputs) and organic (organic N only in the form of green manures) systems over the same period (Kramer *et al.*, 2002). At harvest, they observed that the yield of both grain and N did not differ between the three cropping systems which suggests that optimum crop yields can still be achieved using alternative sources of N (Kramer *et al.*, 2002).

The associated cobenefits of organic amendment applications to soil have shown to reduce the need for herbicide usage by reducing weed emergence (Fennimore and Jackson, 2003), and enhance soil quality which in turn provides a better habitat for beneficial soil fauna. For example, the enhanced presence of decomposers, such as earthworms, can be attributed to inputs of organic amendments where higher levels of organic matter are prevalent (Warnert, 1996). The castings and the channels that earthworms create as they travel and move through the soil layers can improve root growth, water infiltration, and the overall physical structure of the soil (Blanchart *et al.*, 1997, 1999; Ketterings *et al.*, 1997). Earthworms have also been shown to stabilize soil organic matter and contribute to the formation of stable soil aggregates (Shipitalo and Protz, 1989). The role and influence that earthworms play on the stabilization of SOC and N dynamics under varying management practices is still largely an area of uncertainty. A study conducted by Fonte *et al.* (2007) at the LTRAS site within conventional (additions of synthetic fertilizer only), low input (synthetic fertilizer and organic N inputs from legume cover crop), and organic (composted manure and legume cover crop) cropping systems, found that earthworms increased the incorporation of cover crop derived C into both the macro- and microaggregates within the low-input treatment. Unfortunately, inconsistent data from the organic cropping system most likely due to high initial background levels of SOC masked any earthworm effects to make any conclusions toward the organic system (Fonte *et al.*, 2007). The results from this study do show, however, that the type of agricultural management practice has an effect upon the role of earthworms and SOC dynamics and stabilization and that further research is needed within the area of organic amendments and earthworms in the California climate.

4. Greenhouse Gas Emissions Modeling and Future Predictions for California Agriculture

The C and N cycles are intrinsically linked to one another and any change in management practice can influence both soil C stores and the release of GHG emissions. It is economically costly and time consuming to continuously measure GHG emissions under diverse crop management systems within agroecosystems. Process-based models can therefore be employed to estimate gas exchanges and changes in soil C for different cropping systems, soils, climates, and management. These biogeochemical models estimate crop production, decomposition, short- and long-term C dynamics, denitrification, nitrification, and N_2O emissions from a wide range of parameters (crop, soil, climate, and management) (Li *et al.*, 2004; Parton *et al.*, 1998). However, models like DAYCENT or denitrification decomposition (DNDC) need to be calibrated for a specific region's local conditions (soils, crops, and climate) before such estimates can be accurately made (Del Grosso *et al.*, 2006).

In a recent 1 year simulation modeling exercise using the DNDC model for different counties, crops, and management regimes within California, it was estimated that California soils are sequestering C but there are large differences in the C dynamics by crop and region (Li *et al.*, 2004). The baseline scenario used a single year of climate data (1997) with nominal tillage practices, standard fertilizer application rates for each crop, full irrigation, and no manure amendments and no incorporation of above-ground litter over 50%. Li *et al.* (2004) determined that pastures were a significant C sink, while cotton, corn, tomatoes, citrus, deciduous fruit trees, and rice with winter flooding were a much smaller C sink. Lettuce and bean production were a net source of C, while viticulture practices had no net change either as a source or a sink (Li *et al.*, 2004). Baseline C sequestration for the 1997 climate baseline scenario, ranged from a release of -0.6 Tg C to potentially sequestering 6.1 Tg C, whereas baseline N_2O emissions ranged from 0.21 to 0.51 Tg N, over an acreage of 33,344 km^{-2} where most of the emissions were derived from corn, cotton, and vineyards (Li *et al.*, 2004). Modeling scenarios were carried out on 23 sites using alternative scenarios that changed only a single input data set, either climate (data from 1983) or management including (1) alternative litter incorporation amounts (50–90%), (2) alternative irrigation (e.g., over irrigating crops by 10% their needs), and (3) alternative manure management (application of 2000 kg^{-1} C ha^{-1}) for 23 sites. The model predicted that with the incorporation of 50–90% aboveground litter residue, C sequestration estimates ranged from 9.0 to 12.2 Tg C. If all the litter was removed, for instance in the case of crops grown for biofuels, then agricultural systems could become a large source of up to 7.8 Tg C. Manure application could potentially lead to 9.6–16.5 Tg C sequestered (Li *et al.*, 2004). However, the data from this study were inconclusive regarding the amount of N_2O released through these alternative practices and did not consider the effects of other management practices like CT, NT, and cover cropping needed to gain a better perspective of the effects of all conservation management options on C sequestration and GHG emissions. The interactions of other management and amendments with manure on GHG emissions also require further evaluation to better estimate the effects of manure.

De Gryze *et al.* (2009) calibrated and validated the biogeochemical DAYCENT model by using long-term ground-based measurements at four long-term agricultural experiment sites for both standard and alternative (including cover cropping, CT, and organic management) agricultural crop production systems to estimate GHG emissions (CO_2, N_2O, and CH_4) within California's Central Valley. The model predicted the largest mitigation potential of 4.58 Mg CO_2 equivalents ha^{-1} yr^{-1} when adopting organic management practices based on manuring and cover cropping, whereas solely winter cover cropping of legumes was predicted to mitigate between 1.16 and 2.71 Mg CO_2 equivalents ha^{-1} yr^{-1} (Table 2) (De Gryze *et al.*, 2009). Crops under CT, however, had the least mitigation potential of between 0.17 and 0.49 Mg CO_2 equivalents ha^{-1} yr^{-1} (De Gryze *et al.*, 2009). De Gryze *et al.* (2009) also adapted the DAYCENT model to examine the effect of conservation management practices on GHG emissions for six different crops (including alfalfa, corn, safflower, sunflower, wheat, and tomatoes) at the regional level. They calibrated DAYCENT to assess specific crop rotations, soil types, and climate patterns for Yolo County, CA. In this effort, it was determined that organic agriculture had the largest potential (3.6 CO_2 equivalents ha^{-1} yr^{-1}) for CO_2 mitigation

Table 2 Average differences in soil organic C (ΔSOC) (negative values indicate a decrease in soil C), N_2O, and CH_4 emissions, and overall global warming potential (GWP) (negative values indicate a net flux from the atmosphere to the soil) for each of the treatments of four validation sites in California

Site	Treatment	ΔSOC (kg C ha^{-1} yr^{-1})	N_2O–N (kg N ha^{-1} yr^{-1})	CH_4 (kg C ha^{-1} yr^{-1})	GWP (Mg CO_2-eq ha^{-1} yr^{-1})
LTRAS	Standard tillage	95	3.18	1.52	1.18
	Cover cropped and standard tillage	315	2.60	1.44	0.10
	Organic and standard tillage	1324	3.02	1.49	−3.40
	(standard error)	(64)	(0.14)	(0.04)	(0.28)
	(% caused by interannual differences)	(74%)	(37%)	(46%)	(72%)
	(standard error of the difference)	(41)	(0.11)	(0.03)	(0.16)
	Conservation tillage	47	3.01	1.51	1.27
	Cover cropped and conservation tillage	321	2.21	1.46	−0.10
	Organic and conservation tillage	1279	2.98	1.49	−3.26
	(standard error)	(94)	(0.18)	(0.05)	(0.43)
	(% caused by interannual differences)	(65%)	(53%)	(68%)	(61%)
	(standard error of the difference)	(71)	(0.17)	(0.04)	(0.37)
SAFS	Conventional 4-year rotation	407	2.2	1.6	−0.42
	Conventional 2-year rotation	436	1.5	1.4	−0.84

	Cover cropped	998	1.7	1.6	−2.82
	(standard error)	(77)	(0.09)	(0.02)	(0.29)
	(% caused by interannual differences)	(94%)	(80%)	(89%)	(96%)
	(standard error of the difference)	(21)	(0.04)	(0.01)	(0.08)
WSREC	Standard tillage	−90	4.0	2.0	2.25
	Standard tillage and cover cropped	677	4.0	1.9	−0.56
	Conservation tillage	−9	3.3	2.0	1.61
	Conservation tillage and cover cropped	729	3.8	1.9	−0.85
	(standard error)	(38)	(0.10)	(0.03)	(0.15)
	(% caused by interannual differences)	(91%)	(82%)	(38%)	(92%)
	(standard error of the difference)	(14)	(0.06)	(0.02)	(0.04)
Field 74	Standard tillage	128	2.6	1.5	0.79
	Conservation tillage	256	2.4	1.3	0.23
	(standard error)	(27)	(0.12)	(0.06)	(0.13)
	(% caused by interannual differences)	(51%)	(49%)	(19%)	(43%)
	(standard error of the difference)	(22)	(0.09)	(0.09)	(0.12)

Values are provided ± standard deviations.
CO_2-eq, CO_2 equivalents; LTRAS, Long-Term Research on Agricultural Systems; SAFS, Sustainable Agriculture Farming Systems; WSREC, West Side Research and Extension Center (De Gryze *et al.*, 2009).

(De Gryze *et al.*, 2009). The simulations predicted winter cover crops to have the second largest mitigation potential of 1.9 CO_2 equivalents ha^{-1} yr^{-1}. Low-input farming, a farming practice that reduces the use of off-site resources (e.g., synthetic fertilizers and pesticides) while optimizing on farm initial production resources and reduced tillage had the lowest mitigation potentials of all CT-type practices with 0.9 CO_2 equivalents ha^{-1} yr^{-1} and 1.3 CO_2 equivalents ha^{-1} yr^{-1}, respectively. Nevertheless, the low-input system could effectively reduce N_2O emission, which is a permanent mitigation strategy, and thus lower global warming potential (GWP) while providing an economically viable and permanent mitigation option (De Gryze *et al.*, 2009; Howitt *et al.*, 2009).

DAYCENT has proven to be fairly accurate in estimating GHG emissions in field crop systems but more accurate estimates of SOC stocks and the effect and adoption of CT management on GHG emissions requires further field monitoring and model validation. Furthermore, DAYCENT is not calibrated or validated for perennial cropping systems which occupy approximately 34% of the total agricultural land within California. Therefore, future research should focus on biogeochemical modeling of perennial crops because of their potential to sequester more C than annual row crops (Kroodsma and Field, 2006). Lastly, the current model estimates (De Gryze *et al.*, 2009) were based on results from small plots in researcher maintained field experiments, and it is still uncertain whether or not these findings can be scaled up to full scale agricultural operations.

5. Cobenefits of Alternative/Carbon-Friendly Agricultural Practices

It is well acknowledged that intensive tillage practices play a role in decreasing soil C (Lal, 2005; Lal *et al.*, 1998; Post and Kwon, 2000). The adoption of CT practices (including NT and direct seeding) may reduce SOC losses but current studies in California, described above, are inconclusive. Besides GHG mitigation, alternative conservation management practices may simultaneously have secondary or cobenefits associated with them (Table 3). Such cobenefits are important when considering GHG mitigation activities as they may substantially influence which methods should be adopted in a carbon trading market. It is, therefore, important to assess both the environmental and economic cobenefits of GHG mitigation options.

In conjunction with cover cropping, CT also improves soil water infiltration; reduces nitrate runoff, dust, and airborne particulate organic matter (PM-10) production, wind, and water erosion (Ashraf *et al.*, 1999; Cambardella and Elliott, 1994); improves habitat biodiversity (Bugg and Waddington, 1994); and improves the overall quality and fertility of soils

Table 3 Cobenefits of carbon-friendly agricultural practices

Carbon conservation farming practice	Environmental and economical cobenefits
Zero/conservation tillage	Dust control Reduced soil erosion Reduced fossil fuel usage and costs
Winter cover cropping	Reduced nitrogen leaching Increased SOM Increased aggregate stability Increased soil microbial activity and N turnover Increased overall N content of crop Increased water infiltration Potentially decreased instances of soil pathogens due C Increased habitat for beneficial insects
Organic residue and N fertilizer management	Potentially reduced need for synthetic fertilizers Reduced costs of excess synthetic fertilizers Reduced off-site fossil fuel usage and GHG emissions by reducing synthetic fertilizer production usage
Irrigation management	Potentially increased plant productivity and biomass, increasing soil C Potentially Reduced water costs

(Lal, 2004a,b). In California, studies indicate that use of CT alone does not substantially increase soil C, but when integrated with cover cropping, can led to sequestration of soil C (De Gryze *et al.*, 2009; Veenstra *et al.*, 2007). However, added economical cobenefits of CT are gained by the smaller number of tractor passes required which reduces fossil fuel usage and labor costs.

One of the environmental and human health problems associated with arid agricultural is the generation of dust and particulate matter from soil disturbance (Clausnitzer and Singer, 1996, 1997). Active farming operations such as tillage, planting, mowing, ploughing, and other land management methods contribute to the production of PM-10. Soil dust resulting from agricultural practices consists of both mineral (soil origin) and organic (plant origin) particles (Clausnitzer and Singer, 2000) and can affect atmospheric composition, soil texture, quality, and nutrient content. California is a major region of concern for dust related pollution (Nordstrom and Hotta, 2004), where much of the dust produced from soil disturbance is associated with the dry and desert environments in the Southern part of the State. In 2002 only 1 out of the 58 counties within California was in compliance for PM-10 regulations as set out by the California Air Resources Board (2004). Dust pollution is particularly high in the intensively managed agricultural region of the Central Valley (Clausnitzer and Singer, 1996, 2000).

PM-10 is primarily generated when tillage and ploughing operations are conducted while soil moisture is low (Nordstrom and Hotta, 2004), thus increased PM-10 production is often associated with lower soil water content and higher air temperature (Clausnitzer and Singer, 2000). Madden *et al.* (2009) corroborated the results of Clausnitzer and Singer (2000) and found in a 2-year study that PM-10 production within row crop agriculture in California's San Joaquin Valley is greater when soil moisture was low and with increased disking operations. In another study conducted within California's San Joaquin Valley, Clausnitzer and Singer (1997) found that 87% of respirable dust (RD) came from the preparation of agricultural lands (e.g., deep ploughing and tilling) and that approximately 18% of the total RD came from harvesting and cultivation of crops, of which equates to a total of 33% of the total farming operations. Various conservation methods have been actively researched for the reduction of dust and PM-10 production. CT protects the soil through enhanced water retention, potentially increasing SOM content, and stabilizing soil aggregates. It also reduces dust by limiting the amount of tractor passes made over cultivated fields. Madden *et al.* (2008) observed in a 2-year study between ST and CT systems in dairy forage production in the San Joaquin Valley, that CT reduced PM-10 emissions by approximately 85% compared to ST, mainly due to the fewer tractor passes required. Baker *et al.* (2005) measured the quantity and composition of dust produced in the San Joaquin Valley in a cotton–tomato rotation and compared standard tillage with cover crop (STCC) and with no cover crop (STNO), conservation tillage with cover crop (CTCC), and conservation tillage with no cover crop (CTNO). The quantity of dust produced from the CTNO treatment throughout a 2-year crop rotation was approximately one third of the dust generated from the STNO treatment (Baker *et al.*, 2005). There was little difference between the STNO and STCC treatments; however, dust concentrations were twice as high in the CTCC than CTNO treatment. Furthermore, Clausnitzer and Singer (1996) found a 20% increase in dust from a cover cropped field compared to no cover cropped fields. Some of this difference may be associated with the time it takes for the winter cover crop to decompose which in turn could delay harvesting until the drier season, thus increasing the emissions of dust because of higher air temperatures and lower soil water holding capacity (Clausnitzer and Singer, 1996). Further research and longer term studies are necessary to determine the exact benefits of CT and cover cropping practices upon dust emissions versus C sequestration benefits.

Within the top surface layer of agricultural soil (~0–15 cm depth), organic C and nitrogen stocks, microbial abundance, biomass, and activity are highest (Janzen *et al.*, 1992). The readily reactive labile fraction of SOM functions as a transitory pool for nutrients and plays a significant role in N dynamics within the soil. However, when soils are under cultivation and disturbed by ST practices, this labile SOM pool declines (Cambardella and Elliott, 1994). NO_3^- leaching can increase and thereby diminish water quality of nearby watersheds, especially when crops are fallowed and left bare during the winter months (Jackson *et al.*, 1993; Janzen *et al.*, 1992). Postharvest NO_3–N leaching is likely to occur when excess N fertilizer is applied (Weinert *et al.*, 2002). The mitigation of NO_3^- run off and the improvement of water quality are in general cobenefits of agricultural conservation practices leading to GHG mitigation. The planting of a cover crop during winter fallow periods enables the cover crop to serve as an NO_3–N catching crop that will store and absorb NO_3–N within the winter months and, not only facilitates in the reduction of NO_3–N leaching (Drinkwater *et al.*, 1998; Torstensson *et al.*, 2006), but improves surface layer soil N dynamics (Shipley *et al.*, 1992).

Within California, intensive vegetable crop rotations with a winter fallow period are highly vulnerable to NO_3–N losses from leaching and via denitrification processes (Jackson *et al.*, 1993; Ryden and Lund, 1980; Wyland *et al.*, 1996). Furthermore, surface SOM levels in winter fallowed crops from this region have declined as a result of little residue being returned to the soil after harvest (Jackson *et al.*, 1993). Previous research has found that overwinter leaching of N can be decreased by planting a nonleguminous cover crop (Jackson *et al.*, 1993; Poudel *et al.*, 2002), especially from the root zone (Meisinger *et al.*, 1991), and SOM levels can be increased along with other associated cobenefits such as weed suppression and improvement of soil quality related to the use of winter cover cropping. Wyland *et al.* (1996) observed the N balance of the soil profile within a broccoli cropping system in the California Central Valley was influenced by the use of a winter cover crop. Leaching of NO_3–N from residual N fertilizer and overall N mineralization was less within the cover cropped treatment due to the cover crop removing excess N and water from within the soil surface layer (Wyland *et al.*, 1996). By stabilizing N in SOM, cover crops provide an environmental cobenefit that reduces fertilizer and losses of soil N sources to surface and groundwater supplies.

Cover crops can also provide subsequent cash crops with available N that can prospectively lead to reduction of synthetic N fertilizers (Griffin *et al.*, 2000). Incorporation of a winter cover crop resulted in 20–55% of cover crop-N being available to the following summer grown cash crop (Kramer *et al.*, 2002; Malpassi *et al.*, 2000; Sims and Slinkard, 1991), thus reducing the amount of synthetic N fertilizer to be applied. Poudel *et al.* (2001) evaluated the effects of organic (no synthetic N fertilizers added), low input (reduced amounts of synthetic N fertilizers added), and standard conventional (regular recommended additions of synthetic N fertilizers) farming systems on the N balance, storage, and losses in irrigated row crops within the California Central Valley region. They observed a greater cumulative N balance within the organic and low input than conventional system because of their greater capacity to store surplus N in the soil.

Another major benefit to farmers is that growing a cover crop in rotation with a cash crop can reduce pesticide and herbicide use (Pimentel and Pimentel, 1996; Snapp *et al.*, 2005). Cover crops can suppress weed growth and provide a substantial habitat for beneficial arthropods and pests (Lal, 1991), which compete with or prey upon plant-parasitic pests detrimental to cash crops. For instance, the incorporation of green manures from cover crops reduced the incidence and abundance of many crop specific plant pathogens and pests depending upon correct cover crop selection and management (Bugg, 1991).

As mentioned above, weed suppression is also a potential cobenefit of planting cover crops. The use of cover crops in conjunction with CT practices will likely minimize but not completely eliminate the use of herbicides (Teasdale, 1996). The impact of a cover crop upon soil and foliar pests depends not only upon the arthropod population, cover crop, and main crop ecosystem, but also upon the management and type of cover crop chosen. Future research is needed to assess the effects of cover crop diversity as it relates to the environmental aspects of sustainable and alternative management practices (Lal, 1991).

5.1. Issues related to conservation management practices

Although the benefits of adopting CT and cover cropping practices are wide and varied, there are some potentially negative environmental and economic issues (Table 4). One such issue is CT may require more energy than standard conventional practice to (1) control weeds and weed seedlings with herbicides and special weeding equipment, and (2) reduce higher incidence of pests without the use of excessive pesticides (Lal *et al.*, 1990, 1998). Also within NT production practices, surface application of fertilizer N may occur and lead to losses through volatilization and runoff (Uri and Konyar, 1996). However, the actual degree to how these instances determine whether energy is saved or expended relies upon the actual extent of weed and pest infestations and the degree to which they are required to be controlled.

Further limitations associated with understanding CT practices and their interactions with SOM dynamics are associated with the actual measurement and detection of SOC quantity changes. Due to the size of the original soil C pool, detecting any changes is highly difficult because often these changes are small and only occur over a significant amount of time (Lal *et al.*,

Table 4 Environmental and economical issues associated with carbon sequestering farming practices

Management practice	Environmental issues	Economical issues
Zero/conservation tillage	Soil compaction Can effect emissions of non-CO_2 GHGs	Buying new or modifying equipment
Cover cropping and residue management	Weeds/management and suppression Provide an organic food base for pathogen growth Disruption to spring planting schedule if cover crop decomposition is too slow to incorporate residues If C:N ratio too high, reduced N availability Can inhibit the use of postemergent herbicides causing weed reemergence Can increase water requirements for cover crop growth	Lower plant productivity under cover crop Losses of plant yields due to pests, weeds, or disease caused by the cover crop Can increase the costs of herbicides and pesticides Can increases water requirements for cover crop growth

1998; Six *et al.*, 2004). Furthermore, in studies that show little or no changes in SOC with CT systems, it should be taken into account, especially in future studies, that soil C within CT systems may actually be redistributed rather than lost (Veenstra *et al.*, 2007).

Although the conversion to CT management is advocated as being a management tool which can be used to reduce atmospheric CO_2 emissions and store C within soils (Lal *et al.*, 1998; West and Post, 2002), when adopting CT and other sustainable agricultural C sequestration type practices, the effect that such practices have upon the other two main GHG (N_2O and CH_4), both of which have a greater GWP than CO_2 must always be considered (Robertson *et al.*, 2000; Six *et al.*, 2004). Depending upon the magnitude to which non-CO_2 GHG emissions are released from CT practices compared to the amount of C stored, the benefits of CT may be outweighed by the disadvantages from other GHG emissions. Unfortunately, there is very little data available on the effects of alternative practices upon non-CO_2 GHG emissions in California. Even fewer studies exist to examine the long-term effects of interactions between management practices on non-CO_2 GHG emissions. The importance of these interactions is illustrated in a recent study in Minnesota by Venterea *et al.* (2005). They demonstrated that emissions of N_2O varied in degree under different tillage intensity and type of nitrogen fertilizer application. They observed that emissions of N_2O represented a large proportion of the GHG budget for reduced tillage systems and that the importance between the interaction of both tillage and fertilizer management is necessary when controlling and reducing non-CO_2 emissions (Venterea *et al.*, 2005). A comprehensive review by Six *et al.* (2004) found that newly converted NT systems increased GHG emissions, subsequently increasing the GWP of the system. They also concluded that only long-term adoption over many years would reduce emissions and GWP but the prediction had a great degree of uncertainty (Six *et al.*, 2004). Again, with this review it was observed that N_2O emission was the principal GHG to raise GWP, therefore, it is suggested that nutrient management be made more efficient in order to reap the benefits from C sequestration within NT systems.

One further dilemma in balancing non-CO_2 GHGs with C sequestration is related to the high spatial variability of emissions and their complex interactions between physical, chemical, and biological soil properties. In an agricultural field study within California, Lee *et al.* (2006) observed high spatial variability in GHG emissions from ST and NT treatments at the field level. They established the fact that emissions showed little or no difference between ST and NT, however, they did observe a tillage–irrigation interaction which increased non-CO_2 GHG emissions (Lee *et al.*, 2006).

The benefits of incorporating cover crops into a crop rotation has long been recognized; however, cover cropping has a few constraints and disadvantages which have lead to the slow adoption of the practice within California (Mitchell *et al.*, 2007). These issues arise primarily in response to a lack of knowledge, incorrect choice of cover crop, and conflicting timing that interferes with the establishment of summer crops. Furthermore, the economic costs of planting and terminating a cover crop concerns some growers.

Cover crops are often used to suppress weeds and reduce the use of herbicides, however, if improperly managed cover crops themselves can become weeds and out-compete crops for light, nutrients, and water or inhibit herbicides by covering the soil surface (Bugg and Waddington, 1994; Lal *et al.*, 1991). For example, in the Fresno Valley of California, mulches of cover crops are currently undergoing extensive research for their decomposability within tomato crops. A cover crop is usually planted after harvest and left to grow over the winter, then terminated once the tomato crop is planted. The mulch from the cover crop remains on the soil surface and can potentially interfere with postemergence herbicides and results in the escape of weeds (Shrestha *et al.*, 2007). Herrero *et al.* (2001b) also noted that the in-season weed control by the surface cover crop was not sufficient enough alone to control weeds. As well as being a probable weed and because cover crops can potentially provide habitats for beneficial organisms they can also provide a habitat to pest species (Bugg and Waddington, 1994; Luna, 1998). Finally, the residues of the cover crop interfere with the mechanical harvesting of the tomatoes.

One more important concern related to cover crops is the uncertainty of their water requirements. In California, where water supplies can be constrained, additional water needs by a cover crop will result in that practice becoming less economically or environmentally viable. In a 3-year study by Mitchell *et al.* (1999), growing cover crops such as barley and vetch had a negligible impact on winter soil water storage. However, preirrigation needs for ensuing crops may be affected by soil water storage changes from use of cover crops if they are allowed to excessively grow and deplete the soil moisture profile (Mitchell *et al.*, 1999). Therefore, successful integration of cover crops into agricultural production systems in arid and semiarid climates of regions of California will only be feasible with careful consideration of water balance issues. In higher rainfall regions of northern California, planting a winter cover crop can help to improve rain infiltration and enhance soil water storage. Joyce *et al.* (2002) observed in the Sacramento Valley that cover cropping improved soil water storage for succeeding conventional and organic crops.

One of the greatest barriers to replacing winter fallow with a winter cover crop mainly lies in the potential for the cover crop to disrupt spring planting. The time after growing and incorporating cover crop before the proceeding crop is extremely small. Therefore, in order to provide any of the long-term benefits of winter cover cropping, all cover cropping practices must be compatible both with the planting schedules of commodity crops and their management (Mitchell *et al.*, 1999).

5.2. Economics of conservation farming

Economic agricultural productivity must be a major consideration when converting to conservation management practices. Due to limited data and lack of adoption of CT-type practices within California, calculating their economic costs is a difficult process. Confounding this calculation is the lack of precise economic values and performance metrics for crop type and yields, management, and region specific factors. Of the few studies available, Mitchell *et al.* (2003) conducted an economical study on a cotton–tomato rotation under CT management both with and without cover crops. They observed that cotton production profitability was reduced in the CT compared to ST system, whereas for tomatoes, the profitability and productivity was significantly increased in the CT system.

Wyland *et al.* (1996) calculated that approximately 14% of the total operating costs to grow and incorporate cover crops prior to broccoli were attributable to costs that would have been incurred should the soil have been left as fallow during winter. Overall they determined the complete cost of the cover crop to be approximately 5% of the total cost of producing one broccoli crop (Wyland *et al.*, 1996). A large proportion of the cover crop expenditure was obtained from incorporation of the cover crop which required increased major machinery usage, labor, and repairs required for CT practices compared to ST. Another single significant expense was irrigation water needed to germinate the cover crop, accounting for almost half of the total operating costs to grow, maintain, and incorporate the cover crop (Wyland *et al.*, 1996). Despite these costs, Wyland *et al.* (1996) determined that the cost of cover crops to be a negligible proportion of the actual costs to produce broccoli with ST practices. Furthermore, in an 2-year economic analysis of three lettuce crops in the Salinas Valley with minimal tillage (MT) (with or without cover crops) or ST (with or without cover crops), showed that the net returns for either tillage system did not increase but in the MT system cost savings did increase either with or without organic matter additions from the cover crop as compared to the ST system (Jackson *et al.*, 2004).

Nevertheless, costs associated with conversion practices can occur from modification of tillage equipment as well as from other operational changes (Mitchell *et al.*, 2006) and additional soil preparations needed for cover crops, including potential higher expenditure for pesticides and herbicides (Uri and Konyar, 1996). These costs can be partially offset by fossil fuel reduction as a result of fewer tractor passes. Within a future carbon emission trading market, these initial costs can be reaped by trading carbon credits for the carbon potentially stored or offset from CT practices (Howitt *et al.*, 2009). A 2005 survey of Yolo County farmers practicing conservation or conventional practices for six different crops (wheat, corn, rice, safflower, tomatoes, and sunflower) revealed that compared to conventional farming, applying conservational farming techniques resulted in higher returns per hectare (Howitt *et al.*, 2009). However, profits among farmers using organic practices were much more unpredictable compared to conventional growers and this fact was not easily shown in the results of the survey (Howitt *et al.*, 2009). Using the biogeochemical model DAYCENT for the Yolo County survey (De Gryze *et al.*, 2009), in conjunction with economic models for various CT and organic management practices, a C sequestration supply curve was developed. The C sequestration supply curve showed that by adopting CT practices in response to carbon payments between \$3 and \$8 ton^{-1} C yr^{-1}, Yolo County farmers could sequester as much as 39,000 tons of carbon, which equates to approximately 3% of the total carbon release from Yolo (Howitt *et al.*, 2009). Ultimately though, it will come down to the consumer and farmer who must find sufficient value in CT practices and approaches in order to provide the cost equilibrium or profit margin to sustain their adoption (Mitchell *et al.*, 2008).

6. Research Challenges and Recommendations

California is an economically dynamic region as a result of varying microclimates, vast agriculture resources, and its vulnerability to water shortages. The future of California agriculture is hanging in the balance due to the potential effects of climate change on its cropping systems and water resources. Further information is needed regarding GHG mitigation under irrigated agriculture within ecologically vulnerable areas such as California (Bronick and Lal, 2005; Follett, 2001; Lal, 2004a,b). Though, there is a growing knowledge of soil C dynamics and GHG emissions, quantitative assessments of GHG emissions from many cropping systems, and management systems are still needed (Follett, 2001), especially for the semiarid Mediterranean climate and irrigated agricultural systems of California. An effective interdisciplinarily approach is needed to address environmental, biogeochemical, legislative, and socioeconomical research priorities. Table 5 details current research needs to address the effectiveness and processes associated with GHG mitigation within California agriculture.

A major question that must be addressed to enable future predictions of soil C losses and additions is to understand the magnitude of historic losses of soil C that occurred during the conversion of natural lands to the cultivated managed lands of today. These historical losses of soil C during conversion have been reported for many regions of the United States, particularly within the Midwest (Lal, 2004a,b; West and Marland, 2002), but little is known about the semiarid region of California compared to the rest of the nation (De Clerck *et al.*, 2003).

There is also a need for complimentary ground-based monitoring to further quantify and measure soil C stocks, changes in soil C stocks under different management practices, baseline GHG emissions, and GHG emissions occurring under a variety of management practices. The accurate and precise measurement of C sequestration and GHG emissions through a field-based monitoring program will be useful for three purposes. The first is to calibrate and validate biogeochemical models, such as DNDC and DAYCENT. The second is to help in identifying crucial knowledge gaps to quantify future mitigation measures and management options. The third relates to the development of policies, regulations, and emissions caps and other types of government lead incentives to promote the reduction of GHG emissions as an attractive option for agriculture.

Table 5 Interdisciplinary research challenges and recommendations for investigating further carbon sequestration, GHG emission reductions, and agricultural sustainability within California agriculture

Economic research areas	Sociological research areas	Environmental (including monitoring) research areas	Soil biogeochemical and mechanism research areas
– Evaluation of cobenefit values – Evaluating the economic cost of C sequestration under different management regimes/land-use/crop and soil type – Determine an effective C trading market based on all benefits of using conservational management practices – Economy of using CT practices upon crop productivity and yield in a variety of crops present in California	– Establishment of priority awareness and human barriers – Ascertain legislative needs in relation to a C cap and trade market and determine if the agricultural sector needs to be regulated or involved on a voluntary basis – Identify policies that will lead to the adoption of BMP[a]	– Accurate and precise measurement of C sequestration – Ground-based measurements of GHG under different management regimes/land-use/crop and soil type used to validate biogeochemical models – Identify which type of crop has the highest potential to sequester C and reduce GHG emissions. Identify the total sink capacity for various crops. – Ascertain BMP related to the correct use of fertilizers and nutrient/residual amendments to maximize effectiveness and minimize off-site impacts (e.g., N leaching to watersheds) – Further adjustments of biogeochemical models to California baselines and conditions	– Assess aggregate formation relationship to crop and soil type and management practices in order to assess BMP – Find out what mechanistic processes control C and nutrient dynamics within California agricultural soils – Discover the actual rates of C sequestration under various land-use practices and then recommend the use of BMP

[a] BMP is best management practices.

The GHG emissions and C sequestration estimates generated from biogeochemical models can be used to predict responses of soil C and GHG emissions to management changes at the field, regional, and global scale level (De Gryze *et al.*, 2009; Paustian *et al.*, 1997), and such data can be used to create GHG inventories for carbon trading markets that may be exploited by the agricultural sector in the future, for instance as in the provisions of California ''Climate Change Solution Act, AB 32''. However, whereas process-based models such as DNDC and DAYCENT can contribute to estimates of GHG emissions and C stock values, the models have some performance limitations which generate uncertainties, especially if the model is not calibrated for a specific region, crop type, climate, and other environmental conditions. Furthermore, the critical inputs required for modeling soil C dynamics and GHG emissions from both standard and conservation practices used in the various agricultural systems of California must build upon an accurate soil C and GHG emission database generated through a continued field monitoring program.

The GHG emissions and soil C stock inventories generated by computational models and field-based measurements can be used to directly participate in a carbon trading market system, by integrating them with economic assessments to provide the economic value of reducing GHG emissions. The Kyoto Protocol (1997) formalized the concept of carbon credits as a result of the increasing concern and global response to climate change issues. However, in order to trade or purchase carbon credits, there is a need to validate GHG emissions and C stocks and understand the carbon budget at the farm and regional level.

Another potential sociological research limitation concerning C sequestration and adoption of sustainable management practices is understanding and overcoming human barriers, for example, where farmers are not convinced by the benefits of CT practices due to economic and production-related constraints associated with implementation. Sociopolitical research is needed to find ways to better understand and address the concerns of farmers, and other stakeholders, in the formulation of practices and policies to reduce GHGs and increase C sequestration and agricultural sustainability.

From a basic research standpoint, we need to learn more about the role of soil structure, microbial community composition and function, soil food web dynamics, and interactions among these components, in stabilizing SOC and their cobenefits for the agricultural soils of California. It is well known that soil structural stability is fundamental in the protection of SOC (Blanco-Canqui and Lal, 2004; Six *et al.*, 2002; Tisdall and Oades, 1982 and references therein), and that it is linked with root dynamics, faunal activity, and soil microbial activity and abundance, but the effect of agricultural management on the interactions and links between these components is still poorly understood. Consequently, further research on the effects of different management practices on these interactions and linkages is needed to elucidate and assess management effects on C pool residence times and GHG mitigation within a range of soil and crop types within semiarid irrigated agriculture. We also need better information about the effects of multiple OM source inputs and differences in quality of OM source inputs, and their interactions with other edaphic factors, on increasing SOC and soil fertility.

In relation to on-farm amendment of cover crops or other organic manures and wastes, exploration of nutrient use efficiency and cautious use of N fertilizers is needed to ensure that the benefits of C sequestration are not outweighed by any potential drawbacks of off-site nutrient leaching and pollution of water sources.

7. Conclusion: The Potential for California Agriculture to Sequester Carbon

It is well documented that residue management along with CT or MT, cover cropping and N fertilization management can provide opportunities to increase C and N sequestration and improve soil quality and enhance productivity within the Midwestern United States. The information reviewed within this paper demonstrates there is also potential for C sequestration and GHG reduction within the Mediterranean climate of California under CT-type management practices.

The data presented within this paper also indicates that no single land management change or practice could mitigate all of the C released from agricultural practices (e.g., from fossil fuel usage, land,use changes, soil erosion, biomass burning, and nitrogen fertilizer production) and meet the climate change commitments as set out in the directive AB 32. Although studies identify several sustainable management practices able to increase SOC and reduce GHG emissions within the agricultural soils of California, including the use of cover crops and organic amendment additions, there are still areas of uncertainty and conflicting results which show that these conservation techniques do not always sequester C and/or reduce GHG emissions efficiently. However, there is no question that changing to conservational type practices will reduce use of fossil fuels because of the decrease in numbers of tractor passes and reduction in synthetic N fertilizer use; thus reduction in GHG emissions will be ensured as long as such alternative practices are continued.

Despite the numerous studies presented within this paper, there is still a need for more data and improved understanding of the long-term (>10 years) effects of cropping systems, tillage practices, N fertilization and residue management upon soil C sequestration in the complex, and diverse agriculture systems of California. Integrated management, along with future integrated scientific and socioeconomic research, is key to developing a more stable soil resource base for California. The future of conservation-type agricultural practices within California will depend, in part, on emerging technologies which will further reduce the environmental risk of agricultural practices upon soils. Furthermore, the cobenefits associated with conservation-type management practices are highly valuable in their own right and necessary to ensure the future sustainability of California agriculture.

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REFERENCES

Ashraf, M., Pearson, C. H., Westfall, D. G., and Sharp, R. (1999). Effect of conservation tillage on crop yields, soil erosion, and soil properties under furrow irrigation in western Colorado. *Am. J. Altern. Agric.* **14,** 85–92.

Auburn, J. S., and Bugg, R. (1991). An information data base on cover crops. *In* "Cover Crops for Clean Water" (W. L. Hargrove, Ed.). Soil and Water Conservation Society, Ankeny, IA, pp. 11–15.

Baker, J. B., Southard, R. J., and Mitchell, J. P. (2005). Agricultural dust production in standard and conservation tillage systems in the San Joaquin Valley. *J. Environ. Qual.* **34,** 1260–1269.

Beare, M. H., Hendrix, P. F., and Coleman, D. C. (1994). Water stable aggregates and organic matter fractions in conventional and no-tillage soils. *Soil. Sci. Soc. Am. J.* **58,** 777–786.

Bemis, G., and Allen, J. (2005). Inventory of California Greenhouse Gas Emissions and Sinks: 1990 to 2002 Update. California Energy Commission (CEC) Publication, Sacramento, CA (CEC-600-2005-025).

Blake, P. (1991). Measuring cover crop soil moisture competition in north coastal California vineyards. *In* "Cover Crops for Clean Water" (W. L. Hargrove, Ed.). Soil and Water Conservation Society, Ankeny, IA, pp. 39–41.

Blanchart, E., Lavelle, P., Braudeau, E., Le Bissonnais, Y., and Valentin, C. (1997). Regulation of soil structure by geophagous earthworm activities in humid savannas of Cote D'ivoire. *Soil Biol. Biochem.* **29,** 431–439.

Blanchart, E., Albrecht, A., Alegre, J., Duboisset, A., Gilot, C., Pashanasi, B., Lavelle, P., and Brussaard, L. (1999). Effects of earthworms on soil structure and physical properties. *In* "Earthworm Management in Tropical Agroecosystems" (P. Lavelle, L. Brussaard, and P. Hendrix, Eds.), pp. 149–171. CAB International, Oxon.

Blanco-Canqui, H., and Lal, R. (2004). Mechanisms of carbon sequestration in soil aggregates. *Crit. Rev. Plant Sci.* **23**(6), 481–504.

Blevins, R. L., Langdale, G. W., and Frye, W. W. (1998). Conservation tillage for erosion control and soil quality. *In* "Advances in Soil and Water Conservation" (F. J. Pierce and W. W. Frye, Eds.), pp. 51–68. Sleeping Bear Press, Ann Arbor, MI.

Bronick, C. J., and Lal, R. (2005). Soil structure and management: A review. *Geoderma* **124**(1–2), 3–22.

Bruce, R. R., Hendrix, P. F., and Langdale, G. W. (1991). Role of cover crops in recovery and maintenance of soil productivity. *In* "Cover Crops for Clean Water" (W. L. Hargrove, Ed.), pp. 109–120. Soil and Water Conservation Society, Ankeny, IA.

Bruinsma, J. (2003). World agriculture: Towards 2015/2030, an FAO perspective. Earthscan Publishers, London.

Bugg, R. L. (1991). Cover crops and control of arthropod pests of agriculture. *In* "Cover Crops for Clean Water" (W. L. Hargrove Ed.), Proceedings of an international conference, West Tennessee, Experiment Station, Jackson, April 9–11, pp. 157–163. Soil and Water Conservation Society, Ankeny, IA.

Bugg, R. L., and Waddington, C. (1994). Using cover crops to manage arthropod pests of orchards: A review. *Agric. Ecosyst. Environ.* **50,** 11–28.

Bugg, R. L., Horwath, W., Six, J., and Van Horn, M. (2007). Practical soil ecology. *In* "Cover Crops for Vegetable Farming Systems" (R. Smith, M. L. Gaskell, R. L. Bugg, O. Daugovish, and M. Van Horn, Eds.), University of California Agriculture and Natural Resources 8000 Series.

Buschiazzo, D. E., Panigatti, J. L., and Unger, P. W. (1998). Tillage effects on soil properties and crop production in the subhumid and semiarid Argentinean Pampas. *Soil Tillage Res.* **49**(1–2), 107–116.

Calderon, F. J., and Jackson, L. E. (2002). Roto-tillage, disking and subsequent irrigation: Effects on soil nitrogen dynamics, microbial biomass, and CO_2 efflux. *J. Environ. Qual.* **31,** 752–758.

Calderon, F. J., Jackson, L. E., Scow, K. M., and Rolston, D. E. (2000). Microbial responses to simulated tillage in cultivated and uncultivated soils. *Soil Biol. Biochem.* **32,** 1547–1559.

Cambardella, C. A., and Elliott, E. T. (1994). Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. *Soil Sci. Soc. Am. J.* **58,** 123–130.

Carlisle, E. A., Steenwerth, K. L., and Smart, D. R. (2006). Effects of land use on soil respiration: Conversion of oak woodlands to vineyards. *J. Environ. Qual.* **35,** 1396–1404.

Cayan, D., Maurer, E., Dettinger, M., Tyree, M., Hayhoe, K., Bonfils, C., Duffy, P., and Santer, B. (2006). Climate scenarios for California, www.climatechange.ca.gov/.

California Farm Bureau Federation (CFBF). (2002). News division, http://www.cfbf.com.

Clausnitzer, H., and Singer, M. J. (1996). Respirable-dust production from agricultural operations in the Sacramento Valley, California. *J. Environ. Qual.* **25,** 877–884.

Clausnitzer, H., and Singer, M. J. (1997). Intensive land preparation emits respirable dust. *Calif. Agric.* **51**(2), 27–30.

Clausnitzer, H., and Singer, M. J. (2000). Environmental influences on respirable dust production from agricultural operations in California. *Atmos. Environ.* **34,** 1739–1745.

Cole, C. V., Flach, K., Lee, J., Sauerbeck, D., and Stewart, B. (1993). Agricultural sources and sinks of carbon. *Water Air Soil Pollut.* **70**(1–4), 111–122.

De Clerck, F., Singer, M. J., and Lindert, P. (2003). A 60-year history of California soil quality using paired samples. *Geoderma* **114,** 215–230.

De Gryze, S., Albarracin, M. V., Catalá-Luque, Rosa, Howitt, R. E., and Six, J. (2009). Modeling Shows that Alternative Soil Management Can Decrease Greenhouse Gases. California Agriculture, (April–June), pp. 84–90.

Del Grosso, S. J., Parton, W. J., Mosier, A. R., Walsh, M. K., Ojima, D. S., and Thornton, P. E. (2006). DAYCENT national-scale simulations of nitrous oxide emissions from cropped soils in the United States. *J. Environ. Qual.* **35,** 1451–1460.

Drinkwater, L. E., Wagoner, P., and Sarrantonio, M. (1998). Legume-based cropping systems have reduced carbon and nitrogen losses. *Nature* **396,** 262–265.

Elliott, E. T. (1986). Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* **50,** 627–633.

Fennimore, S. A., and Jackson, L. E. (2003). Organic amendment and tillage effects on vegetable field weed emergence and seedbanks. *Weed Tech.* **17,** 42–50.

Follett, R. F. (2001). Soil management concepts and carbon sequestration in cropland soils. *Soil Tillage Res.* **61,** 77–92.

Follett, R. F., and McConkey, B. (2000). The role of cropland agriculture for C sequestration in the Great Plains. *Proc. Great Plains Soil Fertil. Conf.* **8,** 1–15.

Fonte, S. J., Kong, A. Y. Y., van Kessel, C., Hendrix, P. F., and Six, J. (2007). Influence of earthworm activity on aggregate-associated carbon and nitrogen dynamics differs with agroecosystem management. *Soil Biol. Biochem.* **39,** 1014–1022.

Griffin, T., Liebman, M., and Jemison, J. Jr. (2000). Cover crops for sweet corn production in a short-season environment. *Agron. J.* **92,** 144–151.

Haas, H. J., Evans, C. J., and Miles, E. F. (1957). Nitrogen and Carbon Changes in Great Plains Soils as Influenced by Cropping and Soil Treatments. State Agricultural Experiment Stations, (Technical Bulletin No 1164 USDA).

Hartwig, N. L., and Ammon, H. U. (2002). Cover crops and living mulches. *Weed Sci.* **50,** 688–699.

Hartz, T. K., Mitchell, J. P., and Giannini, C. (2000). Nitrogen and carbon mineralization dynamics of manures and composts. *Hort. Science* **35**(2), 209–212.

Hayhoe, K., Cayan, D., Field, C. B., Frumhoff, P., Maurer, E. P., Miller, N. L., Moser, S. C., Schneider, S. H., Cahill, K. N., Cleland, E. E., Dale, L., Drapek, R., *et al.* (2004). Emissions pathways, climate change, and impacts on California. *Proc. Natl. Acad. Sci. USA* **101,** 12422–12427.

Hernanz, J. L., López, R., Navarrete, L., and Sánchez-Girón, V. (2002). Long-term effects of tillage systems and rotations on soil structural stability and organic carbon stratification in semiarid central Spain. *Soil Tillage Res.* **66**(2), 129–141.

Hernanz, J. L., Sánchez-Girón, V., and Navarrete, L. (2009). Soil carbon sequestration and stratification in a cereal/leguminous crop rotation with three tillage systems in semiarid conditions. *Agric. Ecosyst. Environ.* **133**(1–2), 114–122.

Herrero, E. V., Mitchell, J. P., Lanini, W. T., Temple, S. R., Miyao, E. M., Morse, R. D., and Campiglia, E. (2001). Use of cover crop mulches in a no-till furrow-irrigated processing tomato production system. *Horttechnology* **11,** 43–48.

Horwath, W. R., Devêvre, O. C., Doane, T. A., Kramer, T. W., and van Kessel, C. (2002). Soil carbon sequestration management effects on nitrogen cycling and availability. *In* ''Agricultural Practices and Policies for Carbon Sequestration in Soil'' (J. M. Kimble, R. Lal, and R. F. Follett, Eds.), pp. 155–164. Lewis, Boca Raton, FL.

Howarth, W. R., Mitchell, J. P., and Six, J. (2006). Tillage and crop management effects on air, water and soil quality in California. Conservation Tillage Workgroup, ANR publication, p.1–8.

Howitt, R. E., Català-Luque, R., De Gryze, S., Wicks, S., and Six, J. (2009). Realistic Payments Could Encourage Farmers to Adopt Practices that Sequester Carbon California Agriculture, (April–June), pp. 91–95.

Intergovernmental Panel on Climate Change (IPCC) (2006). S. Eggleston, L. Buendia, K. Miwa, K. Tanabe (Eds.). Guidelines for national greenhouse gas inventories. IGES, Cambridge University Press, Cambridge, United Kingdom.

Intergovernmental Panel on Climate Change (IPCC) (2007). B. Metz, O. R. Davidson, P. R. Bosch, R. Dave, L. A. Meyer (Eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jackson, L. E., Wyland, L. J., and Stivers, L. J. (1993). Winter cover crops to minimize nitrate losses during a short fallow period. *J. Agric. Sci.* **121,** 55–62.

Jackson, L. E., Calderon, F. J., Steenwerth, K. L., Scow, K. M., and Rolston, D. E. (2003). Responses of soil microbial processes and community structure to tillage events and implications for soil quality. *Geoderma* **114**(3–4), 305–317.

Jackson, L. E., Ramirez, I., Yokota, R., Fennimore, S. A., Koike, S. T., Henderson, D., Chaney, W. E., Calderón, F. J., and Klonsky, K. (2004). On-farm assessment of organic matter and tillage management on vegetable yield, soil, weeds, pests, and economics in California. *Agric. Ecosyst. Environ.* **103,** 443–463.

Janzen, H. H., Campbell, C. A., Brandt, S. A., Lafond, G. P., and Townley-Smith, L. (1992). Light-fraction organic matter in soils from long-term crop rotations. *Soil Sci. Soc. Am. J.* **56,** 1799–1806.

Jawson, M. D., Shafer, S. R., Franzluebbers, A. J., Parkin, T. B., and Follett, R. F. (2005). GRACEnet: Greenhouse gas reduction through agricultural carbon enhancement network. *Soil Tillage Res.* **83,** 167–172.

Joyce, B. A., Wallender, W. W., Mitchell, J. P., Huyck, L. M., Temple, S. R., Brostrom, P. N., and Hsiao, T. C. (2002). Infiltration and soil water storage under winter cover cropping in California's Sacramento Valley. *Am. Soc. Agric. Eng.* **45**(2), 315–326.

Kern, J. S. (1994). Spatial patterns of soil organic carbon in the contiguous United States. *Soil Sci. Soc. Am. J.* **58,** 439–455.

Kern, J. S., and Johnson, M. G. (1993). Conservation tillage impacts on national soil and atmospheric carbon levels. *Soil Sci. Soc. Am. J.* **57,** 200–210.

Kerr, R. A. (2005). How hot will the greenhouse world be? *Science* **309,** 100.

Ketterings, Q. M., Blair, J. M., and Marinissen, J. C. Y. (1997). Effects of earthworms on soil aggregate stability and carbon and nitrogen storage in a legume cover crop agroecosystem. *Soil Biol. Biochem.* **29,** 401–408.

Khan, S. A., Mulvaney, R. L., Ellsworth, T. R., and Boast, C. W. (2007). The myth of nitrogen fertilization for soil carbon sequestration. *J. Environ. Qual.* **36,** 1821–1832.

Kirschbaum, M. U. F. (1994). The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. *Soil Biol. Biochem.* **27**(6), 753–760.

Kirschbaum, M. U. F. (2000). Will changes in soil organic carbon act as a positive or negative feedback on global warming? *Biogeochemistry* **48,** 21–51.

Kong, A. Y., Six, J., Bryant, D. C., Denison, R. F., and van Kessel, C. (2005). The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. *Soil Sci. Soc. Am. J.* **69,** 1078–1085.

Kramer, A. W., Doane, T. A., Horwath, W. R., and van Kessel, C. (2002). Combining fertilizer and organic inputs to synchronize N supply in alternative cropping systems in California. *Agric. Ecosyst. Environ.* **91,** 233–243.

Kroodsma, D. A., and Field, C. B. (2006). Carbon sequestration in California Agriculture, 1980–2000. *Ecol. Appl.* **16**(5), 1975–1985.

Kuo, S. U., Sainju, U. M., and Jellum, E. J. (1997). Winter cover cropping influence on nitrogen in soil. *Soil. Sci. Soc. Am. J.* **61,** 1392–1399.

Kyoto Protocol, (1997). Kyoto protocol to the United Nations framework convention on climate change. United Nations.

Lal, R. (1991). Soil structure and sustainability. *J. Sustain. Agric.* **1**(4), 67–92.

Lal, R. (1997). Residue management, conservation tillage and soil restoration for mitigating greenhouse effect by CO_2 enrichment. *Soil Tillage Res.* **43,** 81–107.

Lal, R. (2002). Soil carbon dynamics in cropland and rangeland. *Environ. Pollut.* **116,** 353–362.

Lal, R. (2003). Global potential of soil carbon sequestration to mitigate the greenhouse effect. *Crit. Rev. Plant Sci.* **22**(2), 151–184.

Lal, R. (2004a). Soil carbon sequestration to mitigate climate change. *Geoderma* **123,** 1–22.

Lal, R. (2004b). Soil carbon sequestration impacts on global climate change and food security. *Science* **304,** 1623–1627.

Lal, R. (2005). Soil erosion and carbon dynamics. *Soil Tillage Res.* **81,** 137–142.

Lal, R. (2007). Farming carbon. *Soil Tillage Res.* **96,** 1–5.

Lal, R. (2008). Carbon sequestration. *Philos. Trans. R. Soc. Lond., B, Biol. Sci.* **363,** 815–830.

Lal, R., and Bruce, J. P. (1999). The potential of world cropland soils to sequester C and mitigate the greenhouse effect. *Environ. Sci. Policy* **2,** 177–185.

Lal, R., and Kimble, J. M. (1997). Conservation tillage for carbon sequestration. *Nutr. Cycl. Agroecosyst.* **49,** 243–253.

Lal, R., Eckert, D. J., Fausey, N. R., and Edwards, W. M. (1990). Conservation tillage in sustainable agriculture. *In* ''Sustainable Agricultural Systems'' (C. A. Edwards, Ed.), pp. 203–225. United States Agency for International Development.

Lal, R., Regnier, E., Eckert, D. J., Edwards, W. M., and Hammond, R. (1991). Expectations of cover crops for sustainable agriculture: An information data base on cover crops. *In* ''Cover Crops for Clean Water'' (W. L. Hargrove, Ed.). Soil and Water Conservation Society, Ankeny, IA.

Lal, R., Kimble, J. M., and Stewart, B. A. (1995). Global Climate Change and Cold Regions Ecosystems. Advances in Soil Science, Boca Raton, FL.

Lal, R., Kimble, J. M., Follett, R. F., and Cole, C. V. (1998). The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect. Ann Arbor Press, Chelsea, MI.

Lee, J., Six, J., King, A. P., Van Kessel, C., and Rolston, D. E. (2006). Tillage and field scale controls on greenhouse gas emissions. *J. Environ. Qual.* **35,** 714–725.

Lee, J., De Gryze, S., and Six, J. (2008). Effect of Climate Change on Field Crop Production in the Central Valley of California. California Energy Commission (CEC) Publication (CEC-500-2008-xxx).

Li, C., Salas, W., and Los Huertos, M. (2004). Quantifying carbon dynamics and greenhouse gas emissions in agricultural soils of California: A scoping study. (PIER Final Project Report, CEC-500-04-038), pp. 55.

Luna, J. M., and Staben, M. L. (1998). Biologically integrated vegetable production systems using cover crops and strip tillage. *Proc. Soil Biology MiniCourse*, pp. 77–83. Corvallis, OR.

Madden, N. M., Southard, R. J., and Mitchell, J. P. (2008). Conservation tillage reduces PM10 emissions in dairy forage rotations. *Atmos. Environ.* **42,** 3795–3808.

Madden, N. M., Southard, R. J., and Mitchell, J. P. (2009). Soil water content and soil disaggregation by disking affects PM10 emissions. *J. Environ. Qual.* **38,** 36–43.

Malpassi, R. N., Kaspar, T. C., Parkin, T. B., Cambardella, C. A., and Nubel, N. A. (2000). Oat and rye root decomposition effects on nitrogen mineralization. *Soil Sci. Soc. Am. J.* **64,** 208–215.

Meisinger, J. J., Hargrove, W. L., Mikkelsen, R. L., Williams, J. R., and Benson, V. W. (1991). Effects of cover crops on groundwater quality. *In* ''Cover Crops for Clean Water'' (W. L. Hargrove, Ed.), pp. 57–68. Soil and Water Conservation Society, Ankeny, IA.

Minoshima, H., Jackson, L. E., Cavagnaro, T. R., Sánchez-Moreno, S., Ferris, H., Temple, S. R., Goyal, S., and Mitchell, J. P. (2007a). Soil food webs and carbon dynamics in response to conservation tillage in California. *Soil Sci. Soc. Am. J.* **71,** 952–963.

Minoshima, H., Jackson, L. E., Cavagnaro, T. R., and Ferris, H. (2007b). Short-term fates of 13C-depleted cowpea shoots in no-tillage and standard tillage soils. *Soil Sci. Soc. Am. J.* **71,** 1859–1866.

Mitchell, J. P., Thomsen, C. D., Graves, W. L., and Shennan, C. (1999). Cover crops for saline soils. *J. Agron. Crop Sci.* **183,** 167–178.

Mitchell, J. P., Munk, D. S., Prys, B., Klonsky, K., Wroble, J. F., and De Moura, R. L. (2006). Conservation tillage production systems compared in San Joaquin Valley cotton. *Calif. Agric.* **60**(3), 140–145.

Mitchell, J. P., Klonsky, K., Shrestha, A., Fry, R., DuSault, A., Beyer, J., and Harben, R. (2007). Adoption of conservation tillage in California: Current status and future perspectives. *Aust. J. Exp. Agric.* **47,** 1383–1388.

Mitchell, J. P., Southard, R. J., Madden, N. M., Klonsky, K. M., Baker, J. B., DeMoura, R. L., Horwath, W. R., Munk, D. S., Wroble, J. F., Hembree, K. J., and Wallender, W. W. (2008). Transition to conservation tillage evaluated in San Joaquin Valley cotton and tomato rotations. *Calif. Agric.* **62**(2), 74–79.

Nordstrom, K. F., and Hotta, S. (2004). Wind erosion from cropland in the USA: A review of problems, solutions and prospects. *Geoderma* **121,** 157–167.

Ogle, S. M., Breidt, F. J., and Paustian, K. (2005). Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. *Biogeochemistry* **72**(1), 87–121.

Parton, W. J., Hartman, M., Ojima, D., and Schimel, D. (1998). DAYCENT and its land surface submodel: Description and testing. *Glob. Planet. Change* **396,** 1–14.

Paustian, K., Andren, O., Janzen, H. H., Lal, R., Smith, P., Tiad, G., Tiessen, H., Van Noordwijk, M., and Woomer, P. L. (1997). Agricultural soils as a sink to mitigate CO_2 emissions. *Soil Use Manag.* **13,** 230–244.

Paustian, K., Six, J., Elliott, E. T., and Hunt, H. W. (2000). Management options for reducing CO_2 emissions from agricultural soils. *Biogeochemistry* **48,** 147–163.

Pimentel, D., and Pimentel, M. (1996). Food, Energy and Society. Colorado University Press, Niwot, CO.

Post, W. M., and Kwon, K. C. (2000). Soil carbon sequestration and land-use change: Processes and potential. *Glob. Chang. Biol.* **6,** 317–327.

Post, W. M., and Mann, L. K. (1990). Changes in soil organic carbon and nitrogen as a result of cultivation. *In* "Soils and the Greenhouse Effect" (A. F. Bouwman, Ed.), pp. 401–406. John Wiley & Sons, New York, NY.

Post, W. M., Izaurralde, R. C., Jastrow, J. D., McCarl, B. A., Amonette, J. E., Bailey, V. L., Jardine, P. M., West, T. O., and Zhou, J. (2004). Enhancement of carbon sequestration in U.S. soils. *Bioscience* **54,** 895–908.

Poudel, D. D., Horwath, W. R., Mitchell, J. P., and Temple, S. R. (2001). Impacts of cropping systems on nitrogen storage and loss. *Agric. Syst.* **68,** 253–268.

Poudel, D. D., Horwath, W. R., Lanini, W. T., Temple, S. R., and van Bruggen, A. H. C. (2002). Comparison of soil N availability and leaching potential, crop yields and weeds in organic, low-input and conventional farming systems in northern California. *Agric. Ecosyst. Environ.* **90,** 125–137.

PPIC. (2008). Population Projections for California, 2000–2025 Public Policy Institute of California publication, (p2).

Puget, P., and Lal, R. (2005). Soil organic carbon and nitrogen in a Mollisol in central Ohio as affected by tillage and land use. *Soil Tillage Res.* **80,** 201–213.

Rasmussen, P. E., and Collins, H. P. (1991). Long-term impacts of tillage, fertilizer and crop residue on soil organic matter in temperate semiarid regions. *Adv. Agron.* **45,** 93–134.

Reicosky, D. C., and Forcella, F. (1998). Cover crop and soil quality interactions in agroecosystems. *Soil Water Conserv.* **53**(3), 224–229.

Reicosky, D. C., and Lindstrom, M. J. (1993). Fall tillage method: Effect on short-term carbon dioxide flux from soil. *Agron. J.* **85,** 1237–1243.

Reicosky, D. C., Dugas, W. A., and Torbert, H. A. (1997). Tillage-induced soil carbon dioxide loss from different cropping systems. *Soil Tillage Res.* **41,** 105–118.

Rigby, D., and Cáceres, D. (2001). Organic farming and the sustainability of agricultural systems. *Agric. Sys.* **68,** 21–40.

Roberson, E. B., Sarig, S., and Firestone, M. K. (1991). Cover crop management of polysaccharide-mediated aggregation in an orchard soil. *Soil Sci. Soc. Am. J.* **55,** 734–739.

Robertson, G. P., Paul, E. A., and Harwood, R. R. (2000). Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. *Science* **289,** 1922–1925.

Rochette, P., and Angers, D. A. (1999). Soil surface carbon dioxide fluxes induced by spring, summer, and fall mouldboard plowing in a sandy loam. *Soil Sci. Soc. Am. J.* **63,** 621–628.

Rovira, A. D., and Greacen, E. L. (1957). The effect of aggregate disruption on the activity of microorganisms in the soil. *Aust. J. Agric. Res.* **8,** 659–673.

Ryden, J. C., and Lund, L. J. (1980). Nature and extent of directly measured denitrification losses from some irrigated vegetable crop production units. *Soil Sci. Soc. Am. J.* **44,** 505–511.

Sainju, U. M., Singh, B. P., and Whitehead, W. F. (2002). Long-term effects of tillage, cover crops, and nitrogen fertilization on organic carbon and nitrogen concentrations in sandy loam soils in Georgia, USA. *Soil Tillage Res.* **63,** 167–179.

Schlesinger, W. H. (1997). Biogeochemistry: An Analysis of Global Change. Academic Press, San Diego, CA.

Schlesinger, W. H. (2000). Carbon sequestration in soils: Some caution amidst optimism. *Agric. Ecosyst. Environ.* **82,** 121–127.

Seiter, S., and Horwath, W. R. (2004). Strategies for managing soil organic matter to supply plant nutrients. *In* ''Soil Organic Matter in Sustainable Agriculture'' (F. Magdoff and R. R. Weil, Eds.), pp. 269–293. CRC Press, Boca Raton, FL(Chapter 9).

Shih, T. T. (2002). Timberland conversion in California from 1969 to 1998. (Fire and Resource Assessment Program, Technical Working Paper 1-01-02).

Shipitalo, M. J., and Protz, R. (1989). Chemistry and micromorphology of aggregation in earthworm casts. Geoderma, **45**, 357–374.

Shipley, P. R., Meisinger, J. J., and Decker, A. M. (1992). Conserving residual corn fertilizer nitrogen with winter cover crops. *Agron. J.* **84,** 869–876.

Shrestha, A., Mitchell, J. P., and Lanini, W. T. (2007). Subsurface drip irrigation as a weed management tool for conventional and conservation tillage tomato (*Lycopersicon esculentum* Mill.) production in semi-arid agroecosystems. *J. Sustain. Agric.* **31**(2), 91–112.

Sims, J. R., and Slinkard, A. E. (1991). Development and evaluation of germplasm and cultivars of cover crops. *In* ''Cover Crops for Clean Water'' (W. L. Hargrove, Ed.), Proceedings of an international conference, West Tennessee, Experiment Station, Jackson. April 9–11, pp. 121–129. Soil and Water Conservation Society, Ankeny, IA.

Six, J., Elliott, E. T., Paustian, K., and Doran, J. W. (1998). Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Sci. Soc. Am. J.* **62,** 1367–1377.

Six, J., Elliott, E. T., and Paustian, K. (1999). Aggregate and soil organic matter dynamics under conventional and no-tillage systems. *Soil Sci. Soc. Am. J.* **63,** 1350–1358.

Six, J., Elliott, E. T., and Paustian, K. (2000). Soil macroaggregate turnover and microaggregate formation: A mechanism for C sequestration under no-tillage agriculture. *Soil Biol. Biochem.* **32,** 2099–2103.

Six, J., Feller, C., Denef, K., Ogle, S. M., Sa, M. J. C., and Albrecht, A. (2002). Soil organic matter, biota and aggregation in temperate and tropical soils—Effects of no-tillage. *Agron. Agric. Environ.* **22,** 755–775.

Six, J., Ogle, S. M., and Breidt, F. J. (2004). The potential to mitigate global warming with no-tillage management is only realized when practiced in the long term. *Glob. Chang. Biol.* **10,** 155–160.

Six, J., Frey, S. D., Thiet, R. K., and Batten, K. M. (2006). Bacterial and fungal contributions to C-sequestration in agroecosystems. *Soil Sci. Soc. Am. J.* **70,** 555–569.

Smart, D. R., *et al.* (2005). Grapevine root distributions: A comprehensive analysis and a review. ''Soil Environment and Vine Mineral Nutrition'', (American Society of Enology and Viticulture, June 29–30, 2005, San Diego, California).

Snapp, S. S., Swinton, S. M., Labarta, R., Mutch, D., Black, J. R., Leep, R., Nyiraneza, J., and O'Neil, K. (2005). Evaluating cover crops for benefits, costs and performance within cropping system niches. *Agron. J.* **97,** 322–332.

Snyder, M. A., Bell, J. L., Sloan, L. C., Duffy, P. B., and Govindasamy, B. (2002). Climate responses to a doubling of atmospheric carbon dioxide for a climatically vulnerable region. *Geophys. Res. Lett.* **29,** 11.

Sperow, M., Eve, M., and Paustian, K. (2003). Potential soil c sequestration on U.S. agricultural soils. *Clim. Change* **57**(3), 319–339.

Steenwerth, K., and Belina, K. M. (2008a). Cover crops enhance soil organic matter, carbon dynamics and microbiological function in a vineyard agroecosystem. *Appl. Soil Ecol.* **40,** 359–369.

Steenwerth, K., and Belina, K. M. (2008b). Cover crops and cultivation: Impacts on soil N dynamics and microbiological function in a Mediterranean vineyard agroecosystem. *Appl. Soil Ecol.* **40,** 370–380.

Swift, R. S. (2001). Sequestration of carbon by soil. *Soil Sci.* **166**(11), 858–871.

Teasdale, J. R. (1996). Contribution of cover crops to weed management in sustainable agricultural systems. *J. Prod. Agric.* **9,** 475–479.

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., and Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature* **418,** 671–677.

Tisdall, J. M. (1996). Formations of soil aggregates and accumulation of soil organic matter. *In* "Structure and Organic Matter Storage in Agricultural Soils" (M. R. Carter and B. A. Stewart, Eds.), pp. 57–97. Library of Congress Cataloguing-in-Publication Data, Boca Raton, FL.

Tisdall, J. M., and Oades, J. M. (1982). Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **33,** 141–163.

Torstensson, G., Aronsson, H., and Bergstrom, L. (2006). Nutrient use efficiencies and leaching of organic and conventional cropping systems in Sweden. *Agron. J.* **98,** 603–615.

Unger, P. W., Schomberg, H. H., Dao, T. H., and Jones, O. R. (1997). Tillage and crop residue management practices for sustainable dryland farming systems. *Ann. Arid Zone* **36,** 209–232.

Uri, N. D., and Konyar, K. (1996). Conservation tillage and the use of energy and other inputs in U.S. agriculture. *Appl. Energy* **54**(2), 75–102.

Veenstra, J. J., Horwath, W. R., Mitchell, J. P., and Munk, D. S. (2006). Conservation tillage and cover cropping influence soil properties in San Joaquin Valley cotton tomato crop. *Calif. Agric.* **60**(3), 146–153.

Veenstra, J. J., Horwath, W. R., and Mitchell, J. P. (2007). Conservation tillage and cover cropping effects on total of carbon and aggregate-protected carbon in irrigated cotton and tomato rotations. *Soil Sci. Soc. Am. J.* **71,** 362–371.

Venterea, R. T., Burger, M., and Spokas, K. A. (2005). Nitrogen oxide and methane emissions under varying tillage and fertilizer management. *J. Environ. Qual.* **34**(5), 1467–1477.

Waltman, S. W., and Bliss, N. L. (1997). Estimates of SOC Content NSSC, Lincoln, NE.

Warnert, J. E. (1996). News from the recycling front: Agriculture could provide a major market for recycled green waste. *Calif. Agric.* **50**(5), 8–10.

Weinert, T. L., Pan, W. L., Moneymaker, M. R., Santo, G. S., and Stevens, R. G. (2002). Nitrogen recycling by nonleguminous winter cover crops to reduce leaching in potato rotations. *Agron. J.* **94,** 365–372.

West, T. O., and Marland, G. (2002). A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: Comparing tillage practices in the United States. *Agric. Ecosyst. Environ.* **91,** 217–232.

West, T. O., and Post, W. M. (2002). Soil organic carbon sequestration rates by tillage and crop rotation: A global data analysis. *Soil Sci. Soc. Am. J.* **66,** 1930–1946.

West, T. O., and Six, J. (2007). Considering the influence of sequestration duration and carbon saturation on estimates of soil carbon capacity. *Climatic Change* **80,** 1–2.

Williams, W. A., Finfrock, D. C., Davis, L. L., and Mikkelsen, D. S. (1957). Green Manuring and Crop Residue Management in Rice Production. *Soil. Sci. Soc. Am. J.* **21,** 412–415.

Wyland, L. J., Jackson, L. E., Chaney, W. E., Klonsky, K., Koike, S. T., and Kimple, B. (1996). Winter cover crops in a vegetable cropping system: Impacts on nitrate leaching, soil water, crop yield, pests and management costs. *Agric. Ecosyst. Environ.* **59,** 1–17.