Understanding the Science Behind Climate Smart Agriculture in California:

A comprehensive review of literature

Photo by Monika Kost

Presented by:

Community Alliance with Family Farmers & University of California, Davis

www.caff.org

February 2024

Acknowledgments

Author:
• Jessica L. Chiartas, Department of Plant Sciences, University of California, Davis

Contributors:
• Sara E. Tiffany, Community Alliance with Family Farmers
• Emily Ayala, Community Alliance with Family Farmers

A special thanks to our funders that have supported this work:
• The Gaia Foundation
• Globetrotter Foundation
• Clarence E. Heller Charitable Foundation
• 11th Hour Project of the Schmidt Family Foundation

Design and layout by Megan Sabato

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Soil Health Primer

Improving soil structure and increasing soil organic matter are key goals of climate smart agriculture.

While soil texture (i.e. sand, silt, and clay content) changes very slowly and may not be noticeable on a human timescale, soil structure can change rapidly due to management. Heavy machinery can physically compact soil particles, reducing pore space, while tillage can break apart soil aggregates and contribute to a loss of soil structure. Conversely, the physical and biochemical action of roots and other organisms and the overall accumulation of soil organic matter (SOM) can greatly improve soil structure (Lal et al. 2015; Lal et al. 2018). In sandy soils, SOM helps particles bind together, increasing aggregation, whereas in clayey soils it increases aeration.

Improving soil structure and increasing SOM are key goals of climate smart agriculture, as both tend to increase infiltration and drainage, improve aeration, enhance water and nutrient holding capacity, and reduce the risk of compaction and erosive loss (Steenwerth et al. 2014; Lal et al. 2018). Traditionally, soil and agricultural scientists have viewed this process of building organic matter as a simple equation of carbon in, carbon out. SOM, then, could only be increased by increasing total carbon inputs (i.e. roots, residues or organic amendments) or reducing total losses (from tillage, erosion, etc). A more nuanced understanding of SOM highlights its preservation as an ecosystem property dictated by soil structure, microbial physiology and overall efficiency with which soil organisms are able to function (Schmidt et al. 2011). All organic carbon in the soil is "fair-game" for microbial decomposition -- only stabilized by complexation with clays and/or physical occlusion inside a pore or an aggregate -- and once consumed will either be respired as CO_2, stored as living biomass, or converted into extracellular compounds (Schmidt et al. 2011; Lehman et al. 2015).

Heavy machinery can physically compact soil particles, reducing the pore space through which air, water, and nutrients flow and restricting root growth. *Photo by Carlo Fanti*

Advanced imaging tools have recently revealed that 50-80% of SOM is composed of dead microbial biomass and their byproducts (Liang et al. 2010; Miltner et al. 2012). Thus, in order to increase SOM, microbial populations must first increase. To do this, the efficiency with which these organisms convert carbon-based inputs to microbial biomass must increase. The prevailing theory holds that this microbial carbon use efficiency, or the amount of carbon dioxide respired per unit of carbon that enters the system, ultimately dictates the long-term potential for carbon sequestration (Cotrufo et al. 2015). Microbial physiology is thought to improve through reduced physical or chemical disturbance and/or an increase in access to water, carbon, and other resources (Kallenbach et al. 2015).

Keeping a diversity of plants in the ground as much of the year as possible is a crucial part of this equation, as photosynthesis converts carbon dioxide from the atmosphere into sugars, amino, and nucleic acids, and pumps them underground via roots, thereby feeding and growing a diverse population of microbial organisms whose biomass eventually become SOM. Once this population is primed to receive a certain level of carbon inputs (and/or other energy sources), any reduction in the amount of these inputs (i.e. after herbicide spray, tillage, etc.) may force the microbial community into starvation mode; resultantly, they will either invest energy in protective mechanisms or tap into old reserves of soil organic matter to meet their energy needs. A diversity of carbon-based inputs enter the soil system (roots, root exudates, aboveground plant residues, animal residues and manures, etc.), all of which must go through this microbial filter to be stabilized as SOM.

The ratio of carbon to nutrients (i.e. nitrogen, phosphorus, sulfur, etc.) and the availability of other resources (i.e. water, oxygen), determines what portion of carbon will be allocated to building microbial biomass, what portion goes to the production of exudates and enzymes, and what portion returns to the atmosphere as carbon dioxide (via respiration). If any nutrient is lacking from the system, microbes must expend extra energy (i.e. producing an enzyme, moving across the pore space) to access it, in order to build their biomass. In some cases, microbes may even resort to old SOM, breaking down the stable carbon reserves in search of nutrients and producing carbon dioxide in the process. In this way, the quality of carbon inputs may be just as important as the quantity. Inputs made of more complex compounds like lignin require more energy to decompose, whereas simple sugars like those produced by roots are more easily assimilated into biomass.

As temperature and moisture levels increase, rates of chemical reactions and biological activity also increase. In California, relatively high year-round temperatures allow for relatively rapid decomposition of SOM. Although dry conditions in the summer should limit the extent to which decomposition actually occurs, 3/4 of California's cropland is irrigated. This results in

Keeping a diversity of plants in the ground as much of the year as possible is crucial to feeding/growing a diverse microbial community and building or retaining SOM.

Three-quarters of California's cropland is irrigated, which combined with high summer temperatures may allow for rapid decomposition when otherwise dry conditions would impede microbial activity.

higher plant productivity and thus, greater inputs of both above and belowground carbon, and increased moisture content, which encourages greater microbial activity. Whether this increased activity translates to a net source or sink of carbon dioxide is determined largely by the microbial filter -- their physiology, growth rate, and carbon use efficiency. In semi-arid environments like those that occur in non-irrigated agriculture in California, where microbial stress can often be high, specific respiration rates (respiration per unit microbial biomass) are often higher than other climates, indicating lower carbon use efficiency and challenges in increasing SOM (Zhou et al. 2009; Doetterl et al. 2015). While high respiration can be a sign of soil health, indicating a large and active microbial population; an important precursor to SOM formation (Schmidt et al. 2015), high respiration coming from a relatively small microbial population indicates stress and/or poor physiology.

For instance, Kong et al. 2005 found that while cover crops and manure increased the emissions of carbon dioxide in the short-term (i.e., days or weeks), these same systems sequestered carbon over the long-term (years). As such, measuring carbon dioxide emissions is not sufficient to ascertain whether a practice is a net source or sink of carbon and will not be used as such in this review. Capturing emissions also presents logistical challenges (labor, cost, etc.), as fluxes of GHG are highly dynamic and must be monitored frequently (if not constantly) to provide a complete, accurate picture. Furthermore, the time frame of carbon sequestration in the soil is important for assuring that net carbon gain occurs in a cropping system or rangeland; permanence is defined by Kyoto Protocol as 100 years of storage. Finally, with CO_2 accounting for such a small percentage (9%) of total agricultural emissions in California (CARB 2011, Culman et al. 2014), efforts to reduce agricultural emissions must include other important GHGs. Nitrous oxide (NOx), for example, has 300 times the radiative forcing, or GHG effect, of CO_2 and contributes to 33% of agricultural GHG. NOx is also arguably the most sensitive GHG to management and thus offers significant reduction potential, in the near-term. Methane from rice and livestock operations accounts for the remaining 58% of agricultural emissions (CARB 2011, Culman et al. 2014).

Great uncertainty exists as to the impact that climatic shifts will have in any given cropping system, soil type, or microclimate across the state. California agriculture will need to remain innovative, resilient, and adaptable in the coming decades. In recognition of the challenges and opportunities this presents, a multitude of stakeholders must collaborate to integrate existing and local knowledge, the latest scientific research, institutional support, and technical assistance. Here we report on several climate smart practices that offer promise for mitigation and adaptation in California agroecosystems, as well as relevant knowledge gaps that require further investigation in our diverse region.

Introduction

California is unmatched, nationally, in the diversity and productivity of its landscapes. With a rich geologic history and a latitudinal range of nearly 10 degrees, the variation in topography, microclimates, and soil types allows for year-round cultivation of over 400 agricultural crops. This diversity provides immense economic and aesthetic value, but presents challenges in identifying best management practices that are simultaneously productive and sustainable across regions, cropping systems, and soil types (Kanter et al. 2021; Devine et al. 2022); especially in the face of increasing climatic uncertainties.

Climate and agriculture are inextricably linked. The success of a crop depends directly on precipitation, temperature, and their combined impact on water supply as key drivers of photosynthesis, respiration, nutrient cycling, and microbial activity, as well as indirectly, in the ability to resist pests, weeds, and pathogens. While weather variability is a hallmark of California agriculture, regional models project California to be disproportionately impacted in the coming years by rising temperatures and increased severity/frequency of extreme weather events and wildfires (Hayhoe et al. 2004; Cayan et al. 2006; Medellin-Azuara et. al., 2011). Using historic weather data, a report by the California Office of Environmental Health Hazard Assessment (OEHHA) has corroborated these models, identifying 36 measurable indicators that show discernible evidence of shifting climatic patterns in the state (OEHHA 2018).

The OEHHA report highlights a steady increase in average annual temperatures and drought severity since record keeping began in 1895, with temperatures from 2014 through 2018 recognized as the hottest on record. In parts of the Central Valley, winter chill hours declined by 30% (from 1950 to 2009) and the number of extreme heat days increased, directly impacting agricultural production (Byrnes et al. 2017; OEHHA 2018; Pathak et al. 2018). Increased temperatures have also exacerbated the effects of drought by increasing evaporation, decreasing snowpack, and reducing overall soil moisture across the growing season (Hanak and Mount 2015). The 2012 - 2016 drought was the most severe in nearly 500 years (Belmecheri et al. 2016; OEHHA 2018). This "perfect storm" of heat and drought stress leaves crops susceptible to certain pest, weed, and pathogen pressure (Byrnes et al. 2017). As of 2016, the drought resulted in over 500,000 acres of fallowed land, 38,000 lost jobs, and costs of approximately $1.5 billion (Sumner et al. 2015; Medellin-Azuara et al. 2016; CDWR, 2016). By 2060, continued shifts in climate are predicted to decrease yields by 40% in avocados, 20% in oranges, grapes, walnuts, and almonds; as well as in strawberries, cherries, and apricots (Pathak et al. 2018).

California leads the nation in agricultural production with:

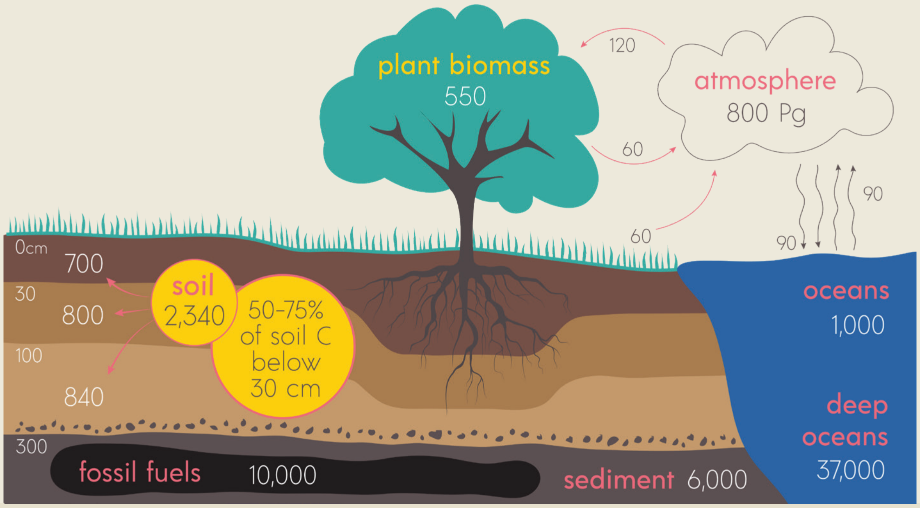
- 69,000 farms, generating over $49 billion in sales (roughly 13% of US agricultural revenue),
- employing around 1.1 million people,
- providing 50% of fruit and vegetables,
- 42% of nuts,
- 20% of rice,
- and 21% of dairy produced domestically (CDFA 2020).

Carbon Sequestration: Challenges, Uncertainties, and Opportunities

The total amount of carbon found in soil is greater than the entirety of carbon found in the earth's plant/animal biomass and atmosphere combined. Moreover, 50-75% of that soil carbon is stored below typical sampling depths of 30 cm, although research has shown it may be susceptible to additions/losses associated with management. While there is great potential for carbon sequestration in California farmland and rangeland, a nuanced look at the challenges surrounding the practices and science of carbon sequestration is essential, as discussed in Bowles et.al (2021) presentation, *Let's Talk About Carbon Sequestration!*

The amount of carbon that soils can store depends on soil physical and chemical properties such as texture and pH, as well as climate and land management. For instance, sandier soils and more arid climates tend to have less soil carbon. Conventional agriculture practices, such as heavy tillage, leaving the ground bare, and overgrazing can result in a net loss of carbon; conversely, climate smart farming practices like cover cropping and conservation tillage can result in net carbon gains in the system. But even when carbon gains occur, sustaining this carbon in the soil requires an ongoing commitment to these types of farming practices. In addition, from the perspective of climate change mitigation, agricultural management that sequesters soil carbon may affect production of potent greenhouse gases like nitrous oxide and methane, and thus need to be evaluated as well, to provide an accurate depiction of the total GHG footprint.

Graphical Depiction of the Distribution of Global Carbon Stocks
1 Pg (or Petagram) = 1 Gigatonne (or 1 billion metric tons)



In order to accurately account for carbon stock changes, soil samples must be collected, which requires substantial resources and technical expertise. In addition to collecting soil samples and measuring the concentration (or percentage) of SOM or SOC, an additional field measurement of bulk density (i.e. the mass of soil per unit volume) must be collected to estimate the actual amount of carbon in the soil (typically expressed in tons/hectare). Measuring and monitoring soil carbon can also be complicated by soil's rather heterogeneous (or variable) nature, which often necessitates a high amount of samples (and thus cost/labor) to achieve accurate measurements. For this reason, models are commonly used to project carbon stocks, despite often misestimating results as they are based on limited data from experiment stations rather than working farms.

Aside from these challenges, Bowles et al. (2021) note several uncertainties related to the future of carbon storage also exist. **One question to consider is, will the warming climate increase the activity of soil microbes, thereby driving decomposition and decreasing soil carbon stocks? On the other hand, will climate change increase plant activity, resulting in higher carbon stocks?** The interplay of these processes and their effects on overall carbon levels in the soil is difficult to predict. Finally, as markets for carbon sequestration are developed, questions remain regarding potential impacts to land access, tenure, and the value of farmland as increased investment in agricultural land and land grabbing may occur. Given these challenges and uncertainties, the myriad co-benefits of building carbon in the soil (further discussed in this text), are still worth considering for growers throughout California.

Illustration Credit: Soil Life

California Climate Smart Agriculture

In 2006, the California state government passed the Global Warming Solutions Act (AB32). The bill mandates that California reduce emissions to 40% of 1990 levels by 2030 (CARB 2017).

Intensive agriculture (i.e. conventional tillage, winter fallow, monoculture) has depleted soil organic matter reserves.

Achieving this ambitious goal requires the adoption of reduction strategies across all economic sectors. Although agriculture and forestry accounts for a mere 8% of the state's current GHG budget (excluding agrochemical production/transportation) and contributes a relatively small amount of direct emissions statewide, it is one of the most vulnerable sectors to climate variability and provides a rare opportunity to achieve negative emissions, or the physical removal of carbon from the atmosphere (CARB 2011, CARB 2017, Hansen et al. 2017). Recent projections indicate that reducing emissions will no longer be sufficient to avoid a 2°C rise in temperature. Rather, we must remove an additional 150 Pg of carbon from the atmosphere and sequester, or store it, in a more stable reservoir (Hansen et al. 2017).

Working lands provide one such reservoir, both aboveground in the form of woody biomass (i.e. trees, shrubs and vines) and belowground as soil organic matter (SOM). SOM is made of approximately 50% carbon and accounts for 2,344-3,012 Pg of carbon globally, dwarfing the atmospheric pool of carbon (800 Pg) and the standing vegetation (550 Pg) combined (Jobbagy & Jackson 2000; Sanderman et al. 2017). Furthermore, SOM contains all 14 plant essential nutrients in a slow-release form and provides a host of co-benefits associated with resiliency and adaptability on-farm, including improved soil structure and reduced risk of erosion, increased infiltration and water holding capacity, enhanced fertility and nutrient retention, and increased biodiversity and resistance to pests and disease (Lal et al. 2015; Bossio et al. 2020). Intensive agriculture (i.e. conventional tillage, winter fallow, monoculture) has depleted soil organic matter reserves to an average 1-2% statewide (DeClerck et al. 2003). Recognizing the potential of improved management to counteract these trends (Lal et al. 2015; Paustian et al. 2016; Bossio et al. 2020), several efforts have emerged in California to increase the adoption of Climate Smart Agriculture including the CDFA's Healthy Soils Program, CARB's Natural and Working Lands Implementation Plan. In 2020, Governor Newsom's Executive Order N-82-20, created the California Biodiversity Collective, a multi-agency collaboration to protect 30% of California's land and coastal waters by 2030. Efforts to encourage and incentivize these practices have been referred to by a number of different terminologies including: Healthy Soils, Climate Smart Farming, Carbon Farming and Regenerative Farming. For the purposes of this review, we will refer to the practices and farming systems referenced by these various terminologies as Climate Smart Agriculture (CSA).

The purpose of this review is to bring to light the recent science supporting the ecosystem service benefits of key CSA practices, with particular focus on the capacity of these practices to mitigate and adapt to climate change.

We emphasize, however, that the ideal suite of practices and the impacts of these practices will vary across climates, soil types, and cropping systems (Fine et al. 2017; Bunemann et al., 2018; Devine et al. 2022) and encourage efforts to better define unique soil health contexts in California (Devine & O'Geen 2020), in order to better grapple with the complexity of our agricultural landscapes.

What is Climate Smart Agriculture?

According to the FAO, CSA is more a set of principles than practices, intended to help in "identifying production systems and enabling institutions best suited to respond to the challenges of climate change for specific locations." Acknowledging the vast diversity of climates, soil types, cropping systems, as well as political, socioeconomic and cultural contexts, CSA employs a systems approach to identify site-specific trade-offs, synergies, and associated costs and benefits (Lippert et al. 2014; Steenwerth et al. 2014). Through multi-stakeholder engagement, CSA strives to achieve the triple bottom line of:

1. **sustainable agricultural production and income generation**
2. **adaptation and resilience to climate change, and**
3. **the reduction and/or removal of GHG emissions (FAO 2013a; Lipper et al. 2014).**

Methods

In an effort to scientifically validate practices recognized for their ability to contribute to climate mitigation via increased soil organic matter (SOM); adaptation via improved cropping systems; and resilience and resistance via improved soil health and ecosystem function, a preliminary literature review was conducted to assess and better understand the unique conditions and constraints relevant to agroecosystems in a "Mediterranean-type" climate with cool, wet winters and dry, warm summers. Literature searches were conducted on Web of Science, Science Direct, Agricola, and Google Scholar for the following combinations: either "cover crop," "green manure," "compost," "manure," "mulch," "crop rotations," "no tillage," "reduced tillage," "conservation tillage," "hedgerows," "filter strips," "windbreaks," "field edges," "perennials," "integrated crop livestock," "crop diversity," or "diversified farming systems," AND "agriculture," "soil, "infiltration," "aggregate stability," "aggregation," "surface hardness," "compaction," "water quality," "erosion," "yield," "productivity," "soil organic matter," "soil C," "profit," "inputs," "reduced use" AND "California." If information could not be found to support management practices capable of conferring climate mitigation, adaptation, and resilience benefits in California specifically, the search was refined using the term "Mediterranean" or "semi-arid." This white paper was compiled by combining and synthesizing the literature with on-the-ground knowledge shared by experts in the field and the experience of Community Alliance with Family Farmers' Climate Smart Farming Program.

Photo by Carlo Fanti

Cover Cropping

Cover cropping involves the planting of an annual, perennial, or mix of species between cash crops, not with the intent to harvest, but to improve soil health and fertility.

By maintaining roots in the ground, cover crops physically hold the soil in place and reduce the risk of erosion, while also promoting aggregation and maintaining channels that allow for improved water infiltration and percolation (Blanco-Canqui 2020). Constant cover also translates to increased carbon inputs (Mitchell et al. 2015; Poeplau & Don 2015) to feed the soil microbial community and habitat for pollinators and beneficial species that aid in pest/pathogen suppression (Bugg et al., 2007; Suddick et al. 2010). By maintaining ground coverage, cover crops also suppress weeds, buffer against extreme surface temperatures, and reduce evaporation rates (Wagger et al 1998; Sharma et al. 2018; Shackleford et al. 2019). In semi-arid environments like California, however, the potential increase in water infiltration and storage must be weighed against the potential increase in transpiration performed by living plants.

Selection of a species, or mix of species, to plant depends on the intended use, as well as a variety of other factors, including climate, soil type, risk of invasiveness, and subsequent crop. For instance, if nitrogen for the cash crop is the priority, a cover crop that includes legumes (i.e. vetch, clover, cowpea, and fava bean) may be the best choice. Legumes have evolved in symbiosis with the soil bacteria *Rhizobia,* which is able to take nitrogen gas (N_2) directly from the atmosphere and fix it into a plant-available form of nitrogen (NH_3 or NH_4^+). When incorporated while immature (and relatively high in nitrogen), microbes are able to break down residues and supply sufficient N for the subsequent cash crop.

If preventing nutrient loss is the aim, catch crops—or cover crops with roots that drill well below the typical root zone—like sudangrass, ryegrass, and triticale, may be best. These roots uptake nitrogen, potassium, and phosphorus deep in the soil profile and biocycle, or bring them back to the surface, storing them temporarily in plant biomass. This is an especially effective strategy in California's wet winters, when nutrients are otherwise susceptible to leaching. Cover crops with a taproot (i.e. brassicas) can combat compaction and increase infiltration by creating large root channels underground (Williams & Weil 2004). This is especially useful in clayey soils and when transitioning to low or no-tillage systems (Williams & Weil 2004; Gruver et al. 2016). Cover crops that establish quickly and have a dense, shallow root system, like oats, white clover, and winter wheat are often used for weed suppression and to resist erosion (Montemurro et al. 2013).

Cover cropping involves the planting of an annual, perennial, or mix of species between cash crops.

Carbon

Models project that cover cropping in California's Central Valley has the potential to increase soil carbon by up to 90% (DeGryze et al. 2011). Cover cropping has also been well-documented in the field to increase soil carbon (Veneestra et al. 2007; Suddick et al. 2010; Aguilera et al. 2013). In a 5-year study in California vineyard systems, cover cropping increased soil organic carbon (SOC) by 40-50% (Steenwerth and Belina, 2008). At the long-term Sustainable Agriculture and Food Systems (SAFS) experiment in Davis, CA, 10 years of cover cropping increased soil carbon by 1.4 tons/acre compared to bare fallow (Poudel et al. 2002). Long-term experiments at Westside Research Station in Five Points, CA have also resulted in 15-20% higher soil organic matter (SOM) levels under cover crops relative to bare fallow (Veenstra 2007; Mitchell et al. 2015). This has been found under both conventionally tilled and no-till systems, although increases were greater when combining cover cropping with no-tillage (Mitchell et al. 2015). Two meta-analyses in Mediterranean systems also found significant increases in SOC (0.27-1.59 Mg C/ha/year) under cover-cropping relative to conventional fallow (Gonzalez-Sanchez et al. 2012; Aguilera et al. 2013). Conversely, increased fallow has been shown to decrease SOC in California (Janzen et al. 1998; Campbell et al. 2000; Seiter and Horwath, 2004; Sherrod et al. 2005).

Most research on cover cropping, however, has been conducted on soil samples taken to an average depth of 25.7 cm (Aguilera et al. 2013). A recent long-term study (Figure 1) examining the impact of 20 years of cover cropping (oats, vetch, fava beans, cowpea) to a depth of 2 meters in corn/tomato systems at the Russell Ranch Long-term Agricultural Research Station at UC Davis found that shallow sampling can misestimate impacts of management on soil C stocks (Tautges et al. 2019). Despite an increase in carbon in the surface 0-30 cm under both cover crop treatments, there was a loss of 13.4 Mg C/ha across the entire soil profile (0-200 cm) in treatments with cover crop plus mineral nitrogen (N). Cover crop plus composted manure (compost + CC), however, increased by 21.8 Mg C/ha (0-200 cm), suggesting either a synergistic effect between cover crops and compost or a priming effect brought on by limiting nutrient(s) in the cover crop plus mineral N system (Tautges et al. 2019). While this study highlights the importance of deep sampling for soil carbon accounting, cover cropping + mineral N was found to contribute to several other on-farm benefits including improved infiltration and a 30-40% reduction in nitrogen fertilizer requirement without a yield penalty for over 24 years, as compared to conventional management (Russell Ranch 2019).

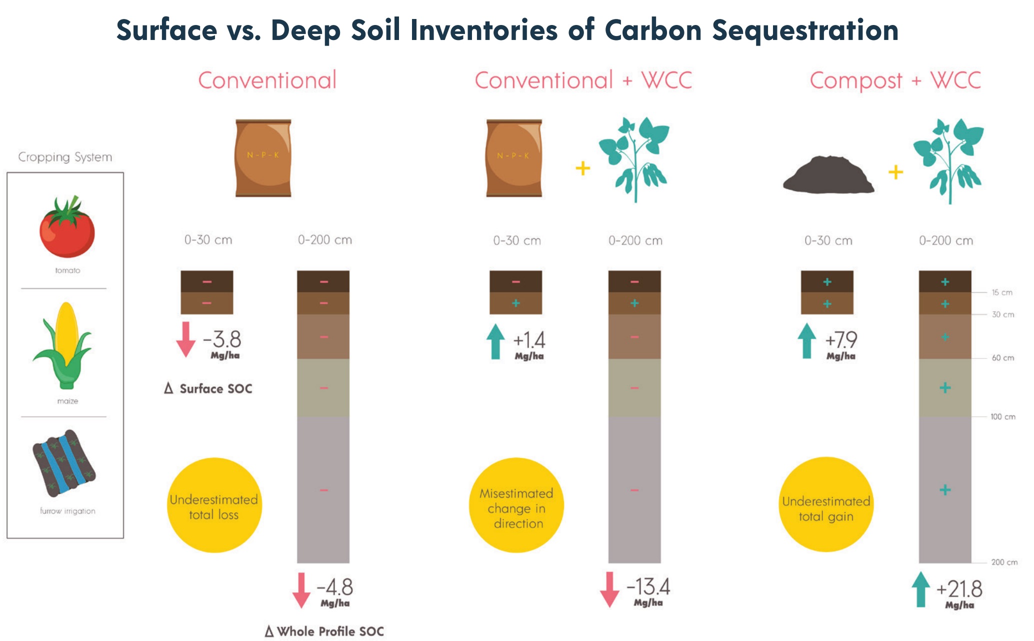


Figure 1. Long-term research in Davis, CA shows organic systems (compost + CC) sequester 3x more carbon to 2-meter depth, as surface 30cm alone; mineral fertilizer + CC loses large amounts of carbon below 30 cm (Figure reproduced from Tautges et al. 2019).

Water

Despite increases in SOM and other co-benefits, growers have concerns as to whether cover cropping will deplete winter water storage (DeVincentis et al. 2020). Past research at the Westside Research Station, in Five Points, CA, found average soil water contents from planting (October) to incorporation (March) decreased 7.9 cm (3.1 in) in barley + vetch, 7.4 cm (2.9 in) in barley, and 6.6 cm (2.6 in) in vetch, while fallow plots increased 9 cm over the same period (4 cm in the 3rd year, which was significantly drier) (Mitchell et al. 1999). In 2013 and 2014, after 15 years of cover cropping, soil moisture content to a depth of 90 cm was reduced by 5.3 and 0.7 cm, while fallow plots increased 4.8 and 0.4 cm (with 2014, again significantly drier). The concomitant use of residue preserving and reduced disturbance practices, such as conservation tillage, however, have been found to mitigate or even offset losses, reducing evapotranspiration by 4 inches (Unver and Vigil 1998; Klocke et al. 2009; van Donk et al. 2010; Mitchell et al. 2012). It has also been suggested that earlier termination of cover crops could mitigate water losses, as the major divergence in water storage between cover cropped and fallow fields occurred 100 days after planting and beyond, when the most vigorous plant growth occurs (Mitchell et al. 1999).

While there is still a need for more research, evidence across a range of climates, soil types, and management systems suggests that cover cropping can have a negligible impact on water storage across the state. A recent three-year study (see next page) spanning 10 field sites of specialty cropping systems (processing tomatoes and almonds) found insignificant differences in soil moisture content between cover-cropped and bare fallow fields 86% of the time, with differences primarily concentrated in the top 1.2 m of the soil profile and on farms with a long history of cover cropping (DeVincentis et al. 2022). Cover cropping at the Long-Term Research Station at Russell Ranch was found to increase soil moisture by 10% relative to bare fallow (Rath et al. 2021). Long-term cover cropping (oats and vetch) at the SAFS research site in Davis, CA, also showed increased soil water content to a depth of 1 m, as well as a 44% reduction in runoff, relative to bare fallow (Battany & Grismer, 2000; Joyce et al. 2002). Vineyards, which often occupy hilly landscapes, highly susceptible to erosion, have also been able to reduce runoff by 23-77% (and erosion by 50-75%) with cover cropping compared to bare fallow (Brennan & Boyd 2012).

California vineyard systems can also benefit from cover cropping.

By decreasing surface strength of the soil and bulk density (Folorunso et al. 1992; Keisling et al. 1994; Bauer and Busscher, 1996) and increasing aggregation and aggregate stability (Tisdall and Oades, 1982; Roberson et al. 1991; Watts and Dexter, 1997; Seiter and Horwath, 2004), cover cropping allows for greater infiltration and less runoff (Roberson et al. 1991; Battany & Grismer, 2000; Horwath et al. 2008; Ruiz-Colmenero et al. 2011). This may lead to enhanced water retention and soil moisture in the surface layer (Folorunso et al. 1992, Gulick et al. 1994; Colla et al. 2000), but in certain soil types, may require greater water application to ensure uniform wetting and/or to overcome increased percolation.

Ultimately, species selection, local climate and soil type all significantly impact the effects that cover crops exert on soil-water dynamics (and for that matter, soil carbon) (DeVincentis et al. 2022). Identifying the right species mix and management/termination strategies for a given region/microclimate/soil type will require local, collaborative,on-farm research and knowledge sharing (Chapagain et al. 2020).

Are Cover Crops Water-wise?

Questions

A common concern amongst growers who are new to cover cropping is whether the cover crop will require significantly more water for growth, thereby depleting valuable water required for the cash crop. Research by DeVincentis et al. (2021) addresses the following questions concerning winter cover crops and water use in California agriculture:

1. **How much water is lost from cover crops to evapotranspiration?**
2. **What is the net effect of cover crops on soil moisture?**

Testing

To provide answers, soil moisture and evapotranspiration (ET) were measured over a period of **3** years in **2** of the highest acreage crops in the state — processing tomatoes and almonds.

At each of the **8** farm sites and **2** UC Davis research farms throughout the Central Valley, bare soil was compared to either cover cropped almond orchards, resident vegetation in almond orchards, or cover crops in annual rotation fields (processing tomatoes).

Results

1. Differences in both ET losses and soil moisture between the cover crop and bare fields were found to be negligible – in 86% of measurements the differences were insignificant.
2. Additionally, there were some differences in the range of soil moisture content between systems; for example, the cover cropped tomato fields showed a larger range of soil moisture content (between 2-29%) compared to bare ground, whereas in cover cropped almonds the soil moisture was more consistent.
3. For both the cover crops and bare fields on the production farm sites there was never more than 15% soil moisture lost at the end of the winter cover crop season.

Conclusions

The study suggests that losses due to ET may require some extra irrigation when winter rainfall is low, amounting to a modest application of 1 inch of additional irrigation water. The authors note that proper timing of cover crop termination is key to avoid greater depletion of soil moisture — a small trade-off that may be worthwhile for many growers. **Overall, the study determined that cover crops do not require significantly more water to grow, and can offer a myriad of on-farm benefits.**

Nitrogen

Cover cropping also has a major impact on nitrogen (N) dynamics with important consequences for fertility management, nitrate leaching, and nitrous oxide emissions. From a fertility perspective, nitrogen-fixing legumes have been found to contribute anywhere from 20-55% of the nitrogen in their biomass to the subsequent cash crop, providing 90-200 kg nitrogen/ha in California agroecosystems (Malpassi et al. 2000; Poudel et al. 2001; Kramer et al. 2002). Providing nitrogen in an organic, slow release form, allows more N to be taken up and stored in microbial biomass, and leads to more tightly coupled nitrogen cycling (Bowles et al. 2015). Less residual nitrogen in the soil results in reduced potential for nitrate pollution in ground and surface waters, as well as lower overall emissions of nitrous oxide (NOx) (Jackson et al. 1993; Drinkwater et al. 1998; Poudel et al. 2001; Smukler et al. 2008; Bowles et al. 2015).

A potential trade-off of growing leguminous cover crops is that if nitrogen supply is not well synchronized with the subsequent crop's needs, the excess may be converted to NOx, increasing overall GHG emissions (Follet 2001; Watson et al., 2002; Sainju et al., 2007). While NOx emissions under a leguminous cover crop may be 60-80% lower in Mediterranean than other climatic regions (Aguilera et al. 2015), research in California has shown higher emissions with cover crops than without. At the Century Experiment, the conventional maize/tomato system with cover crop had an average hourly flux of 0.18 g NOx-N ha-1, while the conventional without cover crop emitted 0.07 g NOx-N ha-1 (Horwath & Burger 2013). Steenwerth & Belina found higher NOx emissions in California under a leguminous cover crop, but also observed a 2-4 times increase in overall N mineralization and microbial biomass nitrogen, indicating positive impacts on nitrogen cycling from an agronomic perspective. Furthermore, when cover cropping is paired with other conservation practices, such as sub-surface drip and reduced tillage, NOx emissions have been shown to decrease drastically (Kallenbach et al. 2010).

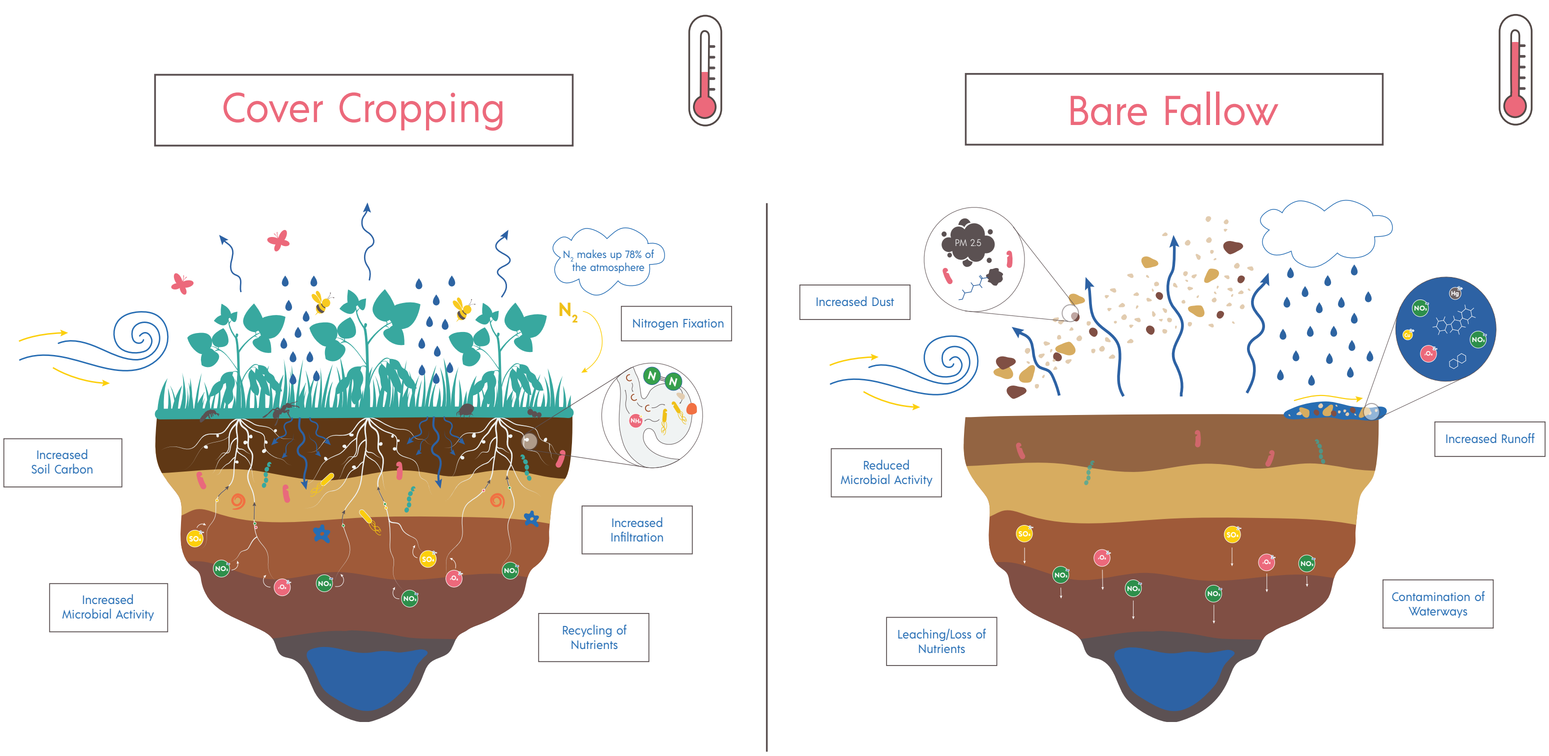


Figure 2. Conceptual diagram of soil health processes in a cover cropped field compared to a bare fallow field. Generally, cover cropping improves physical and biological soil processes, but impacts on water dynamics and nutrient availability vary by species, soil type, and climate. *Credit: Soil Life and CAFF*

While there is uncertainty as to the overall GHG budget of cover cropping, improvements in agronomic and environmental outcomes have been well-documented (Horwath et al. 2008; Mitchell et al. 2015). In California tomato systems, long-term research has shown cover crops to increase soil carbon, improve aggregation, and reduce runoff, while producing comparable (or slightly higher) yields to conventional management (Miyao and Robbins 2000; Poudel et al. 2001; Hartz et al. 2005). Furthermore, six on-farm trials in Oregon resulted in 11% greater corn yields and an additional $50/acre net profit following CC relative to bare fallow (Luna et al. 2012). Still the initial investment in seeds, the increased need for labor/equipment/technical assistance, the tight window for planting and terminating, challenges with residue management, and the fear of reduced winter water storage can deter growers from adopting these practices (Mitchell et al. 2015; DeVincentis, 2020). Such barriers to adoption stress the importance of local/regional collaborations and demonstrations to identify the appropriate cover crops for differing management systems and environmental conditions, including soil type and water availability.

No-till / Reduced Tillage

Since the advent of agriculture, it is estimated that 116-133 Pg C have been lost to the atmosphere, much of this attributed to the initial ploughing and conversion of native lands, as well as the continued intensification of tillage (Sanderman et al. 2017; Amundson & Biardeau 2018). **Tillage has a lasting impact on the biological and physical structure of the soil and often leaves carbon-rich topsoil vulnerable to wind and water erosion** (Lal 1993, Kladivko 2001). On a micro-scale, the disruption of soil aggregates releases carbon (Zakharova et al. 2014), simultaneously injecting a fresh supply of oxygen, driving a pulse of microbial activity and increasing decomposition of SOM (Calderon et al. 2000; Calderon and Jackson, 2002; Jackson et al. 2003). On a macro-scale, the physical action of the blades reduces invertebrate populations (Robertson et al. 1994; Errouissi et al. 2011) and can contribute to a loss of structure that subsequently impacts air, water, and nutrient flow and inevitably feeds back on the biology, impacting physiology and system-wide efficiencies (Kladivko 2001).

There are situations (i.e. clayey, poorly drained, or compacted soils), however, where the use of tillage may be necessary to support primary productivity, increasing both yields and overall carbon inputs to the system (Baker et al. 2007; Pittlekow et al. 2015). In these circumstances, an influx of oxygen after tillage may improve the environment for roots and microorganisms, such that primary productivity, carbon use efficiency, and microbial growth rates increase, ultimately leading to an increase in SOM. Oftentimes, however, the alleviation of compaction is short-lived and the multiple passes of heavy equipment can deteriorate soil structure, increase subsurface compaction and reduce aeration in the long-term (Alvarez & Steinbach 2009; Soane et al. 2012; Li et al. 2020). The impact of tillage ultimately depends on a variety of factors including climate, soil texture, soil moisture at time of tillage, depth and frequency of tillage, and quality/quantity of residues and other inputs (Ogle et al. 2019).

In California, intensive tillage is common practice to maintain beds, create seed beds, ensure evenness of water movement along furrows, loosen compaction, improve root penetration, and control weeds and pathogens (Minoshima et al. 2007).

Conservation tillage (CT), which includes no-till, strip till, ridge till, and mulch till, attempts to minimize the impact by reducing the number of tractor passes by 40% or more, or by retaining at least 30% of plant residues on the soil surface (Blevins et al. 1985; Mitchell et al. 2007).

While a substantial body of evidence in the Midwest and other regions indicates gains in SOM in the upper soil profile (0-30 cm) under conservation tillage, there is a lack of such data in California and the US Southwest

(Suddick et al. 2010, Nunes et al. 2020). Studies that were available showed little to no effect of conservation tillage on SOM in California (Buschiazzo et al., 1998, Veenstra et al. 2007, Geisseler and Horwath 2009). In barley/fallow and continuous barley cropping systems in Spain, Alvaro-Fuentes et al. (2009) found SOC to be 30% higher in no-till compared to conventional tillage, but results were limited to the surface (0-5 cm). Measuring SOC to a greater depth may be especially important in the context of tillage, as the practice redistributes soil C (Six et al. 2004; Baker et al. 2007). This can lead to observed increases in the surface that dissipate when considering greater depths (Six et al. 2004; Veenstra et al. 2007; Cai et al. 2022). A meta-analysis of Mediterranean agroecosystems (79 studies) found an 11.4% increase in SOC (0-34 cm) under no-tillage, or +0.44 Mg C/ha/yr and a 15% increase in SOC (0-27 cm) under reduced tillage, or +0.32 Mg C/ha/yr (Aguilera et al. 2013).

The greatest gains with conservation tillage may be achieved over the long-term and when implemented in conjunction with cover cropping and other OM inputs. Whereas 15 years of conservation tillage led to no significant change in SOC (0-15 cm) in tomato/corn systems of the San Joaquin Valley, conservation tillage plus cover cropping led to a doubling of SOC over the same time (Mitchell et al. 2017). Minoshima et al. 2007 similarly found higher soil C, microbial biomass C and fungal abundance at 0-5 cm in no-till cover crop systems in the Sacramento Valley. In vegetable cropping systems of semi-arid Spain, 20 years of no-till with cover crops resulted in 14% higher SOC at 0-10 cm as compared to standard and conservation tillage (Hernanz et al. 2002). Studies of conservation tillage plus cover cropping in Spanish vineyards similarly reported increases in surface SOC as well (Peregrina et al. 2010; Ruiz-Colmenero et al. 2013).

By reducing evaporation and increasing surface cover (increasing moisture and buffering temperature), conservation tillage in semi-arid environments could fuel microbial activity and lead to an increase in both CO_2 and N-based GHG emissions (Unger et al. 1997). For instance, a meta-analysis of conservation tillage in dry climates reported a 57% increase in nitrous oxide emissions over standard tillage, (Six et al. 2004; van Kessel et al. 2013). However, in California no-till systems maintained for over 10 years, nitrous oxide emissions were found to decrease by 27% compared to standard tillage (van Kessel et al. 2013). The influx of oxygen that accompanies tillage has also been reported to result in an increase in nitrate, which is highly susceptible to both leaching and increased nitrous oxide emissions (Jackson et al. 2003/4; Six et al. 2004).

Conservation tillage offers several co-benefits important to water dynamics and overall on-farm

After mowing, the cover crop residue is left on the surface as mulch to keep the soil cool during the summer.

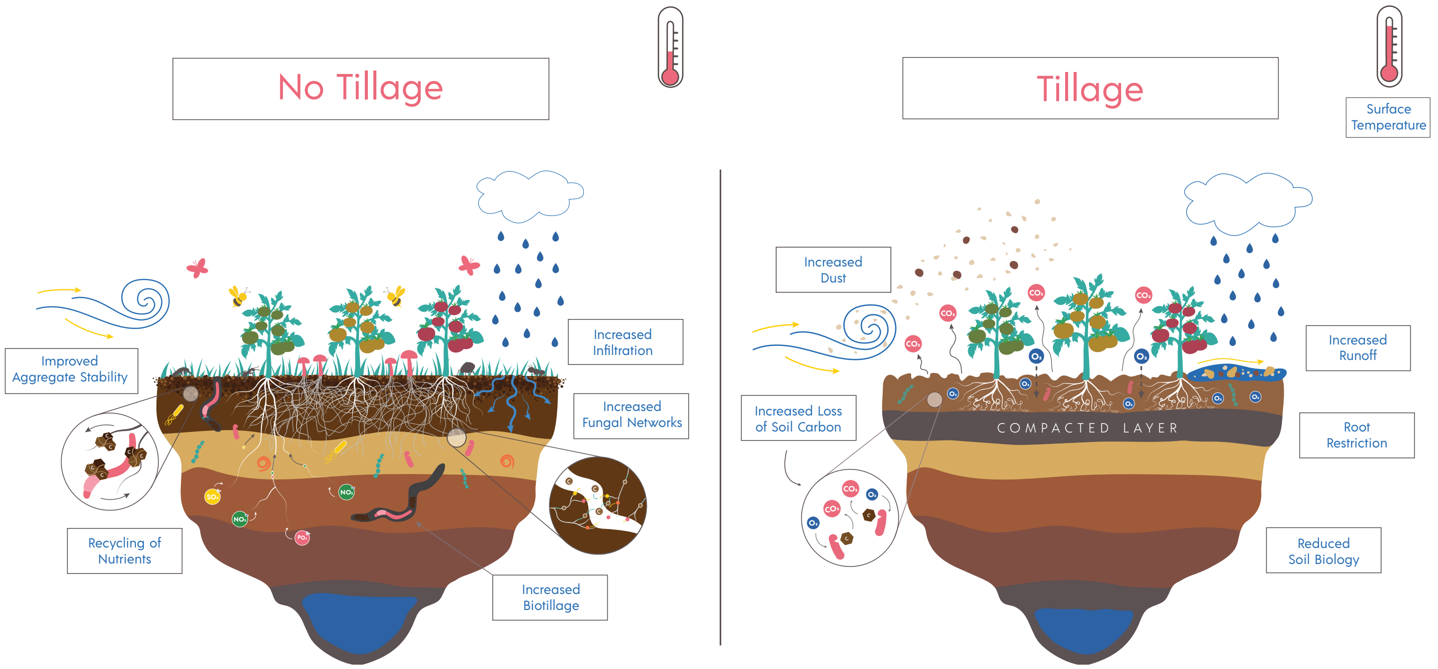


Figure 3. Conceptual diagram of soil health processes in a no-till field compared to a field under conventional tillage practices. Improved aggregate stability and structure, increased infiltration, and fungal biomass have been observed in no-till systems in semi-arid climates compared to conventionally tilled systems. Careful attention must be paid to nutrient dynamics/availability. *Credit: Soil Life and CAFF*

resilience. Long-term no-till wheat/corn systems in the San Joaquin Valley, for instance, were found to reduce soil evaporation rates by 10-12.5 cm (4-5 in)/year, increasing water retention in the surface foot by ~2.25 cm (0.9 in); which if implemented at scale could reduce irrigation by millions of acre-feet across the state (Klocke et al. 2009; Mitchell et al. 2012). Similarly, DeVita et al. 2007 (Spain) found higher soil moisture content and yields under no-till than standard tillage.

In California's annual cropping systems, tillage accounts for 18-25% of overall production costs. Conservation tillage offers potential savings of $100 to $150 per acre (Sutton et al. 2006; Mitchell et al. 2012), while no-till has been shown to reduce costs by $135 and $40 per acre in tomato and cotton, respectively; while producing similar or greater yields (Warnert 2012; Mitchell et al. 2022). By reducing tractor passes 41 to 53%, no-till also has the potential to reduce fuel usage by 48 to 62%, contributing to GHG emissions reductions of 0.25 to 0.5 Mg CO2e/ha/yr (Jackson et al. 2009). Research at the same site found an overall reduction in PM10 of up to 85% relative to standard tillage with 1/3 less dust generated under conservation tillage and ¾ less under conservation tillage plus cover crop (Baker et al. 2005; Mitchell et al. 2005; Madden et al. 2008).

From 2006 to 2010, the use of conservation tillage expanded from 3% to 47% of agricultural acreage statewide (Mitchell et al. 2010). No-till is still thought to be implemented on less than 2% of the acreage (Mitchell et al. 2009, Brennan and Boyd 2012). Best practices adapted to the diversity of specialty crops in California are still necessary to encourage further adoption. Crop residues left on the surface, for instance, can create unfavorable conditions for seed germination, while providing habitat for potential pests and pathogens (Jackson et al. 2003). However, combining no-till with sub-surface drip irrigation can help control weed germination by limiting surface water availability (Sutton et al. 2006) while maintaining or even increasing yields (Burger et al. 2012). Furthermore, conservation tillage can present challenges with weed control and may require drastic increases in herbicide usage, which not only increases input costs, but can contribute to herbicide resistant weeds. Roller crimpers hold promise as a termination strategy. They have been found to increase crop biomass and fruit yield, while reducing weed biomass in Italian zucchini systems, but are still in the discovery phase in California's diverse cropping systems (Ciaccia et al. 2015).

Small Scale No-till Organic Vegetable Production

While no-till research and implementation is gaining momentum in both organic and conventional grain production systems, vegetable cropping systems have remained more elusive (Mischler et al. 2010; Smith et al. 2011; Mirsky et al. 2012). This is particularly true for organic vegetable systems, in which tillage (rather than herbicide), is the primary tool for managing weeds and crop residue, as well as cover crops. Organic growers tend to place a high value on soil organic matter, soil biology, and overall soil health, which has spurred interest in best management practices for no-till, organic production systems in California (Mitchell 2020, *in communication*).

Small-scale, biointensive, no-till vegetable production has also gained popularity in recent years as a means of maximizing productivity while minimizing economic and environmental costs of organic farming (Fortier and Bilodeau, 2014; Lounsbury et al. 2018). This system of farming integrates multiple practices that increase ecosystem function including: heavy compost and/or mulch use, cover cropping, crop diversification, increased planting density/intercropping, minimization/elimination of fallow, immediate transplanting following harvest, and occultation which is the use of reusable black plastic tarps or landscape fabric to control weeds and accelerate decomposition (Lounsbury et al. 2018).

As this approach to management is a recently emerging trend, research into the trade-offs and/or synergies of integrating practices remains limited, especially in California. Similar management systems have been investigated by Cornell University's cooperative extension, although the unique environmental considerations (climate, pests/diseases, policy and access to markets, etc.) likely differ greatly from that of California agriculture (Grubinger 2007). In a replicated, two-year study, undisturbed, unmulched, and uncropped controls were compared to no-till vegetable systems receiving either 12" of mulch hay, 12" of mulch hay over newspaper, or 12" of mulch hay over cardboard. The hay mulch + cardboard resulted in significantly higher pH, SOM, and nutrient content than the control with the other two treatments fell in between (Grubinger 2007). The study also found buffered soil temperatures, higher moisture levels, and fewer weeds, without an impact on yield. Consumer demand for regenerative or "beyond organic" products, increased scrutiny over the environmental impact of agriculture, and the growth of policy initiatives around organic waste diversion and soil health management (much of which is included in biointensive systems) are contributing to growing adoption of these systems.

Small scale organic no-till vegetable systems have shown significant improvement in soil health indicators including carbon storage. *Photo by Elizabeth and Paul Kaiser*

An On-farm, Side by Side Comparison of Biointensive No-till Practices

Starting in 2017, Community Alliance with Family Farmers (CAFF) partnered with two small-scale, diversified vegetable farms in Northern California to better understand the effects of no-till as a management practice and as a farming system on soil health. Together, they addressed the following questions:

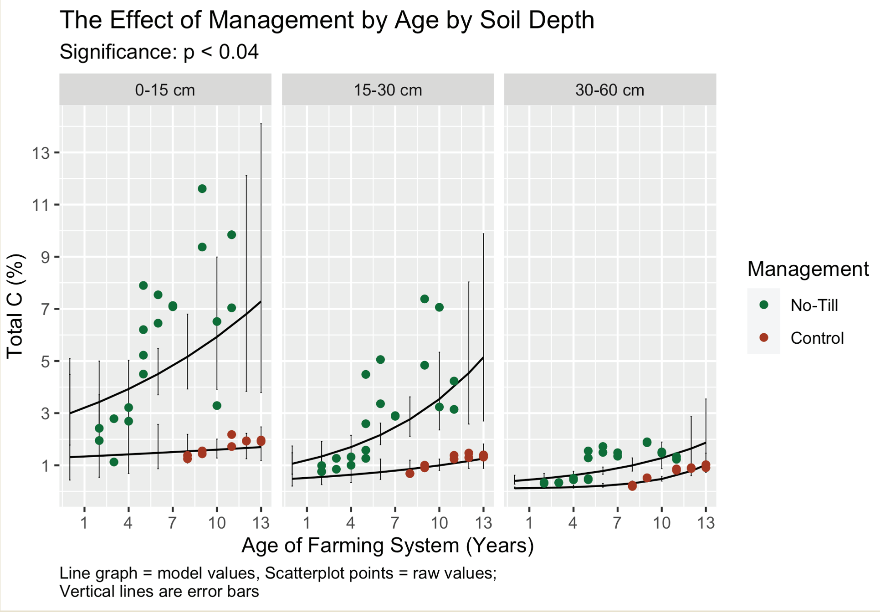
1. **How do no-till farming systems affect soil health?**
2. **How do these effects vary by soil depth and over time?**

At each partner farm, a side-by-side comparison trial was established, comparing a no-till plot to a farm-specific control plot. Annual soil samples were collected at three depths (0-15 cm, 15-30 cm, and 30-60 cm), and assessed for the following indicators of soil health: total carbon, labile carbon, total nitrogen, and total microbial biomass.

At both partner farms, all measured soil health indicators were significantly higher ($p < 0.05$) in the no-till plot than in the control plot[1]. These differences were all significant at the top depth, and some were also significant at the two subsequent depths. The notable accumulation of soil nutrients and microorganisms at the soil surface is typical in no-till farming systems, which is due to nutrient amendments (compost), plant residue and root exudates in the topsoil.

Total carbon, reported as a concentration, was determined to be higher in the no-till plots at every depth. Depending on the farm, at the soil surface the no-till plot had double to triple the amount of total carbon when compared to the control plot — a 112 and 286 percent increase, respectively. Significant differences in total carbon at the two subsequent depths were also found at one partner farm. This increase in subsurface soil carbon in the no-till plot may suggest that carbon is moving down the soil profile, a potential indicator of carbon sequestration.

Overall, the project found that not only are soil health indicators like total carbon affected by no-till management practices, but also by the length of time (or age) that they are implemented. For example, the project's 8-year-old no-till plot had more carbon than its 3-year-old no-till plot. This positive, significant relationship between age of the no-till system and total carbon accumulation is illustrated in the figure below, and suggests that a farmer using no-till as a management practice could anticipate increases in total carbon during the first 13 years of adopting the practice. More research and statistical models are needed to predict beyond this timeline.



The project's partner farmers reinforced the quantitative data with their experiences on the farm. One shared the following: **"The crew and I can feel the difference. When we fork carrots in our no-till fields, it feels like butter. In our newer plots, it's a workout. This is one of the many reasons why we continue down this path. We can finally grow carrots!"**

1 The management practice for the control plot differed at the two farms with one being a fallow field and the other a conventionally tilled field.

Compost, Mulching and Whole-Orchard Recycling

Compost and manures provide energy, carbon and nutrients to soil.

Compost and Manures

Compost and manures provide not only macronutrients (N, P, K), but all 14-17 plant essential nutrients, embedded in a matrix of carbon. This simultaneously provides the energy, carbon, and nutrients necessary to grow microbial bodies, which ultimately provide the feedstock for stable SOM (Liang et al. 2010; Miltner et al. 2012). Increased microbial growth/activity not only helps store carbon out of the atmosphere; it also ties up nutrients in their biomass or in SOM itself, creating a slow turnover nutrient pool and preserving important plant nutrients against leaching/loss (Bowles et al. 2015).

Composting organic residues prior to application allows the materials to reach sufficient temperatures to kill pathogens and weed seeds, while losing little material to volatilization (Moral et al. 2009). There is high variability in compost production (i.e. carbon to nitrogen ratio of inputs, duration of the process, temperature, moisture, pH, and oxygen content). This variation impacts the “quality” of the finished compost and thus dictates the amount to apply, method of application, time between applications, and the impact on SOM and nutrient cycling.

The use of organic amendments has been well documented to increase soil organic matter in California agroecosystems (Drinkwater et al. 1995, Poudel et al. 2002, Kong et al. 2005). A field survey of various cropping systems across six California counties found three times higher SOC (0-15 cm) with compost compared to controls (Brown & Cotton, 2011). **The most comprehensive, long-term study in California to date, the Century Experiment, similarly found that 19 years of compost + CC sequestered 21.8 Mg C/ha at 0-100 cm, whereas cover crop plus mineral fertilizer lost 13.4 Mg C/ha (Figure 1)**. Mineral fertilizer alone did not significantly change soil carbon (Tautges et al. 2019). In a meta-analysis of Mediterranean cropping systems, compost produced the largest increase in SOC (1.31 Mg C/ha/yr) over all other practices examined (cover crop, no-till, etc.) (Aguilera et al. 2013).

In the Century Experiment, compost + CC also led to 40% higher microbial biomass, greater aggregate stability and faster infiltration rates, as compared to conventional plots (Wolf et al. 2016). Tomato yields were not significantly different across systems, but were more stable (year over year) and resistant to external stressors (i.e. drought) in organic systems. Although tomato yields were 36% higher under compost+CC than conventional in years of adverse environmental conditions, maize yields in organic systems were 36% lower on average, less stable and less resistant under adverse conditions (Li et al. 2019). This suggests that yield stability and resilience may be crop specific in relation to inputs such as compost. Several others have reported on the ancillary benefits of compost in California, including increased microbial biomass, porosity, and water holding capacity, as well as reduced surface sealing, erosion, compaction, nitrate leaching, and weed/disease pressure (Fennimore and Jackson, 2003; Jackson et al. 2003; Lepsch et al. 2019; Hargreaves et al. 2008; Brown & Cotton, 2011; Martinez-Blanco et al. 2013).

While organic amendments contribute to systems-wide benefits, they must still be used judiciously, with close attention to quantities and ratios of key nutrients – both to protect crop yield/quality and prevent losses to the environment. Ratios of carbon to nitrogen to phosphorus to sulfur determine the availability of important plant nutrients and whether carbon will be stored as microbial biomass or lost as carbon dioxide, and ultimately, whether a net gain or net loss of SOM will result (Kirkby et al. 2016; Coonan et al. 2020). Carbon to nitrogen (C:N) ratios (Table 1) of composted amendments should remain below 20-25:1 to ensure sufficient nitrogen for microbes to build biomass *and* release excess into the soil. Little nitrogen will be mineralized and made available for plant uptake above this ratio (Graveur 2016). For compost with C:N ratios below or equal to 11 (high N), moist applications should range from 3- 5 tons/acre/year in annual crops and 2-4 tons/acre/year in tree crops. For C:N ratios above 11 (low N), 8 tons/acre/year should be applied in annual crops and 6-8 tons/acre/year for tree crops (Graveur 2016).

While the C:N rule of thumb often holds true, timing matters. If application does not align with crop demand, excess nitrogen in the system can contribute to nitrous oxide and nitrate losses, similar to mineral fertilizer. Tying up nitrogen in microbial biomass can be an effective means of protecting it against losses (Bowles et al. 2015). To avoid losses of nutrients and optimize nutrient availability from compost and crop demand, the 4Rs (right place, right rate, right time, and right source) can be applied (Figure 4). In a healthy, biologically active soil, soil microfauna (i.e. protozoa and nematodes) graze on bacteria and fungi, which contain more nitrogen than protozoa and nematodes need for their own growth/reproduction. As a result, microfauna release excess nitrogen into the soil, equivalent to about 30% of total mineralized N (Griffiths 1994).

Material	Carbon-to-Nitrogen
Whole Orchard Grindings	160:1
Rye Straw	82:1
Wheat Straw / Corn Stalks	80:1
Oat Straw	70:1
Brassica Residues	62:1
Nut Hulls	60:1
Horse Manure (sawdust bedding)	60:1
Corn Stover	57:1
Solid Cattle Manure (heavy bedding)	40:1
Rye Cover crop (anthesis)	37:1
Pea Straw	29:1
Horse Manure (straw bedding)	27:1
Rye Cover crop (vegetative)	26:1
Mature Alfalfa Hay	25:1
Solid Cattle Manure (light bedding)	20:1
Compost (green waste)	13:1-20:1
Legume Hay	17:1
Beef Manure	17:1
Liquid Dairy	15:1
Alfalfa	14:1
Vermicompost	13:1
Soil Organic Matter	10:1-12:1
Soil Microbes	8:1-12:1
Compost (manure)	6:1-8:1
Hairy Vetch Cover Crop	11:1
Poultry Manure (broilers)	10:1
Poultry Manure (layers)	5:1
Pelleted / Granular Organic Fertilizer	5:1-7:1
Liquid Fertilizers	5:1
Blood / Feather Meal	4:1

Table 1. Carbon-to-nitrogen ratios for commonly used organic materials (Wuest & Gollany 2012; NRCS 2011; Jahanzad et al. 2019; Lazicki et al. 2020).

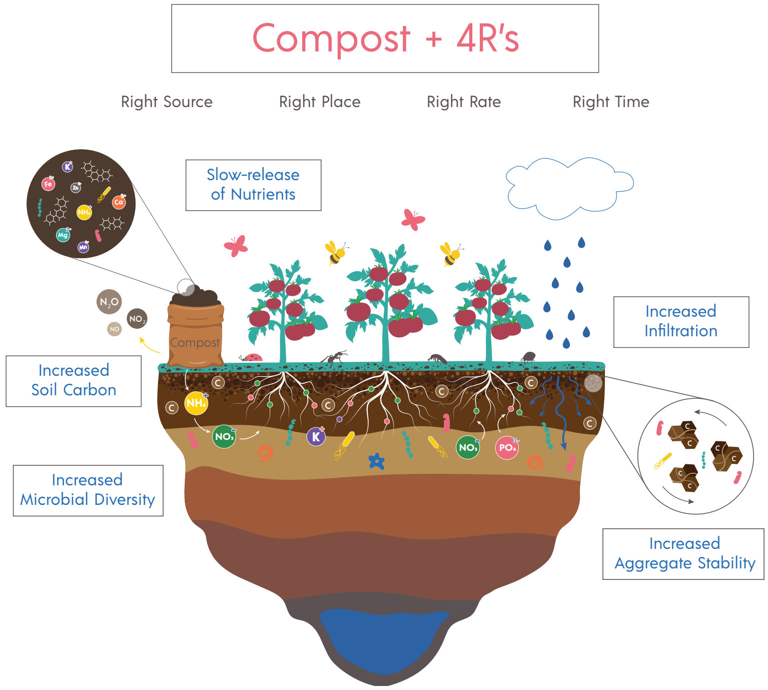


Figure 4. Conceptual diagram of soil health processes with proper compost application. When applied according to the 4Rs (right source, rate, time, and place), the benefits of compost application can be maximized. Over-application, however, can lead to nutrient losses and contamination of waterways. *Credit: Soil Life and CAFF*

Composts and manures are often high in phosphorus (P), relative to nitrogen, which can result in excess P application (Maltais-Landry 2015). Similarly, to excess N, excess P can lead to eutrophication and contamination of waterways. The feedstock and the way it is processed determines whether micronutrients, heavy metals, and/or organic pollutants are present/available (He et al., 2001). Although micronutrients are needed in much smaller quantities than macronutrients (N, P, K, S, Ca, Mg), they are vital to the optimum physiology, activity, and overall efficiency of microorganisms, and thus, may play a role in the accumulation of SOM. If any nutrient is deficient, soil organisms must do excess work to find it or make it available. While heavy metals can have a negative impact on microbial physiology, compost applications have been found to reduce heavy metal uptake in plants, as organic matter binds them up and reduces their availability (Rosen & Chen, 2014).

Compost has been found to significantly reduce nitrous oxide emissions relative to urea-based fertilizer (Alluvione et al. 2010). Compost in conjunction with feedlot manure has been found to mitigate emissions altogether (Dalal et al. 2010), while the combination of compost and ammonium-based fertilizer has been shown to increase nitrous oxide emissions (Zhu-Barker et al. 2013). Applications of raw manure and green wastes alone have also been found to increase nitrous oxide emissions when applied directly on the soil surface (Zhu-Barker et al. 2015), particularly when not timed with crop demand (Lazcano et al. 2016). The impact of composts and manures on nitrous oxide emissions has shown a wide range of results, due to variation in soil type, application rates, amendment quality/composition, etc. (Inubushi et al. 2000).

Growing awareness of the multiple benefits has led to a re-emergence of organic amendments in conventional systems (Hartz et al 2000). At 6 million tons/yr, however, there is still not enough compost in California to support large-scale adoption. Recently passed SB 1383 calls for the diversion of 75% of organic waste (14 million tons) from landfills by 2025, as well as the development of infrastructure to increase compost and mulch production statewide. While transporting compost carries a GHG footprint (especially when shipped wet), a recent analysis by Harrison et al. (2020) indicates that municipal compost can be transported up to 83 miles and still maintain its benefit as a GHG sink. The same analysis confirmed there are enough farms within an 83 mile radius of major California cities to receive all of the compost produced under SB 1383 (Harrison et al. 2020).

Mulches

Similar to compost and manure, the use of mulches provides a food and nutrient source to fuel soil biology. The use of mulch has been shown to increase earthworm populations, fungal biomass, and overall SOC content (Blanco-Canqui & Lal 207; Kahlon et al. 2013). Mulches also provide physical protection of the soil surface, preventing erosion and increasing infiltration, and minimize weed emergence (Pinamonti 1998; Varga and Májer 2004; Frederikson et al. 2011). By providing a layer of insulation, mulches help buffer against extreme shifts in soil temperature and moisture, creating a more favorable environment for soil organisms and their physiology (Pinamonti 1998; Wang et al. 2021). In semi-arid environments like California, surface coverage could, however, lead to increased microbial activity, due to increased soil moisture content (Unger et al. 1997). Considering the persistence of soil organic matter as an ecosystem property (Schmidt et al. 2011), the reduction in temperature may also slow microbial activity and decomposition and importantly, may decrease specific respiration rate (CO_2 respired per unit carbon consumed), which tends to be lowest in hot, dry climates (Doetterl et al. 2015; Luo et al. 2017).

While the advantages of wood mulches are well-known, it may be impractical to apply sufficient quantities to large agricultural fields. If appropriate equipment (e.g. seed drills) is available, cover-crops as natural mulches may be a more practical solution. In conservation tillage systems, innovative implements (i.e. roller-crimpers) are used to roll cover crops over to die as a mulch in place. Natural mulches have obtained comparable yields to standard winter-fallow when combined with tillage, but weed control is often insufficient with cover crop mulch alone (Herrero et al. 2001). Hairy vetch mulch has been shown to reduce weed emergence, soil temperatures, and water loss, while providing a slow-release fertilizer (Abdul-Baki and Teasdale 1994). In the Coachella and Imperial Valleys, where fields are typically fallowed during the hot, dry summers, cowpea is increasingly being planted (June and August). It can reach 2,500 pounds/acre by the time it is chopped into a mulch; controlling weeds, reducing parasitic nematodes, and increasing SOM, while providing comparable yields to bare fallow (Mitchell et al. 2004). Many questions remain about the impact on pest/disease pressure, fertility, and water availability.

The use of mulches provides a food and nutrient source to fuel soil biology.

Whole Orchard Recycling

In whole orchard recycling, orchard trees are chipped and then incorporated back into the soil to build soil health.
Photo by Amber Kerr, courtesy of UC SAREP

Over the next decade, it is estimated that nearly 30-40,000 acres of almond orchards will be removed annually from California orchards (Holtz et al. 2018; USDA NASS 2021). Historically, when tree prunings and/or whole trees needed removal, growers relied on open burning or the removal of materials to a biomass co-generation plant (Holtz 2017). Open burning (OB), however, is increasingly restricted, due to its contribution to methane emissions and poor air quality. While biomass co-generation can offset 20-60% of GHG emissions, California plants have either closed or drastically reduced the amount of materials they accept and the price they offer. In search of alternatives, growers and cooperative extension agents began investigating Whole Orchard Recycling (WOR), or the grinding of whole trees using a tub grinder or woodchipper, to be incorporated back into the soil, providing fertility for the next planting (Holtz 2014, 2016).

Research at the UC Kearney Agricultural Center comparing 3 years of open burning to WOR showed higher levels of soil nutrients, SOM, microbial biomass, beneficial nematodes, and fungal to bacterial biomass; with no significant effects on yield. After 8 years, WOR also produced bigger trunks and better yields (1,956-2,247 lbs/acre in WOR, relative to 1,539-1,872 lbs per acre on burned plots) (Holtz 2014, 2016). Compared to OB, WOR was found to increase carbon dioxide emissions, but simultaneously increased aggregation, soil moisture, and water-holding capacity and decreased nitrate leaching, compaction and bulk density (Holtz 2016; Jahanzad et al. 2022). WOR also showed less susceptibility to drought with higher leaf stem water potential and lower bud failure than OB after 100 days without water (Jahanzad et al. 2019).

With a C:N ratio of ~160:1, the breakdown/decomposition of wood grindings may immobilize important plant nutrients in microbial biomass, temporarily reducing nutrient availability (Jahanzad et al. 2019). Preliminary trials indicate that early spring applications of up to 8 oz of N per tree at a 15-15-15 rate can prevent nutrient deficiencies after WOR (Holtz and Columber 2019). Compared to OB, WOR showed higher nutrients in leaf petioles, but 50% less sodium in soil; reducing risks of salinization. This was attributed to sodium being bound up in the soil organic matter. At an application of 64 tons per acre, wood chips provided an influx of 396 pounds of N, 768 pounds of Ca, 256 pounds of K, and 64,000 pounds of C in a slow-release form (Holtz 2017). The speed of decomposition depends on wood chip size, amount of application, incorporation depth, and methods of pretreatment.

While WOR does require upfront investment, the lack of other viable options and the myriad of co-benefits increase the feasibility of adoption. WOR costs $125-$810/acre more than OB, while grinding and hauling incurs an additional $0-200/acre over traditional practices (Holtz et al. 2019). Given this high capital investment, county and state incentives programs for practices such as WOR have been established in an effort to reduce air pollution and economic barriers to adoption.

Perennial Crops

Perennial systems have deep roots that hold soil in place and fight erosion.

Perennial crops tend to store large amounts of C in their woody biomass and extensive root systems (Smart et al. 2005; Kroodsma and Field, 2006; Williams et al. 2011); can reduce carbon dioxide and nitrous oxide emissions; and often require less fossil fuel to cultivate, relative to annuals. Management of perennial crops also requires far less soil disturbance (and thus, oxidation of SOM) than annual crops, as there is no need to remove plants and prepare beds year after year (Kroodsma and Field, 2006). Rather, perennial plants have deep and extensive root systems that hold soil in place and fight against erosion, feed soil microbes year-round, and contribute to improved structure and aggregation (Smart et al. 2005).

Perennial systems also lend themselves quite readily to cover cropping in between rows and integration of livestock as a possible means of terminating cover crops (Brodt et al. 2019). The integration of livestock can reduce the need for machinery and associated fuel costs and compaction risks (Ryschawy et al. 2021). Perennial systems require large amounts of water, which may become increasingly scarce under future scenarios (Pathak et al. 2018). However, trees and other woody perennials may also be good candidates for deficit irrigation. For instance, they have deep roots that can bring water from deep below ground to surface layers; alternate wetting and drying cycles can induce earlier budding and dormancy; some species have the ability to reduce transpiration when water is limited; and the overstory provides ground cover during hot/dry Mediterranean summers that can increase water use efficiency and reduce evapotranspiration (Brodt et al. 2019).

A recent 13-year simulation in perennial systems (alfalfa, almonds, grapes, pistachios, and walnuts) across California found that delaying the first irrigation of the season can encourage greater root exploration and interception of water (1 vs. 0.5m rooting depth), while irrigating less frequently and more deeply (50% vs. 30% allowable depletion) can reduce overall evaporative loss (Devine and O'Geen 2019). Combining these approaches could save 30km3 of surface and groundwater use, enough to fill Shasta Lake (California's largest reservoir), more than 6 times (Devine and O'Geen 2019). Calvo et al. 2022 found high density plantings and deficit irrigation in semi-arid walnut orchards could also reduce evapotranspiration with little effect on production. Increasingly, growers are using pressure bombs or chambers to achieve benefits of deficit irrigation, while tracking plant stress to determine when to irrigate (Shackel 2011).

Farmscaping

The myriad of benefits farmscaping offers to soil, coupled with the broader ecosystem benefits of reduced nonpoint water pollution and the provision of habitat, demonstrate synergies between climate mitigation/adaptation and conservation.

There are a host of on-farm practices, known collectively as farmscaping, that can be implemented on field edges and other non-production areas to further improve climate mitigation, resilience, and adaptability. Farmscaping involves the incorporation of biodiversity with perennial elements, such as tree crops and vines, hedgerows, riparian buffer zones, vegetative filter strips, and tailwater ponds. These features improve the functionality of the landscape and enhance its ability to provide ecosystem services. Farmscaping has been found to increase carbon sequestration, infiltration rates, and biodiversity, while reducing GHG emissions, runoff, nitrate leaching, and pest pressures (Vickery et al. 2002; Kremen et al. 2004; Young-Matthews 2010; Smukler et al. 2012; Morandin 2014).

Hedgerows provide a physical buffer while building soil health and on-farm biodiversity.

Hedgerows, Windbreaks and Filter Strips

Hedgerows are lines or groupings of dense vegetation (trees, shrubs, forbs, grasses, rushes, and/or sedges) planted along roadways, fences, and other field edges. Hedgerows provide a physical buffer zone that can intercept sediment, nutrients, and contaminants (Ghazavi et al. 2008; Long & Anderson 2010), create on-farm microclimates and shade for on-farm labor (Sanchez et al. 2010), and provide wildlife habitat for pollinators and pest predators (Morandin et al. 2011; Ponisio et al. 2015). Hedgerows used to control or redirect wind are referred to as windbreaks and consist of taller trees and shrubs planted in single or multiple rows, reducing wind intensity and protecting plants from damage and airborne dust (Bentrup 2008).

Hedgerows contribute to reduction in the GHG footprint on farm via reduced carbon dioxide and nitrous oxide emissions and increased storage of C both aboveground (in woody biomass) and belowground (in roots and SOM) (Falloon et al. 2004; Follain et al. 2007). A recent study on 21 farms across Yolo County found that maintaining hedgerows along field edges (10+ years), increased SOC an average of 38.3 Mg/ha (to a depth of 1m), relative to the adjacent cultivated fields (Chiartas 2022, in review).

Hedgerows satisfy many of the key goals of soil health management, including continuous ground cover and roots in the ground, reduced disturbance (tillage), and increased overall diversity on-farm (Long & Anderson 2010; Heath et al. 2017). By providing additional ground cover, hedgerows have been found to reduce weed pressure and buffer against erosion and runoff (Long et al. 2010). By maintaining roots in the ground, they have been shown to increase infiltration, increasing soil water storage on-farm and improving water quality off-farm (Marshall and Moonen 2002; Caubel et al. 2003; Long et al. 2010). Hedgerows also serve as dispersal corridors for wildlife, providing year-round food, habitat, and protection (Marshall and Moonen 2002; Ouin and Burel 2002; Long 2010), which has been shown to increase pollination and biocontrol of pests in nearby cropping systems (Brodt et al. 2010; Long 2010; Morandin et al. 2011; Ponisio et al. 2015).

A two-year study in the Sacramento Valley showed hedgerows attracted more beneficial insects relative to pest insects, whereas weedy areas at field edges attracted more pests (Morandin et al. 2011). Despite growers' concerns that hedgerows may detract bees from nearby crops, hedgerows have been shown to increase the presence of native bees in production areas (Morandin 2013). Similarly, farms with hedgerows along field edges in the Central Valley harbored 3-6x the total abundance of birds, relative to bare or weedy margins in the same region (Heath et al. 2017). Hedgerows provide birds with resources and protection from predators while perching, nesting, and foraging (Vickery et al. 2004). Together, the provision of habitat and resources for beneficial species helps promote additional ecosystem function, moving the agroecosystem toward self-sustainability.

Hedgerows attract pollinators and other beneficial insects.

Planting a diversity of hedgerow and windbreak species can provide successive, overlapping bloom periods, ensuring consistent pollen and nectar for beneficial insects (Bugg et al. 2998; Long et al. 1998). There is growing interest in hedgerows and windbreaks that produce a cultivable crop and contribute additional income on farms. Potential candidates well-adapted to California include citrus, pomegranate, persimmons, mulberries, pineapple guava, elderberry, nut trees, vine crops, perennial medicinal/culinary herbs, and traditional indigenous food plants (Brodt et al. 2019). A recent "costs and returns" study in the Sacramento Valley found that cultivating elderberry in a 1,000 ft. multispecies hedgerow could generate $2,700-$4,800 (after harvest and de-stemming costs) by the 2nd year of planting with yields and overall revenues expected to increase as the shrubs further establish themselves (Brodt et al. 2020).

Vegetative filter strips consist predominantly of low-lying herbaceous plants, situated along waterways. Planting perennial vegetation along waterways can increase SOM and reduce overall GHG emissions (Hill 1996; Rowe et al. 2005). Filter strips are intended to slow surface water runoff, intercepting contaminants and pathogens, and controlling soil erosion (Tate et al. 2006). Filter strips have been found to trap 75-100% of sediment, 50-80% of nutrients, and 44-100% of the herbicide atrazine in surface water; effectively improving water quality downstream (Grismer et al. 2006). They are also capable of removing pesticides, but this ability is highly variable depending on the chemical composition of the pesticide and the design and management of the filter strip (Grismer et al. 2006).

Grassed waterways and tailwater ponds similarly help remove sediments, nutrients, and contaminants from agricultural runoff, improving overall water quality (Blanco-Canqui et al. 2004; O'Geen et al. 2007; Smukler et al. 2010). Tailwater ponds have even been found to reduce nitrate in groundwater by 97% (Smukler et al. 2012). While initial and on-going maintenance costs can be prohibitive, tailwater ponds reduce dependency on external water sources, building resilience on-farm.

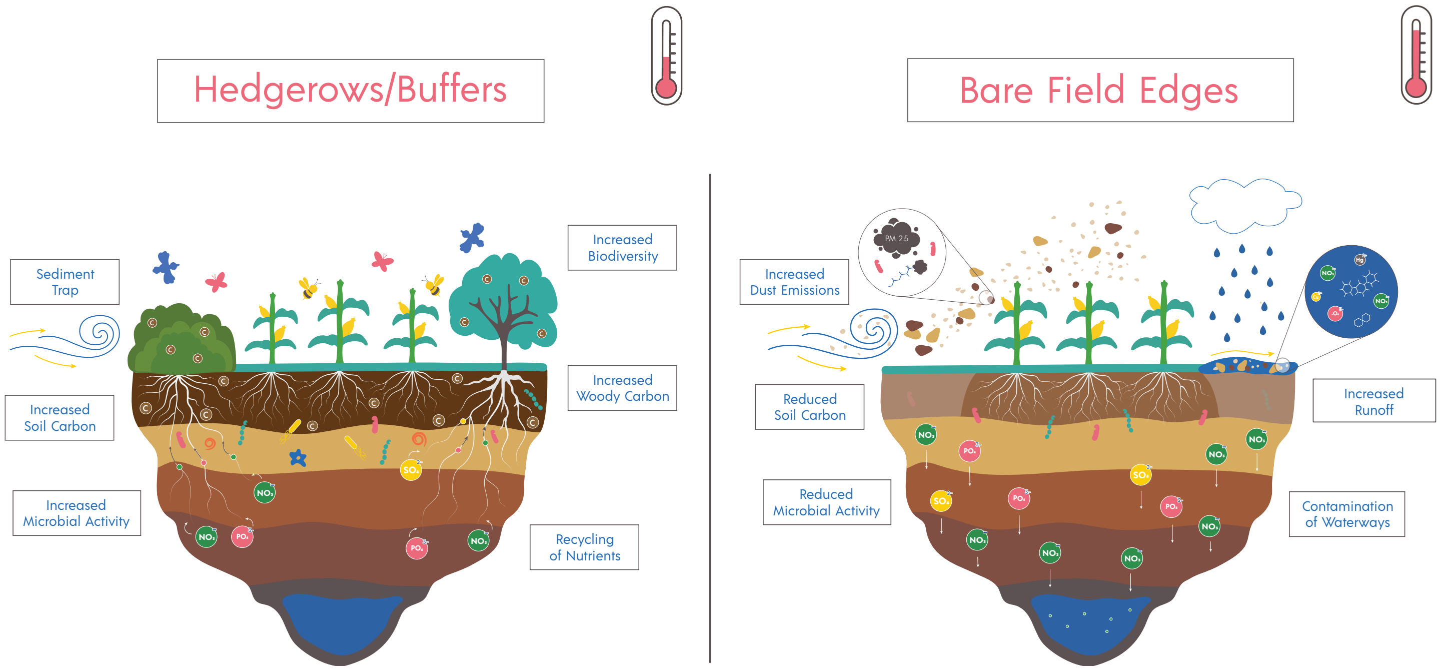


Figure 5. Conceptual diagram of soil health processes observed in field edge plantings compared to bare field edges.
Credit: Soil Life and CAFF

Growers perceive several barriers to establishing these permanent plantings. By the same principle that additional vegetation provides safe haven for beneficial insects and wildlife, they may harbor pests (i.e. rodents) and allow for the spread of weeds into cultivated areas. Growers are further deterred by the additional time to manage, cost to implement, and removal of valuable land from production (Earnshaw 2004; Brodt et al. 2019). However, a recent California study showed that a typical 300-meter hedgerow planting costs approximately $4,000 and takes only 7 years to break even, when factoring in the value of reduced insecticide use and increased pollination (Morandin et al. 2016). Marginal lands that do not interfere in production can be targeted (Brodt et al. 2019).

Riparian Restoration

Restoration of riparian corridors involves the planting of vegetation adapted to wet environments, along river margins, creeks and banks. A study in Yolo County found these corridors to harbor twice the total carbon stocks of hedgerows and three times that of crop fields, accounting for approximately 16% of the total carbon storage on-farm (140 Mg C/ha), but only ~6% of the total farm area (Smukler et al. 2010). Riparian corridors contribute neither to an increase nor a decrease in overall GHG emissions (Smukler et al. 2010). They have been found to mitigate the excess nitrogen and phosphorus loading on-farm via plant interception; removing up to 85% of water contaminants overall and reducing the load of nitrate in streams and groundwater by 28-42% (Bedard-Haughn et al. 2004; Zhang et al. 2009).

Riparian corridors mitigate excess nitrogen and phosphorus.

Integrated Crop-Livestock Systems

Integrated crop-livestock systems achieve climate-smart goals, building resilience by mimicking natural ecosystems and providing multiple income streams.

The increased industrialization and specialization of farms in the 1940's encouraged the decoupling of crop and livestock systems (Dimitri et al. 2005; Sulc and Tracy 2007). Removing animals from the landscape leads to a further decoupling of nutrient cycles and loss of ecosystem services (Lemaire et al. 2014; Brewer and Gaudin 2020). In an effort to close loops, build resilience, and reduce environmental pollution associated with industrial agriculture, some farms have begun reintegrating crop and livestock systems (Martin et al. 2016; Garrett et al. 2017; Ryschawy et al. 2021).

Integrated crop-livestock systems (ICLS) offer promise in achieving the climate-smart goals of agronomic, economic, and environmental sustainability (Lin, 2011; Franzlubbers 2011; Sousanna & Lemaire 2014). These systems rely less on external inputs, provide diversified income streams, and promote improvements in soil health; reducing vulnerability to climate/market fluctuations (Sulc and Tracy 2007; Hendrickson et al 2008; Garrett et al. 2017; Ryschawy et al. 2012). Rather than relying on imported feed, which is subject to cost fluctuations, ICLS practitioners grow their own feed on-farm in the form of cover crops, residues, and weeds. The integration of livestock into cropland allows land managers to use their own or contracted animals to manage residues, which can result in reducing or eliminating reliance on heavy machinery, fuel, and manual labor (Ryschawy et al. 2012). Rather than rely on fertility from off-site, ICLS recycles organic material through the gut of a livestock animal, returning it directly to the soil in the form of urine and manure (Brewer and Gaudin 2020).

Using animals in residue and fertility management can lead to fewer passes, less compaction and a reduction in GHG emissions from fossil fuel use; without negatively impacting crop quality or yield (Meadows 2008; Buller et al. 2015; Mckenzie et al. 2016; Garrett et al. 2017). Grazing alfalfa with sheep in the fall in the Sacramento Valley, for example, resulted in no significant differences in yield or bulk density/ compaction, relative to ungrazed alfalfa (Pelton 1988). Integrating animals is also thought to reduce the use of pesticides and herbicides (Meadows 2008; Franzlubbers et al. 2011; Niles et al. 2018).

While little research has been conducted in California, it has been noted that ICLS has strong potential to increase SOC in semi-arid environments by increasing overall net primary productivity and belowground carbon inputs, improving nutrient cycling, and promoting biological activity and ecosystem services (Brewer and Gaudin 2020). The integration of crops and livestock has been found to significantly increase soil carbon in other regions, especially when combined with reduced or no-tillage (Ernst & Siri-Prieto, 2009; Carvalho et al. 2010; Gamble et al. 2014; Salton et al. 2014; Silva et al. 2014; Garrett et al. 2017).

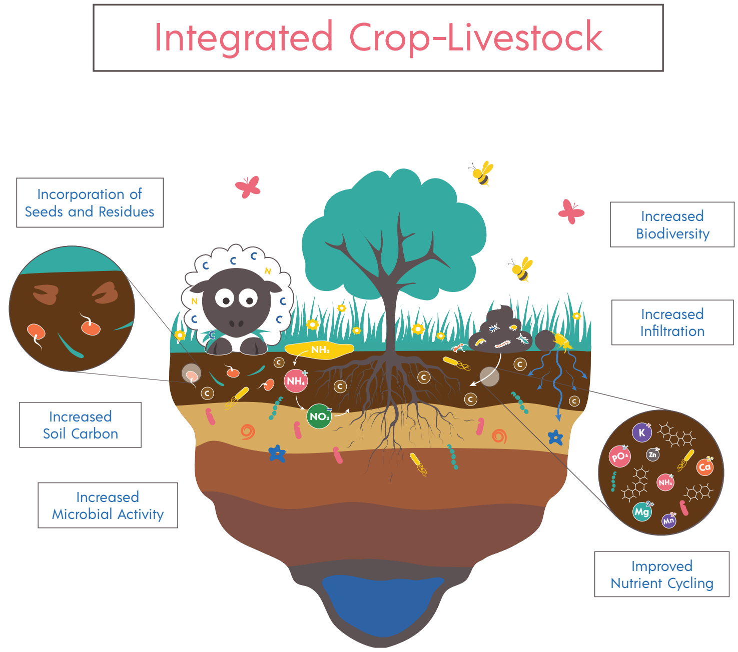


Figure 6. Conceptual diagram of integrating livestock into cropping systems. Potential benefits range from increased soil carbon content to increased agroecosystem biodiversity. *Credit: Soil Life and CAFF*

ICLS has also been studied extensively for the last 20 years in cotton systems of the Southern High Plains (SHP) in Texas. The SHP receives much of its rain during summer, rather than winter months, but, similar to California, is characterized by a semi-arid environment with low annual rainfall. Over the 20-year trial, researchers have observed a 25% reduction in water use, 40% reduction in N fertilizer, and 63% reduction in erosion potential (7 Mg/ha/yr relative to 19 Mg/ha/yr in continuous cotton) (Acosta-Martinez et al. 2004). Concurrently, soil organic carbon has increased by 22% with a 112% increase in protected (intra-aggregate) pools; a 6-fold increase in aggregate stability; greater fungal diversity; and increased enzyme activity (an indicator of healthy nutrient cycling) (Acosta-Martinez et al. 2010; Allen et al. 2012; Fultz et al. 2013).

These findings are indications of a healthy, active microbial community and thus, a greater likelihood of long-term carbon sequestration (Acosta-Martinez et al. 2004). The increased abundance of fungal symbionts known as arbuscular mycorrhizal fungi (AMF) is also a strong indicator of long-term carbon storage, as well as enhanced resilience. AMF extend the reach of plant roots up to 1,000 times, improving access to water, phosphorus, and other nutrients (Rillig, 2004; Allen et al. 2012). At the plant scale, animal grazing stimulates root production, increasing belowground carbon inputs (Souza et al. 2009; Brewer & Gaudin 2020). It has been recognized that root carbon may contribute more to stable soil organic carbon than residue carbon (Rasse et al. 2005; Schmidt et al. 2011; Rumpel et al. 2015) as it enters directly into the microbially active root zone, has a shorter distance to travel to mineral surfaces, and is introduced in frequent, low volume inputs, rather than few, high-volume pulses (Sokol & Bradford 2018). Furthermore, the subsurface soil may provide more opportunity in California for stabilizing soil carbon, as it often exhibits greater concentrations of clay surfaces and reduced temperatures (Albaladejo et al. 2013; Garcia-Franco et al. 2018). ICLS has also been found to promote abundance and diversity of soil fauna, improving overall ecosystem functions, including decomposition of organic materials, nutrient cycling, and enzyme activity (Acosta-Martinez et al. 2004).

A recent global meta-analysis found that continuous grazing can contribute to rapid declines in SOC with greater losses in dry, semi-arid grasslands (-16%) than in wetter climates (-8%) (Dlamini et al. 2016). Whereas climate-smart grazing practices such as rotational and other precision grazing approaches have been found to increase SOC and reduce topsoil erosion in semi-arid environments (Sanjari et al. 2008; Mcsherry and Ritchie, 2013; Teague et al. 2015; Waters et al. 2017). Relative to continuous grazing, rotational grazing practices tend to improve plant diversity and net primary productivity, even in low precipitation years (Bakoglu et al., 2009; Pineiro et al., 2010; Abdalla et al., 2018). Increases in productivity increase total carbon inputs to the system and have been found to contribute to SOC accumulation in semi-arid grasslands, rangelands, and croplands alike (Briske et al. 2011; Hoyle et al. 2013).

In an inductive analysis of sheep-viticulture systems of California, growers reported 2-4 fewer mowing passes, supporting labor and fuel savings of about $87-$174/acre (Ryschawy et al. 2019). ICLS has also been found to reduce input costs via lower water, nutrient, and energy use relative to monocropping (Allen et al. 2005; Allen et al. 2012; Acosta-Martinez et al. 2010). Conversely, the cost of hiring a contract grazier to mow a cover crop with sheep in California generally ranges from $80-$120/acre. These systems have been found to produce higher yields (up to 60% increase), reduce input costs, and maintain higher profits overall (Hoshide et al. 2005; Asai et al. 2018). However, implementation of ICLS will vary among different farms and requires careful attention to detail; expertise in both animal and crop management; and the ability to deal with complex interactions between animals, forage species, soil, and the microbial communities they support (Garrett et al. 2017; Niles et al. 2018; Brewer and Gaudin 2020). Growers' unique understanding of their system must be leveraged to determine crop-livestock rotations, stocking density, frequency of grazing, and duration of rest period, while ensuring compliance with food safety regulations. More research is needed to better understand crop-livestock pathogen dynamics, but recent research indicates these systems can produce food safe for human consumption (Patterson et al. 2018).

Example of an integrated sheep vineyard system with protective covers around the young vines. *Photo by Kelly Mulville*

Research Site	ΔSOC (Mg C ha^{-1} yr^{-1})	Depth (cm)	Time (years)
Long-Term Research Agricultural Sustainability			
Standard Till + Organic	1.32	15	10
Standard Till + CC	0.32	15	10
Standard Tillage	0.10	15	10
Conservation Till + Organic	1.28	15	10
Conservation Till + CC	0.32	15	10
Conservation Tillage	0.05	15	10
Century Experiment			
Organic (CC + Compost)	1.14	200	19
Conventional + CC	-0.71	200	19
Conventional	-0.25	200	19
Sustainable Agriculture Farming Systems			
CC	1.00	15	12
Crop Rotation (2 year)	0.44	15	12
Crop Rotation (4 year)	0.41	15	12
West Side Research and Extension Center			
NT + CC	1.19	30	20
ST + CC	0.84	30	20
NT	0.67	30	20
ST	0.73	30	20
Kearney Research and Extension Center			
Whole Orchard Recycling	0.58	15	9
Regional Surveys			
Hedgerow Plantings	2.24	100	17

Table 2. Change in soil organic carbon stocks under climate-smart management practices at long-term research sites across California (Suddick et al. 2010; Tautges et al. 2019; Jahanzad et al. 2019; Chiartas et al. 2022, *in review*).

Improving Efficiencies

Three-quarters of California cropland is irrigated, allowing a level of productivity that would otherwise not be possible under California's hot, dry summers (Johnston 2004; Hanak et al. 2019).

Irrigation Management

Irrigation has enabled a shift from drought-tolerant crops like barley and wheat towards more water-intensive crops like lettuce, corn, stone fruit, nuts, and rice (Johnson & Cody 2015). While these high-value crops contribute greatly to California's thriving agricultural industry, they are also heavy water users, increasing the overall GHG footprint (via pumping) and dependence on external water sources (i.e. groundwater and surface delivery) for crop production (Johnson & Cody 2015).

Reliance on external sources has significantly strained California's water system, particularly at times when surface water is limited and farmers must turn to groundwater to meet their irrigation needs. Monitoring of Sacramento and San Joaquin River Basins from 2003 to 2010 indicated extreme levels of groundwater depletion (31-89 mm yr^{-1}) as a result of this shift (Famiglietti et al. 2011; Scanlon et al. 2012). The subsequent drought from 2012 to 2016 resulted in pumping of over $40km^3$ of groundwater from the Central Valley (OEHHA 2018). Significant levels of land subsidence occurred, largely in areas with the highest water demand and greatest concentration of groundwater wells (Jeanne et al. 2019) and sinking up to 28 ft in parts of the San Joaquin Valley (Sneed et al. 2015). This ultimately led to the Sustainable Groundwater Management Act (SGMA) to regulate groundwater pumping (OEHHA 2018).

Pumping and pressurizing water carries an energy cost, contributing anywhere from 15-60% (average 42%) of total on-farm GHG emissions (Shaffer & Thompson, 2015) and a total of 10 billion kilowatt hours (kWh) of electricity annually (Marks et al. 2013; Water in the West, 2013). Given greater climate variability and water scarcity, growing urban pressure on water supplies, and policy GHG reduction goals, there is increasing interest in irrigation systems that improve water use efficiency and conserve water.

The most practiced methods in California have long been furrow and flood irrigation, where runoff, evaporative losses, and deep percolation contribute to low water-use efficiency. While deep percolation can replenish groundwater supplies, it can also increase leaching of nitrates and other agricultural inputs that are better kept out of the groundwater (Sharma et al 2012;

The most practiced methods in California have long been furrow and flood irrigation.

Grinshpan et al. 2021). Flood irrigation has both the highest soil CH_4 emissions and highest global warming potential (GWP - CO_2, CH_4, and N_2O) overall (Sapkota et al. 2020), however modifications to this method demonstrate improved efficiencies. In California rice systems, the practice of alternate wetting and drying reduced GWP by 57-74% over constant flooding, while simultaneously reducing arsenic concentrations (59-65%) and N fertilizer use, and maintaining or increasing yields (LaHue et al. 2018). In California tomato systems, alternate furrow irrigation reduced water use by up to 25%, while maintaining yield and fruit quality (Barrios-Masias & Jackson 2016). As the root zone dries out partially, the plant responds by reducing stomatal conductance, to reduce transpiration, using less water without reducing carbon assimilations (i.e. biomass/total residue) (Barrios-Masias & Jackson 2016).

Still, the need for more efficient systems in California has driven the demand for micro-irrigation systems (micro sprinklers, surface drip, subsurface drip), now implemented on over 40% of California cropland (DWR 2013). Micro-irrigation technologies, however, require significant investment and labor to maintain. Surface drip involves a pressurized tubing system with regularly occurring emitters, running along the soil surface. Subsurface drip (SSDI) is similar but buried at 10-18" deep. **It has been estimated that drip irrigation can achieve water use efficiencies of 90%, compared to 60-85% with flood/furrow irrigation** (Salas et al. 2006).

The need for more efficient systems in California has driven the demand for micro-irrigation systems.

Subsurface drip irrigation (SSDI) can significantly reduce water use and GHG emissions over flood and furrow irrigation (up to 40-50%) (Hartz and Bottoms, 2009; Kallenbach et al. 2010, Kennedy et al. 2013; Zhang et al. 2016), even when considering the carbon footprint associated with pressurization of water. SSDI has been found to reduce nitrous oxide emissions by 25% compared to furrow irrigation in leguminous, cover-cropped tomato systems (Kallenbach et al. 2010). Recent simulations estimate drip systems could reduce N_2O emissions up to 55-67% compared to surface applications (Deng et al. 2018). This is attributed to fertigation (or the soluble delivery of nutrients directly to the root zone via irrigation water), as the surface remains dry and low in nitrogen, while the root zone remains consistently moist, providing nitrogen at the "right place" and the "right time." Diffusion (or movement) of N_2O is reduced in moist conditions, providing an opportunity for it to be consumed by microorganisms and converted back to N_2 gas before reaching the atmosphere (Chapuis-Lardy et al. 2007; Yang et al. 2011). SSDI has also been found to maintain or increase yield, lower weed pressure by up to 95% (Sutton et al. 2006; Horwath et al. 2008; Ayars et al. 2015; Mitchell et al. 2015; Schmidt et al. 2018), and reduce overall susceptibility to disease (Goldhamer et al. 1997; Subbarao et al. 1997; Xiao et al. 1998).

Frequent, small applications of water to the root zone, as is common in SSDI, may maximize water use efficiency, but also may create unintended negative

Micro sprinklers are often used in California orchards.

consequences for overall soil health and on-farm resilience (Stork et al. 2003; Schmidt et al. 2018). By providing water and nutrients directly to the root zone, plants may be discouraged from investing heavily in roots to scavenge for resources (Ayars et al. 2015; Schmidt et al. 2018). It has been speculated that reductions in root biomass may lead to reduced carbon inputs, aggregate formation (Schmidt et al. 2018), microbial activity (Wolf et al. 2016), and thus, overall SOM. Reduced moisture at the field scale has also been shown to increase the accumulation of salts and surface crusting, reducing overall resilience to climate extremes (Ayars et al. 2015; Rath et al. 2017). Nearly 50% of semi-arid, irrigated landscapes are impacted by salinity (UNEP 2014), which can encourage loss of SOC (Setia et al. 2013) both through dispersion of aggregates (Wong et al. 2010) and lower microbial carbon use efficiency (Rietz and Haynes, 2003) and threaten yields.

Micro sprinklers are often incorporated in California orchards, as they support a higher water application rate; maximizing the wetted diameter, without wetting the crown of the tree (Schwankl 1999). Like drip systems, micro sprinklers have been shown to increase tree growth and almond yields, while providing more uniform applications and improving water use efficiency. The use of micro-sprinklers in almond systems has been shown to reduce nitrous oxide emissions relative to drip, while improving almond tree vigor and yields (Schellenberg et al. 2012; Alsina et al. 2013). Though less efficient, overhead sprinklers which are commonly used in Midwest annual crops are slowly gaining interest in California (~2% of acreage). Research has shown they can produce similar or higher yields in many cropping systems, while requiring less maintenance than drip (Mitchell et al. 2016). Overhead sprinklers may also be helpful in dealing with salinity, as they uniformly wet the root zone and allow for leaching of salts.

On-farm water use can also be reduced through deficit irrigation and dry farming. Preliminary studies in California have shown promise for dry-farming tomatoes, vines, and tree crops, but much work remains to determine how much and at what stage of the crop life cycle to reduce irrigation, especially across the range of crops, climates, and soil types statewide. Tomatoes experienced no significant yield differences when irrigation was cut off 45 days prior to harvest, saving 0.5 ac/ft of water and increasing overall water use efficiency by 19% (Ory et al. 2016). Similarly, Johnstone et al. 2005 found that reducing irrigation by 25-50% during the 4-7 weeks prior to harvest increased soluble solids with no significant loss of brix yield (Mg fruit solids ha^{-1}). Preliminary research at the Century

Experiment also indicated that reducing irrigation 20-30% in the six weeks prior to harvest can increase soluble solids without impacting yield (Tautges et al. 2018). In vineyard systems, full irrigation has been shown to have negative consequences. By promoting excessive vegetative growth; full irrigation decreased sugar content of the fruit, subsequently impacting overall wine quality (Matthews et al. 1990; Esteban et al. 2001). Deficit irrigation, on the other hand, has resulted in similar or even improved quality (higher anthocyanins and total phenols) with no impact on yield (Goldhamer 1999; Lampinen et al. 2004; Chaves et al. 2007; Iniesta et al. 2009; Abrisqueta & Ayers 2018).

Several crops in the state still rely on rain-fed systems (i.e. safflower, oat hay, wheat, barley) and several growers are experimenting with dry farming in tomatoes, pumpkins, watermelons, cantaloupes, winter squash, garbanzos, apricots, apples, grains, and potatoes. Dry farming traditionally relies on the use of tillage, surface coverage, and drought-tolerant varieties to grow a crop in the dry season using residual soil moisture accumulated during the winter months. Landscapes that receive a minimum of 15-20” of annual rainfall and are situated on clayey soils are strong candidates for dry farming (Schillinger et al. 2006). Although yields may be significantly reduced, dry farming allows for crop production without the substantial investment in or reliance on irrigation infrastructure. A case study of Frog’s Leap Vineyard, for instance, showed a savings of 16,000 gallons/acre, or a total of 10 acre feet across 200 acres, while still achieving yields of 4 tons/acre (Runsten, 2019). Further research in California is needed to determine impacts on soil health indicators and carbon sequestration, particularly when used in combination with reduced tillage and cover cropping.

While converting to more efficient irrigation and monitoring systems can be cost prohibitive to growers relative to furrow and solid set sprinkler systems, the California State Water Efficiency and Enhancement Program (SWEEP) has provided financial support to growers for the installation of soil moisture sensors and micro- or subsurface drip irrigation systems. Such investments have supported the adoption of subsurface drip on 39% (~3 million acres) of California’s irrigated farmland (Johnson & Cody, 2015). The SWEEP program has also supported the installation of solar panels to reduce energy use and costs associated with irrigation.

On-farm water use can also be reduced through dry farming.

Mineral Fertilizer Management

In addition to irrigation, the development of mineral fertilizers allowed for drastic gains in agricultural productivity (Robertson and Vitousek, 2009). The low cost of mineral fertilizers relative to the tight profit margins growers face encouraged frequent over-fertilization, as a proactive measure against yield losses. This, however, comes at a great environmental (and, increasingly, economic) cost with an estimated 46% of nitrogen in California agroecosystems (~310,000 Mg of N) lost to runoff, leaching, and volatilization (Tomich et al. 2016). These losses lead to eutrophication of waterways, groundwater nitrate pollution, and elevated nitrous oxide emissions, respectively (Jackson et al. 2009; Burger and Horwath 2012).

Nitrogen in the form of nitrate, although readily taken up by plants, can easily be lost when applied in excess of crop demand. Nitrate's negative charge is repelled by negatively charged clay particles in soil, allowing it to move with water down the soil profile, often leaching into groundwater. Nitrate losses across California are estimated at ~333 Gg N yr^{-1}. Baram et al. 2016 measured nitrate losses of 80-240 kg^{-1} ha^{-1} yr^{-1} in a single California almond orchard. Excessive moisture combined with either high or low oxygen concentrations also increases the potential for losses. As microbial activity and decomposition of C/N-based compounds speeds up, nitrogen-based GHG are produced as byproducts (Sapkota et al. 2020).

The use of N fertilizer beyond crop demand leads to exponential increases in emissions, as excess nitrate is converted to ammonium and/or excess ammonium to N2 gas. Minimizing this excess is thought to provide one of the greatest opportunities for mitigating agricultural GHG (McSwinney and Robertson 2005; Van Groenigen et al. 2010). It is estimated that nitrous oxide emissions associated with fertilizer could be

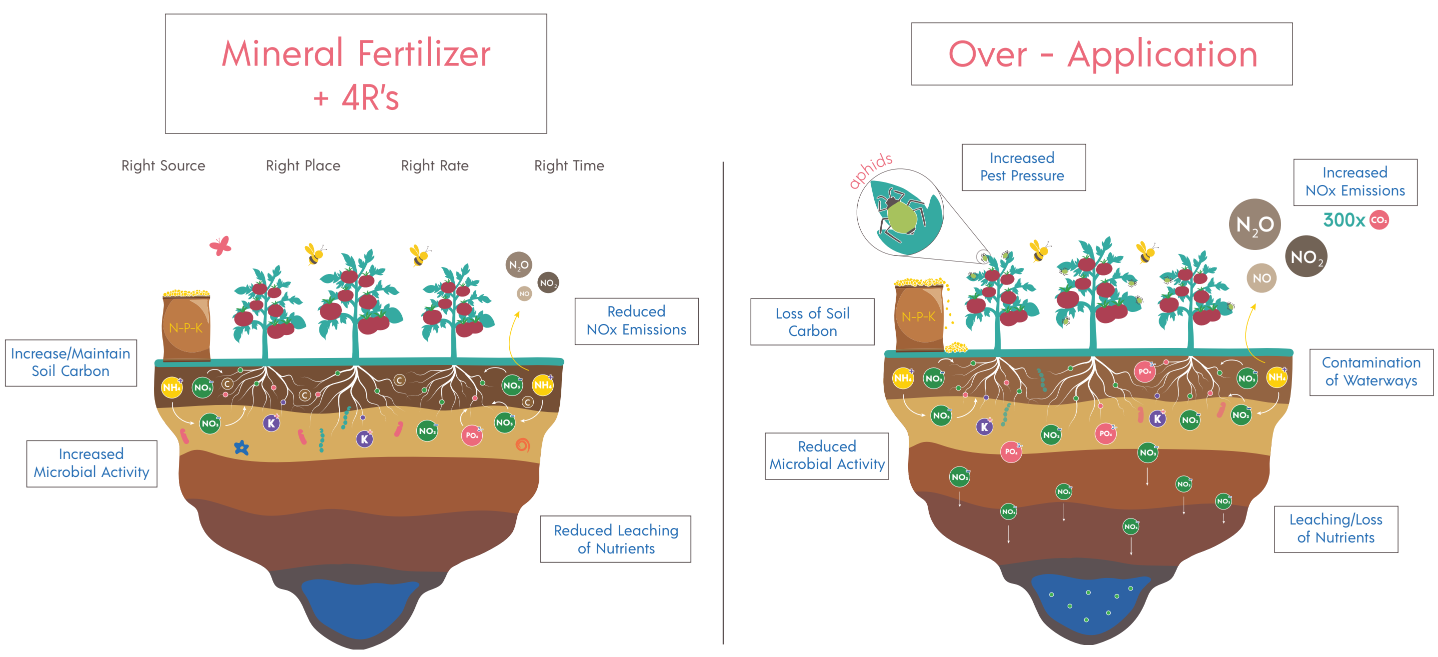


Figure 7. Diagram of expected impacts of mineral fertilizer application utilizing the 4R's compared to over-application. Implementing the 4R's can significantly reduce nutrient losses in the forms of emissions and leaching.
Credit: Soil Life and CAFF

reduced by up to 50% simply by matching the rate of application to crop demand (Millar et al. 2010; van Groenigen et al. 2010). Under-fertilizing can also lead to increased emissions of carbon dioxide, as roots and microbes turn to the mineralization of SOM to meet their nutrient requirements (Kuzyakov et al. 2000; Fontaine et al. 2003).

Applying 4R's (right place, right time, right rate, and right source) is a key nutrient management practice to minimize nitrogen losses and better synchronize fertilizer application and crop uptake (Smil 2001; Johnson et al. 2007; Snyder et al. 2009). Implementing the 4Rs has been shown to reduce emissions 17-30% in California, without significantly impacting yields (Burton et al. 2008; DeGryze et al. 2009; Aita et al. 2015).

Splitting fertilizer applications into two smaller events, for instance, reduced emissions in California cropping systems (Hutmacher 2012; Bottoms et al. 2013; Orloff et al. 2013). Spatially splitting up nitrogen fertilizer can also reduce emissions and improve nutrient use efficiency. For instance, 200 lbs/ac of urea ammonium nitrate applied in two bands, rather than one, led to 70% less nitrous oxide emissions in corn systems (CARB 2016). While less mobile nutrients like phosphorus and potassium need to be placed close to the root zone, nitrogen can be broadcast more generally. Changing the source from anhydrous ammonium to ammonium sulfate also produced 30% less nitrous oxide emissions (Zhu-Barker et al. 2015). Drip irrigation allows for fertilization (or fertigation) to occur in smaller, more frequent doses applied directly to the root zone; reducing potential losses. Fertigation also reduces tractor passes and associated fuel costs.

Fertilizer use has been shown to both increase and decrease SOM (Zhang et al. 2010; Geisseler and Scow, 2014; Zhu et al. 2016). The boost in primary production brought on by fertilization has traditionally been thought to increase total carbon returned to the soil (via increased plant biomass), but may also be a result of improved carbon use efficiency due to improved stoichiometry of nutrients. Two meta-analyses of long-term trials found that fertilizer use increased both microbial biomass and total SOM by 15.1% and 12.8%, respectively, relative to unfertilized controls (Aguilera et al. 2013; Geisseler & Scow 2014). Historical breeding for aboveground traits combined with a consistent, reliable supply of fertilizer may discourage plants from investing in roots, reducing total overall carbon inputs to the system (Waines and Ehdaie 2007; Schmidt 2018). Certain fertilizers (ammonium and urea based) can reduce soil pH, leading to reduced availability of many plant nutrients and subsequent decreases in crop yield and thus, C inputs (Barak et al. 1997; Zhang et al. 2008; Francioli et al. 2016). Microbial diversity, activity, and biomass are also sensitive to fluctuations in pH (Fierer & Jackson 2006), and may explain the reduction in microbial biomass and SOM observed in several field-based studies (Khan et al. 2007; Roberts et al. 2011; Lazcano et al. 2013).

A systems perspective would consider more than just the use or disuse of fertilizers, but also the amount of carbon-based amendments returned to the soil and the resulting ratio of C:N, C:P, and C:S (Lal et al. 2003; Kirkby et al. 2013). In order for SOM to form, there must be carbon, nitrogen, phosphorus, and sulfur available in quantities in excess of crop demand to build microbial biomass. Insufficient quantities of any nutrient may limit the ability of microbes to convert carbon into biomass and subsequently, stable SOM (Kirkby et al. 2013). Conversely, insufficient quantities of carbon may lead to an inability of microbes to utilize nutrients and/or encourage the consumption of old SOM to meet microbial energy/carbon needs (Kuzyakov et al. 2000; Fontaine et al. 2003).

The 4 R's can also be applied to organic sources of nutrients such as compost. *Photo by Bonnie Veblen*

Integrating Towards More Diversified Cropping Systems

It is widely understood that diversified cropping systems have positive impacts, for both farms and the environment.

Diversified cropping systems strive to emulate nature by incorporating diversity at various scales -- crop rotations, intercropping, and compost at the plot scale; polycultures, pollinator strips, and crop/livestock integration at the field scale; and hedgerows, buffer strips, pastures, and ponds at the landscape scale (Lin 2011; Kremen and Miles 2012). Diversification reduces vulnerability to extreme events and breeds resilience, spreading economic risks across multiple crops, reducing the dependence on off-farm inputs (i.e. fertilizers, pesticides, fossil fuels), and increasing reliance on biological processes for fertility and pest and disease control (Altieri et al. 1999; Jackson et al. 2011, McDaniel et al. 2014; Hodson and Lewis 2016). It is widely understood that diversification, whether at a temporal (across time; i.e. crop rotations) or spatial (across space, i.e. intercropping) scale, has a positive impact both environmentally and agronomically (Culman et al. 2010; Brennan & Acosta-Martinez 2017). Across a gradient of agricultural intensification within the Sacramento Valley, plant diversity was consistently found to be correlated with increased soil carbon, microbial biomass, and diversity; reduced soil nitrate and phosphorus loadings; and improved riparian health (Culman et al. 2010).

Diversification reduces on-farm vulnerability.

Since the 1940's, mechanization and inexpensive fertilizers and pesticides have made it increasingly economical to forego diversity in favor of high-density monocultures (Altieri 1999; Pimentel et al. 2005). This shift has accelerated in recent years with the number of commodities produced per farm dropping from five in 1990 to less than two in 2002 (Dimitri et al., 2005). Mechanization favors monocultures because they are easier to manage at scale. Equipment can be specialized to one crop, improving efficiencies and reducing labor time and costs (Pimentel et al. 2005; Asai 2018). Fertilizers and pesticides provide an inexpensive substitute for manures, legumes (for fertility) and crop rotations (to disrupt weed, pest and pathogen cycles), and tend to require less labor (Altieri 1999). In recent years, however, input costs have become less stable, leaving growers even more vulnerable to market pressures (USDA 2016; Maples et al. 2019). Simultaneously, there has been growing awareness as to the benefits of diversification and the externalities of large-scale monoculture cropping systems (pesticide-resistant weeds and pests, runoff, etc.) (Brummer, 1998; Randall, 2003; Asai 2018).

Diversified systems often have lower overall GHG footprints compared to conventional systems as they tend to use less fossil fuels, store more carbon (both belowground as SOM and aboveground as woody biomass), and emit less GHG from the soil (Kremen and Miles 2012; Morugán-Coronado et al. 2020). The soil microbial communities of these systems are more likely to receive adequate fuel and nutrition throughout the year from a diversity of sources and, thus, may exhibit improved physiology (Kallenbach et al. 2015). As such, microbial populations tend to grow faster and exhibit greater carbon use efficiency. In the process of building biomass, microbes store more than carbon. They also store important plant nutrients like N and P, protecting them against loss until the crop needs them and creating more tightly coupled nutrient cycles (Smukler et al. 2011; Bowles et al. 2015; Kallenbach et al. 2015). This results in fewer overall losses of nitrogen in the form of nitrate and nitrous oxide; offsetting increases in respiration (carbon dioxide) that commonly accompany increases in organic inputs and/or SOM content (Jackson et al. 2004; Bowles et al. 2015).

The meta-analysis on Mediterranean agroecosystems by Aguilera et al., 2013 found that the combined use of climate-smart practices (compost/manure, crop residues, cover cropping, and/or reduced tillage) in Mediterranean cropping systems increased SOC levels by 50% over conventional management, a greater gain than any conservation practice in isolation. Similarly, models have shown that while individual conservation practices produce modest emissions reductions, combining (or stacking) practices produced significantly larger reductions (De Gryze et al. 2009). Several long-term, controlled studies in California have corroborated these findings, showing that the use of compost, cover crops, and crop rotation has a synergistic effect, resulting in significantly greater microbial biomass, microbial diversity, and overall SOM, as compared to conventional crop rotations (Horwath et al. 2002; Wolf et al 2016; Brennan & Acosta-Martinez 2017).

Co-benefits of diversified systems have also been tied to positive economic impacts. Studies have found yields to be similar, and, at times, higher and/or more stable under diverse polycultures (Altieri 1999; Tilman et al. 2002). Even where weed densities are higher, diversified systems in California have been found to have similar yields as conventional systems (Poudel et al. 2001; Horwath et al. 2002). Under reduced yields, diversified systems have also been shown to experience increased profits, as savings on inputs often outweigh reductions in yield/revenue (Palm et al. 2014; Asai et al. 2018). Diversified systems can be tailored to fit various farm models based upon operation size, labor inputs, and equipment/resource availability, from small-scale, biointensive no-till systems which rely more heavily on labor, to reduced or conservation tillage systems which typically utilize reduced-disturbance machinery.

Consistent ground coverage in these systems is able to outcompete or suppress weeds, while providing enhanced wildlife habitat on-farm (Andrews et al. 2002; Hodson & Lewis 2016). In a 3-year study in the San Joaquin Valley, CA, where high temperatures and heavy use of tillage contribute to rapid decomposition of organic matter, the combined use of crop rotation, compost, and cover crop was found to significantly improve 16 out of 18 soil health indicators over conventional management. Indicators included bulk density, aggregate stability, SOM, total N, microbial biomass C and N, available P, K, Fe, Mn, and Zn (Andrews et al. 2002). Numerous studies have found that as crop diversity increases, there are more diverse arthropod communities, greater populations of pollinators and fewer crop pests (Risch et al. 1983; Russell 1989; Andow 1991; Letourneau et al. 2011; Kennedy et al. 2013; Lichtenberg 2017).

Diversification promotes beneficial microbes in soil.

Diversified systems often enhance the provision of ecosystem services, as well, exhibiting improved structure and aggregate stability, increased infiltration and water holding capacity, reduced runoff, erosion, and leaching, and improved water quality (Andrews et al. 2002; Altieri 2002). These systems also promote healthy populations of beneficial microbes and insects. This leads to increased defense against pests/disease (Blundell et al. 2020), as well as functional redundancy within communities, thereby contributing increased resilience under environmental extremes that are expected to increase with climate change (Lin 2011; Hodson and Lewis 2016).

Conclusions and Recommendations

There is substantial evidence that climate smart agriculture practices provide a net benefit to farms in terms of climate resiliency, mitigation, and adaptation. Through carbon sequestration, tighter nutrient cycling, and improved overall efficiencies, climate smart practices have the ability to significantly reduce the footprint of California agriculture. At the same time, climate smart agriculture plays a critical role in conserving key natural resources such as topsoil, biodiversity, and water, and the provision of ecosystem benefits both on and off the farm. These practices build resilience through improved soil health, biodiversity and increased long-term productivity. **As such, climate smart agriculture is an opportunity, not only to address the increasingly serious threat of climate change, but also to ensure the longevity and health of California agroecosystems.**

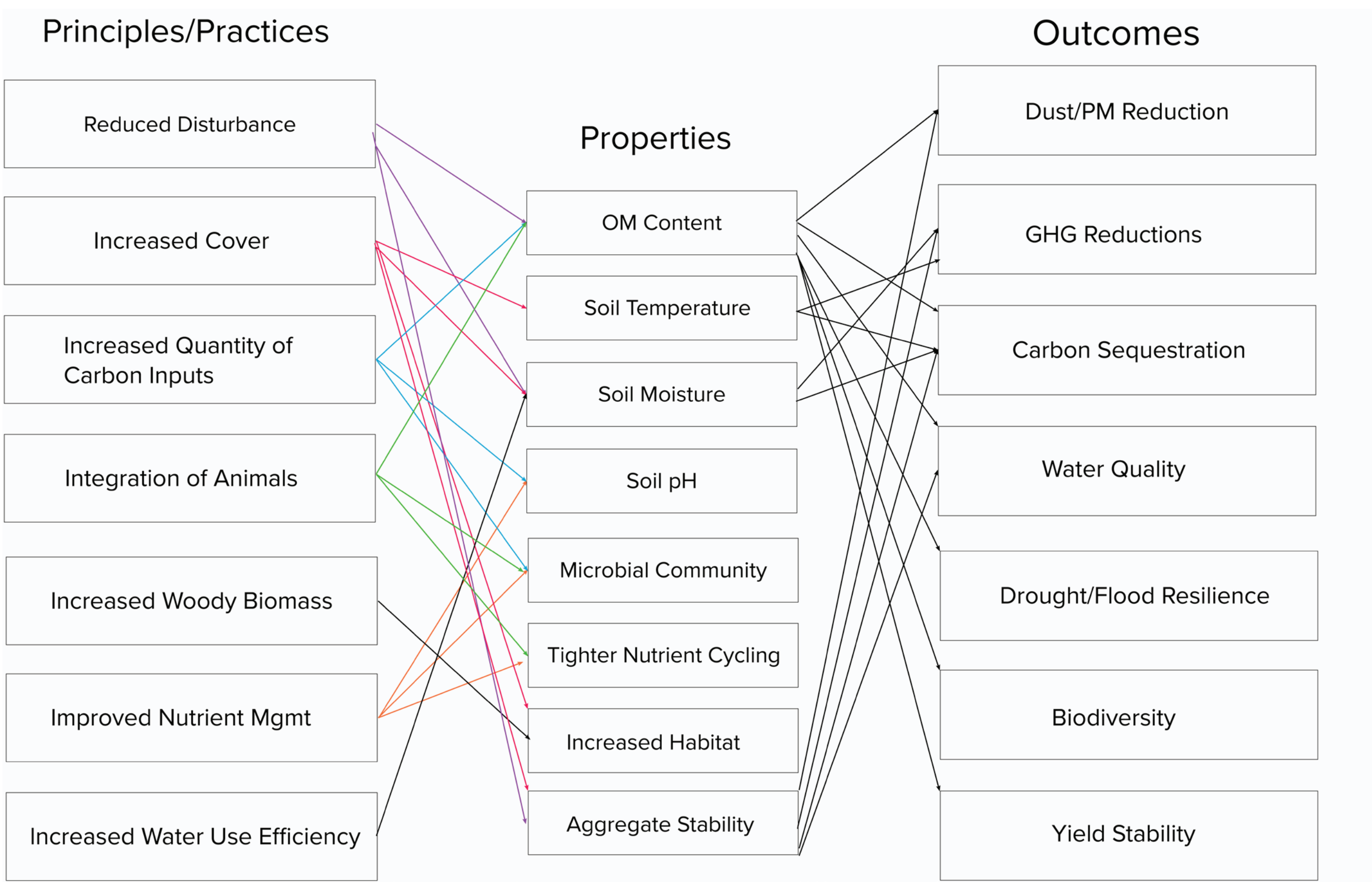


Figure 8. Conceptual diagram of climate smart principles/practices and their influence on soil properties, which in turn can result in beneficial outcomes and ecosystem services (varies by soil type, climate, cropping system), many of which feedback to soil properties.

The success of climate smart agriculture will depend on producers' willingness to implement new practices for the long-term. *Photo by Singing Frogs Farm / Elizabeth and Paul Kaiser*

While climate smart agriculture provides opportunities to achieve the triple bottom line of environmental, agronomic and economic benefits across the diverse cropping systems of California, it also presents challenges. California is an incredibly diverse state, and the variation in microclimate, topography, soil type, and cropping systems makes it difficult to generalize about the impact of a given practice on an agricultural landscape; or the impact of shifting climate on an agroecosystem. **Successful implementation of climate smart practices requires consideration of environmental factors such as soil type, climate and water availability. There must be recognition that not every conservation practice or combination of practices is appropriate for every region or farm.** Greater understanding of the intricacies of different systems will allow growers and the extension and technical assistance communities to hone and optimize practices in their unique contexts to achieve the host of benefits possible through climate smart agriculture.

Research is needed that considers the complex synergies of stacking multiple practices on various soil types and microclimates across the state. While difficult to measure, a better understanding of integrated systems will benefit both the research and agricultural communities of California. Additional research is also needed on: the impact of climate smart practices on resilience to droughts and floods; the impact of cover crops (including diversity of mix, planting date, termination strategy) on the overall water budget; the impact of no-till/cover crops on nutrient cycling/availability; the impact of compost on weed establishment; the impact of cover crops on trace gas emissions; the impact of increased SOM on water holding capacity in the field; and the impact of livestock in the understory of perennial cropping systems.

In addition to further research, **the success of climate smart agriculture will ultimately depend on California's agricultural producers' willingness to implement and adopt these practices for the long-term.** In order to scale adoption, farmers must be supported through policy initiatives that not only incentivize implementation, but comprehensively address barriers to adoption across the diversity of California's cropping systems, farm scales and agricultural communities, including socially disadvantaged farmers and ranchers. A collaborative effort that brings together the expertise of farmers, researchers and the extension and technical assistance communities will improve the efficacy of such policy initiatives, leading to the success of climate smart agriculture in California.

Finally, maintaining agricultural lands will also be crucial to state climate mitigation goals. Urban development has contributed to the loss of nearly 3.4 million acres of farmland over the last decade (Liu et al. 2003, Norman et al. 2006, NASS 2007). Emissions from an urban acre have been found to be 70 times that of an agricultural acre in California (Haden et al. 2012). Bridging the rural-urban divide will also be critical to increasing consumer awareness and demand for climate smart commodities.

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