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Identifying Greenhouse Gas Emission Sources and Mitigation Opportunities in California Specialty Cropping Systems

Xia Zhu-Barker[a], Mary Carlson[b], Mark Easter[b], Amy Swan[b], Lucas Thompson[b], William R. Horwath[a], Keith Paustian[b], Kerri L. Steenwerth[c,d]

[a]Department of Land, Air & Water Resource, University of California, Davis, CA 95616

[b]Natural Resource Ecology Laboratory, Colorado State University, Ft. Collins, CO

[c]USDA-ARS, Crops Pathology and Genetics Research Unit, Davis, CA 95616

[d]USDA California Climate Hub, Davis, CA 95616

Contact: kerri.steenwerth@ars.usdsa.gov

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Executive summary

Greenhouse gas (GHG) emissions from California agriculture represent 8% of the state's total emissions. The contribution from California specialty crop systems to this percentage is still highly uncertain. Identification of GHG mitigation opportunities in California specialty crop systems requires sound information on how current and emerging land management practices affect those emissions. This report presents a review of scientific literature examining GHG emissions, focusing on California's annual and perennial specialty crop systems. It also invokes research from other geographic regions to build a comprehensive background on agricultural practices and underlying processes that lead to GHG emissions. A total of 18 studies capturing GHG emissions were identified in five specialty cropping systems within California. Nitrous oxide emission factors were developed when possible, and state-wide emissions for certain specialty cropping systems were calculated under specific agricultural management practices. However, large uncertainties caused by low gas sampling frequency in these studies diminished the reliability of estimates of the biophysical mitigation potential of various agricultural management practices. This uncertainty can be remediated by robust and standardized estimates of GHG emissions from changes in agricultural management practices in California specialty cropping systems.

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1. Introduction

Agriculture contributes about 8.3% to total U.S. greenhouse gas (GHG) emissions, mainly in the form of nitrous oxide (N_2O) and methane (CH_4), and adds approximately 80% to total U.S. N_2O emissions (USEPA, 2016). Nitrous oxide is mainly produced by soil processes that vary with management activities, and it has a global warming potential (GWP) approximately 298 times that of CO_2 (IPCC, 2013). As such, development of practices to reduce N_2O emissions from agriculture plays an important role in mitigating GHG emissions. In contrast, methane production in the U.S. agricultural croplands is relatively low, with most of the CH_4 emissions contributed by animal agriculture and manure management (USEPA, 2016).

California contributes 7% to total U.S. agricultural GHG emissions, making it the nation's largest agricultural emitter (CARB, 2015; USEPA, 2016). Yet, California is the nations largest agricultural producing state (in terms of cash receipts) and has the largest specialty crop acreage (Hart, 2003; Culman *et al.*, 2014). Specialty crops are a diverse group: the USDA lists 208 fruits, tree nuts, vegetables, and herbs that are commonly considered specialty crops (Agricultural Marketing Service, 2016). California is the nation's only commercial producer for many specialty crops, including almonds, artichokes, figs, raisins, walnuts, pistachios, nectarines, olives, dates, and prunes,. For other specialty crops like wine grapes, strawberries, leaf lettuce, garlic, broccoli, and brussel sprouts, California's share approaches or exceeds 90% of U.S. production (Starrs and Goin, 2010). However, only a handful of crops (e.g. grapes, almond, strawberry, lettuce, walnut, tomato, cole crops, and stone fruit) account for the large majority of specialty crop value and land area in California. GHG emissions from California agriculture have been reported to account for 8% of the state's total emissions (CARB, 2015), but the contribution from California specialty crop systems is still highly uncertain due to the diverse range of regional microclimates, soil types and crops, and complex rotation schedules currently in use (Hart, 2003; Culman *et al.*, 2014; USDA, 2014a; CARB, 2015). Ongoing efforts are dedicated to the analysis of GHG emissions and to the identification of GHG mitigation opportunities in California agriculture (DeLonge *et al.*, 2014; Owen *et al.*, 2014; Sumner, 2014). Still, the incomplete calibration and application of process-based models in California specialty crop systems hampers the progress of these efforts (De Gryze *et al.*, 2009; Gryze *et al.*, 2010; De Gryze *et al.*, 2011) and limits the extent to which well-constrained estimations can be made concerning GHG fluxes.

2. Overview of nitrous oxide sources and production pathways in cropping systems

Nitrous oxide is the dominant GHG emitted from California specialty cropping systems and is mainly produced in soils (Bai *et al.*, 2012; Schlesinger, 2013; USEPA, 2016) through ammonia oxidation pathways, heterotrophic denitrification, and abiotic chemodenitrification reactions (Firestone and Davidson, 1989a; Van Cleemput and Samater, 1996; Zhu *et al.*, 2013a; Zhu-Barker *et al.*, 2015a). In Appendix 1, we synthesize current knowledge on the mechanistic understanding of N_2O production pathways and controls on these pathways in cropland soil systems. Although all of these pathways are not explicitly accounted for in the current methods, we briefly outline the individual processes and factors that lead to N_2O emissions so that such factors can be adjusted by agricultural management practices to reduce N_2O emissions.

3. General effects of management practices on greenhouse gas emissions

Specialty cropping systems are sources of N_2O emissions, yet they have a large potential to mitigate GHG emissions by changing soil management (Smith *et al.*, 2008; USDA, 2014a). N_2O emissions derived from agricultural management practices like fertilization, irrigation and tillage are key sources of GHG emission in specialty cropping systems (Culman *et al.*, 2014; USEPA, 2016). These N_2O emissions are highly variable in time and space due to numerous factors such as water content, oxygen levels, pH, and the availability of C and N (Bremner *et al.*, 1980; Bouwman *et al.*, 2002b; Bateman and Baggs, 2005; Stehfest and Bouwman, 2006; Zhu *et al.*, 2013a; Maharjan *et al.*, 2014). Management practices impact N_2O emissions from soils by influencing these factors and the activities of microorganisms involved in heterotrophic denitrification and ammonia oxidation pathways (Avrahami *et al.*, 2002; Avrahami and Bohannan, 2007; Garland *et al.*, 2011; Venterea *et al.*, 2011; Kennedy *et al.*, 2013; Zhu-Barker *et al.*, 2015b; Zhu-Barker *et al.*, 2015c). California's Mediterranean climate influences the adoption of these management practices and associated soil N_2O emissions. For example, irrigation is a common and requisite practice in California specialty cropping systems due to the Mediterranean climate. As a consequence, the soils in these cropping systems experience frequent dry-wet cycles during warm summers, resulting in episodic N_2O emission patterns that differ from the cropping systems in temperate or continental climates.

This section summarizes current state of the science across multiple geographic regions and focuses on how management practices influence GHG emissions from cropping systems. Specific practices and their general effect on N_2O emissions, primarily, will be discussed first and then research from California specialty crops will be covered.

3.1 Fertilization

In California's specialty crop systems, synthetic and organic N fertilizers are an essential input to maintain high crop yields and quality. They are also the main source of N_2O emissions from the agricultural sector (Bouwman *et al.*, 2002a; Davidson, 2009; USEPA, 2016). Fertilizers drive N_2O emissions from soil through their contribution to the availability of N substrates for soil microorganisms (e.g. nitrifiers and denitrifiers) and abiotic reactions, and through their influence on soil pH and oxygen availability during nitrification (Harrison *et al.*, 1995; Mulvaney *et al.*, 1997; Smith *et al.*, 1998; Venterea and Rolston, 2000; Zhu *et al.*, 2014). The influence of fertilizers on N_2O emissions is also soil- and climate-specific since variations in temperature and soil water content under different climates can significantly affect N_2O production through their influence on microbial activity and substrate availability (Stark and Firestone, 1995; Avrahami *et al.*, 2003). Here, this section will cover the effects of fertilizer rate, type, placement, application timing and efficiency enhancers on N_2O emissions from agricultural systems.

3.1.1 Fertilizer rate

The application rate of N fertilizer strongly influences N_2O emissions (Grant *et al.*, 2006; Zebarth *et al.*, 2008b; Burger and Horwath, 2012; Zhu *et al.*, 2014). Generally, the relationship between N rate and N_2O emissions has been assumed to be linear and a default emission factor (EF) of 1% was adopted for use by the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2007). This N_2O EF approach has been used to construct most national GHG inventories (de Klein *et al.*, 2006; USDA, 2014a). However, field studies on multiple N fertilizer rates indicate that N_2O emissions often respond nonlinearly to increasing N rates across a range of climate, soil textures, and fertilizer types (McSwiney and Robertson, 2005; Ma *et al.*, 2010; Hoben *et al.*, 2011; Zhu-Barker *et al.*, 2015c). Shcherbak et al (2014) demonstrated this nonlinear response of soil N_2O emissions to fertilizer N rates by conducting a meta-analysis on emissions data from 78 published studies with 233 site-years and at least three N input levels. These studies indicate that the Tier 1 N_2O accounting method adopted by IPCC, whereby N_2O emissions are assumed to be a simple fraction of N inputs, has the potential to underestimate or overestimate fertilizer derived N_2O emissions.

In California specialty cropping systems, N fertilizer application rates vary from crop to crop and location to location (https://quickstats.nass.usda.gov/). Nevertheless, the rate of N fertilizer application can be refined to reduce N_2O emissions from these systems as long as it does not lead to undesired declines in crop yields (Burger and Horwath, 2012; Pereira, 2014). Loss of optimal crop growth in response to reductions in N fertilizer can promote soil organic matter mineralization and increase residual fertilizer in soil, leading to loss of soil C and increased GHG emissions. For example, in a California tomato system, the N_2O emitted per unit of N fertilizer applied (i.e. emissions factor, or EF) was significantly higher in plots receiving N fertilizer at 75 kg N ha^{-1} compared to plots receiving 162 kg N ha^{-1} N fertilizer (EF 1.63% vs. 1.11%) (Burger and Horwath, 2012). Higher EFs were also found with lower N fertilizer application rates in California wine grape vineyard systems (Smart *et al.*, 2006; Verhoeven and Six, 2014). For example, using the flux rates reported by Smart et al. (2006), the EF of N_2O was calculated as 0.96% in plots receiving N fertilizer at 6 kg ha^{-1} while it was 0.2% in plots receiving N fertilizer at 45 kg ha^{-1}. Recent efforts to consider the importance of crop yields when examining agricultural emissions have led to the usage of yield-scaled emissions (Van Groenigen *et al.*, 2010; Murray and Baker, 2011; Zhu-Barker *et al.*, 2015c). Such scaling balances the inherent tradeoff between crop productivity and N_2O emissions by providing a metric that reflects the efficiency of N fertilizer use in a given cropping system.

In general, practices that increase crop N use efficiency (NUE) are expected to reduce N_2O emissions, as applied N taken up by crops would not be available to soil microorganisms that produce N_2O. However, strategies that improve NUE do not always reduce N_2O emissions. Other practices, such as changes in fertilizer type and placement, can result in different N_2O emissions irrespective of NUE effects (see section 3.1.3) (Gagnon and Ziadi, 2010; Fujinuma *et al.*, 2011; Gagnon *et al.*, 2011; Zhu-Barker *et al.*, 2015c). In most California specialty cropping systems, data on the effect of N fertilizer rates on N_2O emissions when fertilizer is applied via drip irrigation, especially subsurface drip irrigation, are still lacking.

3.1.2 Fertilizer type

The effect of N fertilizer type on direct N_2O emissions from soils is influenced by the form of available N substrates, i.e. NH_4^+, NO_3^- or organic N, and short- and long-term changes in soil pH after application. In California, commonly used synthetic N fertilizers include urea-ammonium-nitrate (UAN), calcium-ammonia-nitrate (CAN), anhydrous ammonia, ammonium sulfate, and urea, with an annual average of 0.576, 0.258, 0.237, 0.177, and 0.150 megatonnes, respectively, sold between 2007 and 2012 (CDFA, 2012; ERS, 2016). These fertilizers are ammonia-based and their long-term application generates soil acidity through nitrification (Hauck and Stephenson, 1965; Bouman *et al.*, 1995; Mulvaney *et al.*, 1997). Once the soil environment becomes anoxic, soil acidity can promote N_2O emissions during denitrification (Koskinen and Keeney, 1982; Mulvaney *et al.*, 1997), likely because the substrate affinity of nitrate reductase is higher than that of N_2O reductase (Betlach and Tiedje, 1981; Ottow *et al.*, 1985; Nägele and Conrad, 1990). Furthermore, these fertilizers can be divided into alkaline-forming (e.g. anhydrous ammonia and urea) and acidifying N fertilizers (e.g. ammonium sulfate and ammonium nitrate), depending on the immediate change in soil pH after their application. Nitrification rates are generally higher for alkaline-forming than for acidifying N fertilizers (Hauck and Stephenson, 1965; Mulvaney *et al.*, 1997). Application of alkaline fertilizer initially increases the soil pH and the presence of free ammonia, which together promote the formation of nitrite (Park and Bae, 2009; Hawkins *et al.*, 2010; Venterea and Coulter, 2015; Breuillin-Sessoms *et al.*, 2017), a substrate for N_2O production (Venterea and Rolston, 2000; Venterea, 2007). Under anoxic soil conditions, alkaline-forming fertilizers can also stimulate denitrification by increasing the solubility of soil organic matter (Norman *et al.*, 1987). However, the effect of dissolved organic C (DOC) on N_2O emissions after applications of alkaline-forming fertilizers is difficult to ascertain because DOC can also promote the consumption of N_2O to N_2. In comparison to alkaline-forming fertilizers, acidifying fertilizers tend to promote higher N_2O emissions through denitrification under anoxic soil conditions, especially in soils that have a low initial pH (Mulvaney *et al.*, 1997). In spite of these effects, a review conducted on a global scale has concluded that the differences in emissions among fertilizer types are often only marginal (Stehfest and Bouwman, 2006).

In California cropping systems, the effects of N fertilizer type on N_2O emissions are varied. In almond orchards, higher N_2O emissions were associated with UAN application compared to CAN application (Schellenberg *et al.*, 2012; Wolff *et al.*, 2016). In a side-by-side field trial in California wheat cropping systems, knife injection of anhydrous ammonia increased N_2O emissions compared to ammonium sulfate (Zhu-Barker *et al*. 2015c). Higher N_2O emissions from urea application than from ammonium sulfate, ammonium nitrate, and calcium nitrate have also been found in studies of field barley cropping systems and laboratory soil incubations (Tenuta and Beauchamp, 2003; Zhu *et al.*, 2013a). In some of these experiments, both fertilizer type and placement varied among treatments to reflect typical fertility practices, and this may have influenced the results. In most California specialty cropping systems, side-by-side trials to compare N_2O emissions from different fertilizer types and identify consistency of fertilizer effects are needed.

3.1.3 Fertilizer placement

Crop yield and N_2O emissions can be significantly affected by fertilizer placement. In the field, fertilizer can be applied by spraying, banding, subsurface banding, broadcasting, and fertigation (i.e. delivered through knife injection, surface, subsurface drip irrigation, or microsprinkler), depending on crop and fertilizer types. Improper placement of fertilizers can lead to N loss, causing environmental impacts, reduced yield potential, and decreased nutrient use efficiencies, ultimately resulting in economic losses to farmers.

Fertilizer placement affects N concentration, oygen availability and soil pH in the surrounding soil, and therefore also affects N_2O emissions. Generally, fertilizer band application results in higher soil N concentrations around the banding location as compared to a broadcast application. Concentrated ammonical fertilizers increase the consumption of oxygen (Zhu *et al.*, 2014), as well as the potential for nitrite accumulation and N_2O production (Mulvaney *et al.*, 1997). In a greenhouse and field study in Montana, N_2O production increased from 2.8 to 5.0 to 6.1 g N ha^{-1} d^{-1} when fertilizer placement switched from broadcasting to banding in a nest application, respectively (Engel *et al.*, 2010). Similarly, in an irrigated corn field study in northeastern Colorado, Halvorson and Del Grosso (2013) found that a surface-banded urea application led to higher N_2O emissions than a surface broadcast application. In Minnesota, Maharjan and Venterea (2013) also reported significantly higher N_2O emissions from mid-row banding of 180 kg N ha^{-1} urea relative to broadcasting followed by incorporation. However, slightly less N_2O emissions from subsurface banding than from surface broadcasting occurred in wheat and canola systems in Canada (Hultgreen, 2003). Together, these findings indicate the importance of considering the effects of climate variation between growing regions and fertilizer placement on N_2O emissions when scaling up field observations.

The effect of fertilizer application depth on N_2O emissions also depends on how tillage practices affect soil water status, oxygen availability, and vertical distribution of C availability (Alcántara *et al.*, 2016). In a wheat system in Eastern Canada, higher N_2O emissions were observed from a side dress of ammonium nitrate placed at 10 cm compared to N placement at 2 cm (Drury *et al.*, 2006). This was likely due to the limited oxygen availability in the deeper soil. The highly volatile nature of anhydrous ammonia restricts its mode of application to subsurface injection, which generally results in a highly-concentrated band of accumulated ammonium and nitrite (Chalk *et al.*, 1975; Venterea *et al.*, 2010). In a corn field in Iowa, Breitenbeck and Bremner (1986) reported that N_2O emissions from soils injected with anhydrous ammonia at 30 cm was 107% and 21% greater than that injected at 10 cm and 20 cm, respectively.

In California specialty cropping systems, fertilizer placement often is independent of tillage but dependent on irrigation practices. For example, in tomato and almond cropping systems where drip irrigation (i.e. surface, subsurface drip irrigation or microsprinkler) is used, fertilizers are delivered with water. Therefore, it is difficult to ascertain the isolated effect of fertilizer placement on N_2O emissions from these systems.

3.1.4 Fertilizer application timing

It is a major challenge to synchronize soil N availability with crop demand when managing N fertilizer for crop production. Generally, crop N requirements are relatively low at seeding but increase markedly by several weeks after planting and decrease sharply as the crop approaches maturity (Eagle *et al.*, 2011). An active and well-developed root system can utilize the most fertilizer N if it is applied to meet crop requirements at a corresponding growth stage. By doing so, the potential for soil microbial and chemical processes to transform the applied N into N_2O and other mobile forms such as nitrate is reduced; this in turn may lead to less water pollution and indirect N_2O emissions.

Switching fertilizer application timing from fall to spring or from pre-plant to post-plant can reduce N_2O emissions from certain areas, such as those receiving high rainfall or irrigation (Matson *et al.*, 1998; Hao *et al.*, 2001; Hultgreen, 2003; Burton *et al.*, 2008). Optimization of N fertilizer application timing may not always result in direct reductions of N_2O emissions, in part due to the interactive effects of climate and crop conditions. Changing fertilizer application to side-dress and reducing the N application rate at each application event reduced soil nitrate accumulation when previous N applications met or were in excess of crop N requirement, but this did not significantly affect cumulative N_2O emissions (Zebarth et al. 2008a). Venterea and Coulter (2015) also showed that later in-season N application does not always reduce N_2O emissions in rainfed corn. Under California's Mediterranean climate, specialty crops and soils experience high variations in temperature and water availability, limiting application of results from cropping systems under other climates to specialty cropping systems in California. Additional field studies will elucidate how N_2O emissions from California specialty cropping systems respond to different fertilization timing.

3.1.5 Fertilizer efficiency enhancers

In recent decades, enhanced efficiency fertilizers, i.e. controlled- or slow-released fertilizers and inhibitors, have been developed with the goal of synchronizing soil N availability with crop requirement and decreasing N losses through nitrate leaching and N_2O emissions. These efficiency-enhancing fertilizer products include fertilizers coated with polymers, sulfur, or calcium magnesium phosphate, in which a physical barrier controls the release of plant available N, as well as nitrification inhibitors and urease inhibitors. Nitrification inhibitors suppress nitrifier activity over a certain period of time and prevent the oxidation of NH_4^+ to NO_2^- and consequently to NO_3^-; urease inhibitors slow down the rate of urea hydrolysis in the soil (Shaviv, 2001) and slow down the increase in soil NH_4^+ concentration.

Applying enhanced-efficiency fertilizers in cropping systems has received much attention in recent years. Several field studies have examined the possible effects of these fertilizer products on N_2O emissions (Oenema *et al.*, 2001; Dalal *et al.*, 2003; Akiyama *et al.*, 2010). In a study conducted in a potato cropping system in Minnesota, the application of polymer-coated fertilizer decreased N_2O emissions compared to a conventional split application of 270 kg N ha^{-1} (Hyatt *et al.*, 2010). Similarly, in California tomato cropping systems, polymer-coated fertilizer tended to be more effective when background N_2O emissions were higher and less effective under lower background emission rates (Burger and Horwath, 2012). In contrast, no change in N_2O emissions occurred with polymer-coated urea compared to conventional urea in a rainfed corn system in Minnesota (Venterea *et al.*, 2011).

Nitrification inhibitors can potentially decrease N_2O emissions from ammonia oxidation and denitrification. In a review that analyzed data collected from 85 field studies worldwide, Akiyama et al (2010) observed that nitrification inhibitors significantly reduced N_2O emissions by an average of 38% compared with conventional fertilizers, and their effectiveness was consistent across inhibitor types, fertilizer types, and land uses (e.g. grassland and upland). In addition, urease inhibitors were less effective in reducing N_2O emissions compared to nitrification inhibitors, though the application of urease inhibitors delayed the formation of NH_4^+ in soil and reduced final N_2O production when the release of NH_4^+ is synchronous with plant NH_4^+ uptake. in California tomato cropping systems where subsurface drip irrigation has been gradually adopted (Mitchell *et al.*, 2012), the application of nitrification inhibitor has had no impact on annual N_2O emissions (data not published), most likely because of low background N_2O emissions from these systems. To date, no field trials have been conducted to investigate the effect of urease inhibitors on N_2O emissions from California.

Work by Venterea et al. (2016) in a corn system in Minnesota suggests that combining N management practices such as improved timing and inhibitors, i.e. the '4R' approach of using the 'right rate, right source, right timing, and right placement', will reduce the required N fertilizer inputs and result in less N_2O emissions (Venterea *et al.*, 2016). More robust data across various soil types, regional climates, and management practices unique to California specialty crops will strengthen general conclusions about the effectiveness of nitrification and urease inhibitors.

3.2 Irrigation

Irrigation practices include:

- Flood irrigation: the practice of flooding the entire field with water
- Furrow irrigation: the practice of delivering water through furrows adjacent to crop beds
- Sprinkler irrigation: water is delivered to vegetation and the soil through a top-down approach
- Surface drip and subsurface drip irrigation: water is supplied through drip lines placed adjacent to crop rows or targeted to the root zone using buried pipes and tubing
- Subirrigation: the practice used in areas with relatively high water tables or where the water table can be artificially raised to allow the soil to be moistened from below the root zone

Innovative irrigation practices have emerged recently in response to California's Mediterranean climate and intense drought. Here, the role of irrigation in N_2O emissions is examined briefly. During irrigation, soil moisture approaches saturation and oxygen diffusion is limited. This results in anoxic conditions that promote N_2O production (Zhu *et al.*, 2013a), but once the soil is completely flooded only trace amounts of N_2O can be emitted to the atmosphere. At that point, the produced N_2O is trapped in the water and is further reduced to N_2 before diffusing to the atmosphere (Davidson, 1991; Dunfield *et al.*, 1995). Optimal moisture conditions for N_2O production via denitrification and nitrification have been identified at a water filled pore space of 70-90% and <70%, respectively (Dobbie *et al.*, 1999; Bateman and Baggs, 2005; Venterea *et al.*, 2010). A large amount of N_2O produced from ammonia oxidation pathways has also been observed under very low oxygen levels (i.e. 0.5% oxygen) soil (Zhu *et al.*, 2013a), indicating their role as a contributor to N_2O emissions. Under California's Mediterranean climate, irrigation is an integral practice in specialty cropping systems due to dry and warm summers. In this climate, most agricultural soils experience cycles of wetting and drying, which typically stimulate microbial activity and lead to large N_2O pulses through denitrification and ammonia oxidation (Kieft *et al.*, 1987; Rudaz *et al.*, 1991; Fierer and Schimel, 2002).

In flood irrigation systems, soils temporarily experience highly anoxic conditions, thus promoting denitrification. As indicated, high denitrification rates do not necessarily indicate high N_2O emissions as extremely anoxic conditions can lead to complete denitrification (Firestone *et al.*, 1979; Bonin *et al.*, 1989). For example, less N_2O is emitted from continuously flooded plots than from intermittently flooded plots in rice systems (Katayanagi *et al.*, 2012; Xu *et al.*, 2013). In furrow irrigation systems, near-saturation conditions occur transiently for one or more days after irrigation. A large pulse of N_2O and CO_2 often occurs after the first irrigation event, with additional pulses after subsequent irrigation events dependent on substrate availability (Davidson, 1992; Fierer and Schimel, 2002). Spatial variability of N_2O emissions from furrow irrigation systems can be very high. For example, significantly higher N_2O emissions were observed from furrows compared with adjacent beds in California tomato cropping systems (Burger and Horwath, 2012). Spatial and temporal heterogeneity in environmental conditions, variation in frequency and volume of irrigation used, and resulting flux rates make it difficult to quantify N_2O emissions from furrow irrigation systems with high levels of accuracy and precision.

Compared with flood and furrow irrigation, low-volume irrigation systems (i.e. surface, subsurface drip, and microsprinkler techniques) usually have lower N_2O fluxes (Nelson and Terry, 1996; Kallenbach *et al.*, 2010; Burger and Horwath, 2012). California specialty crop growers continue to adopt these irrigation systems due to their higher water use efficiency (Mitchell *et al.*, 2012). In subsurface drip irrigation systems, the soil surface is usually dry and soil moisture near the drip line is maintained between 20% and 30% WFPS between irrigation events. Evaporative losses are minimized, and the entire soil profile is rarely saturated (Hanson *et al.*, 2000; Hanson and May, 2007). Subsurface drip irrigation reduced N_2O by 1.95 kg N $ha^{-1}yr^{-1}$ compared to furrow irrigation in a California tomato system (Kallenbach *et al.*, 2010) (Table 1). In a meta-analysis of water management and fertilization in Mediterranean climates, Aguilera *et al.* (2013) reported that cumulative N_2O emissions from furrow irrigation and drip irrigation systems were 7.8 and 2.0 kg N ha^{-1} yr^{-1}, respectively. The impacts of sprinkler irrigation and surface drip irrigation on N_2O fluxes are expected to be similar. However, N_2O emissions were lower during a fertigation event with a microsprinkler system than those from a surface drip irrigation system in California almond orchards (Smart *et al*. 2011, Alsina *et al*. 2013). Compared to sprinkler and surface drip systems, fewer N_2O pulses are expected in subsurface drip irrigation systems due to less temporal variation in soil moisture (i.e. wet-dry cycles). However, studies with side-by-side experimental comparisons of subsurface drip irrigation and microsprinkler/surface drip irrigation are rare.

In subirrigation systema, since water is supplied to roots from below, evaporative losses are lower than they would be with surface irrigation systems. However, this system may increase N_2O emissions by saturating the soil profile (Elmi *et al.*, 2003; Munoz *et al.*, 2005). In California specialty cropping systems, subirrigation is rarely used due to the low water tables, particularly in recent drought years (Cabrera *et al.*, 2014; Howitt *et al.*, 2014).

3.3 Tillage

Tillage practices include:

- Conventional (full tillage): represents the greatest level of disturbance, with little to no crop residue left on the soil surface.
- Conservation (reduced tillage): classified as intermediate disturbance of the soil, with significantly more crop residue left on the soil surface compared to intensively tilled soils.
- No tillage (no-till): characterized by the use of seed drills and fertilizer or pesticide applicators with no additional tillage events or implements

Tillage is a fundamental practice in agricultural management that changes soil chemical, physical and biological characteristics relevant to seedling germination and plant growth. The effects of tillage practices on soil N_2O emissions are due to changes in soil aeration status (Six *et al.*, 2004; Munkholm *et al.*, 2016) and microbial activity (Broder *et al.*, 1984; Smith *et al.*, 2010). The effect of conservation tillage and no-till relative to conventional tillage on soil N_2O emissions has received much attention. N_2O emissions vary widely in response to conservation tillage and no-till, ranging from increases, decreases, and no change (Grandy *et al.*, 2006; Mosier *et al.*, 2006; Rochette, 2008). These contrary results have been attributed to differences in climate regime, N fertilizer placement, duration of tillage practice, and soil texture. By comparing N_2O emissions from 25 field studies under conventional tillage or no-till practices, Rochette (2008) concluded that the influence of tillage on N_2O emissions depends foremost on soil texture. For example, in a poor-drained fine-textured soil, no-till increases N_2O emissions compared to conventional tillage, while the opposite occurred in well-drained soil. Many studies have also reported that the conversion from conventional tillage to no-till increases N_2O emissions (Almaraz *et al.*, 2009; Kong *et al.*, 2009; Venterea *et al.*, 2011; Abdalla *et al.*, 2013). However, lower N_2O emissions have been observed from no-till compared to conventional tillage in temperate climates after adopting no-till practices for a certain amount of time (>10 years) (Six *et al.*, 2004; Kessel *et al.*, 2013). The overall impact of no-till management on N_2O emissions is therefore highly dependent on the duration of tillage adoption (Kessavalou *et al.*, 1998; Six *et al.*, 2004; Kessel *et al.*, 2013). No-till practices are considered integral to achieving California's air quality standards regulating airborne particulates smaller than 10 μm (California Air Resources Board 2002) and could be mandated in the future to reduce the environmental impact of agriculture.

3.4 Cover crops

> Cover crops
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> Cover crops include numerous crops that are typically planted, grown and returned to the soil through mowing or tillage (Culman *et al.*, 2014). Cover crops provide many services, including improving nutrient retention soil organic carbon content, improved water infiltration, and aggregative stability and reducing erosion.

Cover crops suppress weeds (Shipley *et al.*, 1992; Wolfe, 1997; Matthiessen and Kirkegaard, 2006), and serve as 'inter-crops' to fix N and build soil organic matter (Kallenbach *et al.*, 2010). The mineralization and release of nutrients from cover crops and other organic amendments into plant available forms is generally slower compared to synthetic fertilizers, allowing for immobilization by soil microorganisms, uptake by plants, increased soil organic matter, improved nutrient use efficiencies, and reductions in leaching and nutrient loss (Crews and Peoples, 2004; Seiter and Horwath, 2004; Drinkwater and Snapp, 2007). The planting time and functionality of cover crops can also vary with cropping system and climate (Snapp *et al.*, 2005).

Cover crops have been associated with increases in N_2O emissions from soils in

Mediterranean climates (Steenwerth and Belina, 2008; De Gryze *et al.*, 2009; Kallenbach *et al.*, 2010; Smukler *et al.*, 2012). In a meta-analysis of 26 published studies, including 106 observations of cover crop effects on N_2O emissions from the soil surface, Basche et al. (2014) reported that over half of the studies observed higher N_2O emissions from soils under cover crops than without cover crops. The incorporation of cover crops into the soil also promoted N_2O emissions in the short term (< 4 weeks). This is because cover crops add a substantial amount of labile C and N to soil, thereby decreasing oygen availability due to microbial respiration and increasing the microbial activity responsible for N_2O production (Varco *et al.*, 1987; Aulakh *et al.*, 1991; Follett, 2001; Watson *et al.*, 2002; Christopher and Lal, 2007; Sainju *et al.*, 2007). Nevertheless, Basche et al. (2014) also found that the temporal increases of N_2O emissions observed during cover crop decomposition were balanced by the periods when cover crops decreased emissions, underscoring the importance of long-term (i.e. entire year) monitoring to quantify the effect of cover crops on N_2O emissions.

Poor synchronization of mineralized N availability from cover crop decomposition and uptake by subsequent crops can lead to increases in soil N_2O production. Excess N, combined with available soil C, can increase N_2O derived from heterotrophic denitrification and ammonia oxidation pathways (Smid and Beauchamp, 1976; Firestone *et al.*, 1979; Stark and Firestone, 1995; Zhu *et al.*, 2013a). These effects vary with cover crop type. Leguminous residue may increase available soil N and subsequent N_2O emissions or, after harvest of the cash crop, non-leguminous species may rapidly take up surplus N in soil depending on ther phenology and have a greater chance of reducing N_2O emissions (Basche *et al.*, 2014; Culman *et al.*, 2014). Future studies focused on empirical data collection are needed to examine this hypothesis.

In California specialty cropping systems, the benefits of adopting cover crops are gradually being recognized and the effects of cover crops on N_2O emissions have been examined in tomato cropping systems and vineyards (Belmonte et al. in press) (Steenwerth and Belina, 2008; Kallenbach *et al.*, 2010; Garland *et al.*, 2014; Wolff, 2015). Wide adoption of cover crops still faces barriers due to the logistics and cost of operations and climate variation. The time between the end of the rainy season and the ideal period for cover crop establishment is usually only 2-3 weeks; cover crop incorporation in annual systems must occur at least 4-6 weeks before the agronomic crop to avoid problems with seed germination and seedling growth caused by immobilization of N or other nutrients. The potential competition between cover crops and woody perennial crops for soil resources raises concerns in grapes and orchard systems (Celette *et al.*, 2009), though recent work in irrigated wine grape vineyards suggests a decoupling of competition between grapevines and annual cover crops (Guerra and Steenwerth, 2011; Steenwerth *et al.*, 2013; Steenwerth *et al.*, 2016).

4. Contribution of changes in soil carbon stocks to greenhouse gas emissions

Changes in soil C stocks reflect the balance between the rates of SOC formation and decomposition, and determines the amount of CO_2 released to the atmosphere from agricultural soils (Breemen *et al.*, 1990). Croplands play a key role in soil-atmosphere C cycling, including preservation of SOC to enhance soil quality and sustain nutrient supply to cropping systems. Management strategies to increase the potential for soil C sequestration in cropping systems have been promoted as a partial means to mitigate agriculture's effect on atmospheric GHG levels and reduce its effect on global climate change (Lal *et al.*, 1998; Robertson *et al.*, 2000; Mosier *et al.*, 2006). Such practices include conservation tillage, crop rotation/residue management systems, and efficient management of nutrients and water. Policy makers and scientists still face a challenge to develop and implement effective GHG abatement strategies for agriculture. To address this challenge, development of best management practices for each cropping system that meet both production and mitigation goals will be enhanced by delineating the mechanisms and magnitude of C sequestration (De Gryze *et al.*, 2004; Six *et al.*, 2004). For an overview of these processes and the role of agricultural management practices in SOC sequestration, please see Appendix 2, which provides a summary of current knowledge on changes in soil C stocks as affected by management practices, including available research from California specialty crops.

5. Greenhouse gas emissions from California specialty annual crops

This section summarizes the literature on management practices (e.g. fertilization, irrigation and tillage) that influence GHG emissions from California annual cropping systems. These typical practices were captured as best as possible, based on 'best available data', in the development of the DayCent model for the crops described here. Nitrous oxide EFs reported under each practice were either obtained from the literature directly or were calculated based on the field data reported in the literature. We were not able to conduct a meta-analysis of available N_2O emission data on California specialty crops due to the limited number of studies, i.e. only 18 studies were identified in five specialty crop systems. Therefore, we reported N_2O EFs (both uncorrected and corrected for background flux) and estimated state level emissions for each specialty cropping system under different management practices.

5.1 Method for estimating EF and state-wide emissions

Emission Factors: Calculation Method

1. Emission factor (EF) is defined as the percentage of input N emitted as N_2O.

   EFs (%)= (N_2O-N / input N) ×100

2. In this report, if the N_2O background (no fertilizer input) emission is available in the literature, the EF calculation is adjusted as follows: adjusted EFs (%)= [(N_2O-$N_{fertilizer\ treatment}$ − N_2O-$N_{background}$) / input N] ×100.

3. If a study does not investigate N_2O background emissions, N inputs from unharvested crop residue (as recommended by IPCC guidelines), in addition to fertilizer N, are included in the EF calculation: adjusted EFs (%)= [N_2O-$N_{fertilizer\ treatment}$ / (input fertilizer N + crop residue N)] ×100. This value is reported as 'background corrected EF'.

4. Emission factors that are not corrected for the background flux are also reported.

5. The N_2O emission baseline for each cropping system is defined as N_2O emission under standard practices, i.e. furrow irrigation, conventional tillage, and recommended N rate.

6. The EFs used to calculate the state-wide emissions are averaged from the EFs under the same practices (regardless of fertilizer rate) that are reported in the literature. If studies investigated more than one fertilizer N rate (excluding zero fertilizer rate) or fertilizer type, the EF is calculated for each treatment and the mean of the EFs is averaged across these treatments.

7. State-wide emissions are calculated by applying the same EF across all the planting area (determined by CDFA Agricultural Statistics Review or USDA-NASS report) regardless of soil types and microclimates by assuming the cropping systems are well-managed and growers use the N application rates recommended by USDA-NASS.

5.2 Tomatoes

California produces one-third of the fresh tomatoes and 96% of the processed tomatoes in the U.S., with a total value of over $2 billion in annual farm cash receipts (Starrs and Goin, 2010; USDA, 2014b). California produces fresh-market tomatoes and processing tomatoes on approximately 12,140 and 121,400 hectares, respectively (CDFA, 2016). In the summer months, the Central Valley is the main production area, while in the spring and fall, Southern California supports significant production. Both processing and fresh-market tomatoes are available in many varieties with a diversity of fruit shapes and sizes, flavors, shelf lives, disease tolerances, and climatic requirements (Le Strange *et al.*, 2000). Here, we focus on the management practices that are commonly adopted in California tomato cropping systems, summarize the N_2O EFs under these practices based on published studies, and assess the GHG inventory of tomato cropping according to the EFs and planting acreage in California.

5.2.1 Fertilization

Tomatoes have a high N requirement. For example, Hartz and Bottoms (2009) reported a mean aboveground biomass N accumulation of 296 kg ha^{-1} in processing tomato fields that were produced conventionally for high yields. Generally, a seasonal N application rate of 170 kg ha^{-1}, with the majority applied by early-season side dressing, is sufficient for optimum yield (Hartz *et al.*, 2008). Krusekopf *et al.* (2002) reported that a seasonal total of no more than 112 kg N ha^{-1} fertilizer was required to maximize tomato fruit yield in fields with soil nitrate content higher than 16 mg kg^{-1} prior to side dress application. However, such low seasonal N application rates would be insufficient to support high-yield production over long periods; this is because fruit removal at harvest results in a loss of 150 to 230 kg N ha^{-1} from the field (de C. Carmello and Anti, 2004;

Burger and Horwath, unpublished data). In California, the most commonly used fertilizers are UAN-32 (urea ammonium nitrate, 32% N) and CAN-17 (calcium ammonium nitrate, 17% N) in subsurface drip and furrow irrigation tomato systems (Miyao *et al.*, 2014a, b). The N application rates recommended by the USDA National Agricultural Statistics Service are around 108 and 178 kg N ha^{-1} yr^{-1} in well-managed fresh-market tomato and processing tomato cropping systems, respectively (https://quickstats.nass.usda.gov/), though the actual N application rate for these crops is likely 45 kg N ha^{-1} higher than the recommended rates (Rosenstock *et al*., 2013).

Burger and Horwath (2012) measured annual N_2O emissions under different N application rates, which ranged from zero to above recommended input quantities in tomato fields. They reported that the background uncorrected EFs ranged from 1.11±0.11 to 1.80±0.22 % when N application rates ranged from 75 to 300 kg N ha^{-1} (Table 1). Unexpectedly, applying N at a rate (75 kg N ha^{-1}) below that required (165 kg N ha^{-1}) to support optimal crop growth increased the EF (1.61 % vs. 1.11%). However, when the N_2O background flux was considered, the difference in EF between these two fertilizer application rates vanished. In a conventionally managed (standard tillage, furrow irrigation) tomato field, Kennedy et al. (2011; 2013) reported that 0.76±0.05 % (background uncorrected) or 0.64±0.04% (background corrected, Table 6) of the applied N was lost as N_2O when N was applied at a rate of 237 kg ha^{-1}. Averaging the EFs reported by these studies (5 observations) across different N fertilization rates, we obtain an EF of 0.79±0.11% (background corrected) from N fertilization in tomato cropping systems under standard tillage and furrow irrigation practices. This EF is comparable to the average overall EF of 0.5% for Mediterranean agriculture reported by Cayuela *et al.* (2017), in which the EF was calculated based on a meta-analysis and corrected for background flux. Assuming all tomato systems are well-managed and growers use the recommended N application rates (108 and 178 kg N ha^{-1} for fresh-market and processing tomatoes, respectively), the total annual N_2O emission is 149 ± 25.8 Gg CO_2eq (calculated using uncorrected EF), while 84.8 ± 11.8 Gg CO_2eq N_2O is emitted from the added N (calculated using corrected EF) in California tomato systems under standard tillage and furrow irrigation practices (Table 6).

5.2.2 Irrigation

In California, furrow irrigation has been commonly used for tomato production in the past. Currently, subsurface drip irrigation (SDI) is the primary irrigation method in the southern San Joaquin Valley and is increasing in other areas, but some tomato fields are still furrow irrigated in the northern San Joaquin Valley and in the Sacramento Valley (Mitchell *et al.*, 2012). Kennedy et al. (2013) reported that the annual EF of N fertilization was lower in tomato systems under SDI combined with reduced tillage practices (background uncorrected EF: 0.46%) than under furrow irrigation and standard tillage practices (background uncorrected EF: 0.76%). However, it is difficult to ascertain the effect of SDI on EF of N fertilization based on Kennedy et al. (2013)'s study because no SDI treatment with standard tillage was applied. In a cover crop and irrigation study, Kallenbach et al. (2010) compared the N_2O fluxes from tomato plots that received furrow irrigation and SDI. They observed that mean N_2O flux from the furrow irrigation treatment in rainy seasons tended to be higher than from SDI. When a cover crop was planted, the mean N_2O flux from furrow irrigation was two times higher than from SDI, suggesting that SDI offers the potential to decrease N_2O emissions from tomato systems under cover crop management. Using the mean value of N_2O fluxes provided by Kallenbach et al. (2010), we calculated the background corrected EFs from furrow irrigation and SDI treatments. We found that the corrected EFs of N fertilization under furrow irrigation and SDI are 2.65 and 1.89% under cover crop management, and 2.08 and

1.07% under no cover crop management. However, these values have a high uncertainty and are not reliable because the frequency of gas sampling in Kallenbach et al. (2010) was low and no samples were collected between September and December. As a result, the mean N_2O fluxes, and therefore EF, could be overestimated. Nevertheless, the comparison of EFs between furrow irrigation and SDI acquired from this study still can be used to estimate the EF under SDI and standard tillage management (Equation 1 in Appendix 3); this can be achieved by assuming that the difference in N_2O emissions between SDI and furrow irrigation is constant across studies and using the EF under furrow irrigation and standard tillage management reported in Kennedy et al. (2013).

Here, the background corrected EF of N fertilization under SDI and standard tillage practices is calculated as 0.41%, comparable to the EF of 0.51% under drip irrigation derived from a meta-analysis of N_2O emissions from cropping systems in Mediterranean climates (Cayuela *et al.*, 2017). Assuming tomato growers use standard tillage, SDI practice, and recommended N application rates of 108 and 178 kg N ha^{-1} for fresh-market and processing tomatoes, respectively, the total annual N_2O emission from California tomato systems will be approximately 75 Gg CO_2eq, 44 Gg CO_2eq of which will come from the N inputs (calculated using corrected EF). If all these systems are managed under standard tillage combined with furrow irrigation practices,149 Gg CO_2eq N_2O will be released, 84.8 Gg CO_2eq N_2O of which will come from the N inputs (calculated using corrected EF) (Table 6). Although no statistically significant difference was detected between SDI and furrow irrigation, N_2O emissions tended to be reduced in SDI compared to furrow irrigation (Kallenbach *et al.*, 2010), meriting further investigation of SDI's potential to reduce N_2O emissions in California agriculture.

5.2.3 Tillage

Tomato production systems in California rely heavily on tillage for bed preparation, weed control, and postharvest residue incorporation (Miyao *et al.*, 2008). They are one of the most tillage-intensive annual crops produced in California (Mitchell *et al.*, 2007). Intensive tillage and cultivation practices that are used throughout the tomato production season also help growers manage risks such as seedling pests (Mitchell *et al.*, 2012). These tillage and cultivation practices can be costly due to diesel fuel and equipment costs. Moreover, compare to reduced tillage, conventional tillage reduces soil C sequestration potential in tomato systems (Veenstra *et al.*, 2007; Mitchell *et al.*, 2015b). Incentive programs such as USDA NRCS's Environmental Quality Incentive Program (EQIP) have been launched to encourage tillage reduction. However, the majority of California tomatoes continues to be produced using traditional, multiple-pass tillage practices largely because these systems are familiar to growers and because they have provided historically reliable yields (Miyao *et al.*, 2008; Mitchell, 2009; Mitchell *et al.*, 2015b).

Conservation tillage practices such as no-till and strip-till are usually defined as management practices that reduce tillage intensity and soil disturbance to maintain 30% or more of the soil covered by residues from previous crops after seeding (Mitchell, 2009; Institute *et al.*, 2012). Currently, less than 5% of tomato fields use conservation tillage practices in California (Mitchell and Horwath, personal communication). A "minimum-tillage" approach, which reduces the total number of tillage passes but not necessarily the overall disturbance of soil, is now being used with SDI to control weed densities (Sutton *et al.*, 2006; Mitchell, 2009). Recent estimates from UC Cooperative Extension suggest that this minimum tillage approach has been implemented in 90% of SDI tomato acres in the central San Joaquin Valley (Mitchell and Miyao, 2012). Kong et al. (2009) reported that minimum tillage (5 to10 tractor passes) led to greater N_2O fluxes compared to standard tillage (12 to 15 tractor passes) across conventional, low-input, and organic systems. However, robust annual N_2O emissions could not be calculated because measurements were only conducted monthly. Since minimum tillage is now being used widely with SDI, we calculated the total N_2O emissions from tomato systems under minimum tillage practices by adopting the background uncorrected EF (0.46 ± 0.02%) reported by Kennedy et al. (2013). Therefore, if all California tomato systems were managed under minimum tillage combined with SDI practices, the total annual N_2O emission from these systems would be approximately 49±2.03 Gg CO_2eq, 33.3 Gg CO_2eq N_2O of which would come from the N inputs (calculated using corrected EF). If these systems were managed under standard tillage combined with SDI practices, 75.1 Gg CO_2eq N_2O would be released, 44 Gg CO_2eq N_2O of which would come from the N inputs (calculated using corrected EF) (Table 6). No study has been conducted yet to compare N_2O emissions from California tomato systems under different tillage practices combined with furrow irrigation.

5.2.4 Cover crops

Currently, most cover crops used in California tomato production systems are the relatively short-season, October- or November-seeded small grain crops (e.g. Trios 102, *Triticale* × *Triosecale*; Merced rye, *Secale cereale*; barley, *Hordeum vulgare*) and legumes (e.g. vetch, *Vicia sativa*). Generally, at least four weeks before tomatoes are transplanted, these cover crops are flail-mowed and disked into planting beds when they are grown as green manures, or they are chopped or burned down using herbicides and then left as a surface mulch (Mitchell and Miyao, 2012). The benefits of using cover crops to improve the sustainability and productivity of cropping systems while minimizing adverse environmental impacts such as soil C degradation have been well documented (Jarecki and Lal, 2003; Snapp *et al.*, 2005; Sainju *et al.*, 2007; Veenstra *et al.*, 2007; Schipanski *et al.*, 2014). However, cover crops are currently not widely used in tomato production systems in California due to farmer concerns about lost opportunity costs involved in foregoing cash crop income and uncertainties about water use (Brennan and Boyd, 2012; Mitchell *et al.*, 2015a).

Veenstra et al. (2007) reported that five years of cover cropping in California tomato/cotton rotation systems increased soil C content in the top 30 cm soil by 4.0 to 4.9 Mg C ha^{-1} compared to the absence of cover crops. Mitchell et al. (2015b) also observed that cover cropping sequestered 0.46 to 0.63 Mg C ha^{-1} yr^{-1} more SOC than no cover cropping. However, this benefit can be offset by subsequent increases in N_2O emissions. For example, Kallenbach et al. (2010) found significantly higher N_2O fluxes under cover cropping practices than without cover crops. The EFs (background corrected) of N input that we calculated based on their study are 1.89 to 2.65 % under cover crop management and 1.07 to 20.8 % without cover crops. As discussed above, these values cannot be directly used to calculate the total N_2O emissions from tomato systems under cover cropping management. Assuming that the difference in N_2O emissions between cover crop and no cover crop is constant across studies, the EF of N input under cover cropping management can be calculated using equation 2 (see Appendix 3).

The EF of N input under cover crops combined with SDI and standard tillage practices is calculated as 1.01 % (background uncorrected) and 0.72% (background corrected) according to equation 2 (see Appendix 3), while the EF of N fertilization is 1.41% (background uncorrected) and 1.01% (background corrected) under cover crops combined with furrow irrigation and standard tillage practices (Table 6). Therefore, if all California tomato systems were managed under cover crop combined with SDI and standard tillage practices, the total annual N_2O emissions from these systems would be approximately 108 Gg CO_2eq (calculated using uncorrected EF), 72% of which would be produced from the N inputs (calculated using corrected EF). If all California tomato systems were managed under cover crop combined with furrow irrigation and standard tillage, the total annual N_2O emissions from these systems would be approximately 151 Gg CO_2eq, 72% of which again would be produced from the N inputs (Table 6). Assuming the soil sequesters C at a rate of 2.94 Mg CO_2 ha^{-1} yr^{-1} in tomato systems under cover cropping practices (Veenstra *et al.*, 2007), the global warming potential is -285 Gg CO_2eq when SDI and standard tillage practices are adopted and -242 Gg CO_2eq when furrow irrigation and standard tillage practices are adopted. However, caution should be used when scaling these values to a national level because no fuel consumption during implementation of these practices was considered in the calculation and also because soil C sequestration potential is largely dependent on climate, soil texture, initial soil C content, and the quality of C inputs (Halvorson *et al.*, 2002; Sainju *et al.*, 2007; Chambers *et al.*, 2016). More studies are needed to provide robust data to estimate the benefits and tradeoffs associated with adoption of cover crop management in California tomato systems in aspects of soil C storage, nutrient availability, and GHG emissions.

5.3 Strawberries

California produces about 90% of the strawberries in the U.S. with a total value of more than $2 billion in annual farm cash receipts (Starrs and Goin, 2010). Total California strawberry acreage was reported at 13,400 hectares for 2016 and most are located in the coastal areas of Central and Southern California. Strawberries are considered as California's most valuable annual crop due to their high value and low acreage. Strawberries prefer a cool coastal climate, and warm weather can shorten the growing cycle and promote pests and diseases. In modelling studies, Lobell and Field (2011) predict that by 2050, 10% of current California strawberry's yield will be reduced by the warming weather, while Deschenes and Kolstad (2011) predict that strawberry yields will decline 43% by 2070-2099. Direct GHGs emitted from strawberry cropping systems have not been published, but one ongoing project exists (Kortman et al., in process). Here, we summarize the management practices that are commonly adopted in California strawberry cropping systems. If necessary, the IPCC default factor is used to derive a base emission rate (Table 6).

5.3.1 Fertilization

Many researchers have reported that seasonal rates of 150 kg N ha^{-1} are sufficient to maximize fruit yield in an annual strawberry system (Hochmuth *et al.*, 1996; Miner *et al.*, 1997; Kirschbaum *et al.*, 2006). These studies report systems that produced fruit yields < 45 Mg ha^{-1}, unlike the strawberry systems in California, which typically produce over 50 Mg ha^{-1} each season (http://www.calstrawberry.com/csc/resource/industry-fact-sheets). Nitrogen fertility is managed by a combination of preplant application of controlled release fertilizer and N fertigation. Seasonal N fertilization rates currently range from less than 200 to more than 300 kg ha^{-1}, and the relative portion of N applied preplant vs. fertigation during the growing season varies widely among growers. The N application rate recommended by the USDA National Agricultural Statistics Service is around 110 kg N ha^{-1} yr^{-1} in California strawberry cropping systems (https://quickstats.nass.usda.gov/). Generally, a slow release fertilizer, 18-6-8, is drilled preplant in the bed using a fertilizer drill with bed shaper. During the growing season, growers apply various fertilizers and amounts through the drip system or as a foliar spray. The most commonly used fertilizers are CAN-17 (17-0-0-8Ca) and CN-9, an NPK fertilizer (16-20-0, 15-15-15, 20-10-15)

(University of California Cooperative Extension, 2011).

5.3.2 Irrigation

All strawberries in California are irrigated and the water demand varies from about 10 to 40 inches per year, with an average of 21 inches (Cahn, 2011). Water quality (salinity) and quantity are key factors affecting strawberry performance (Serrano et al., 1992; Levy and Christian-Smith, 2011). Although it is not a high-water demand crop, the water supply for strawberry is problematic since most strawberries are grown in the coastal areas where groundwater salinity is high. In California, drip irrigation is the most common practice used underneath plastic mulching; this reduces disease by keeping moisture away from the foliage (Commission, 1999).

5.3.3 Tillage

Strawberry production systems in California rely on tillage for bed preparation. Typically, the field is prepared disking, plowing, subsoiling and land leveling. The beds are listed, shaped, rolled, pre-plant fertilizer incorporated, irrigation lines buried and plastic mulch laid. After laying the mulch, roads are cut using a tracklayer tractor with blade to divide the field into smaller blocks, 280 to 400 feet long. The application of plastic mulch blocks gas exchange between soil and atmosphere, and therefore has the potential to change soil biochemical processes underneath the mulch. This makes the expansion of process-based model (i.e. DayCent) for California strawberry cropping systems problematic because the accommodation of the use of mulch in the model is currently not feasible. Therefore, new studies on the effect of plastic mulch application on GHG emissions and soil C sequestration will enable parameterization model for California strawberry cropping systems.

5.3.4 Cover crops

In California strawberry cropping systems, cover crops are generally planted on the furrow bottoms to maximize infiltration into the soil and reduce erosion (Brennan *et al.*, 2013; Smith *et al.*, 2016). Cover crops are usually planted in late October or early November and mowed or sprayed with herbicides before their height reaches the top of the bed and shades the strawberry plants. It has been reported that strawberry yields are not affected by well managed furrow-bottom cover crops (Smith *et al.*, 2016). Given that cover crops add a substantial amount of C and certain amount of N to the soil (Brennan and Boyd, 2012; Brennan *et al.*, 2013), it is crucial to understand the effect of cover cropping management on GHG emissions from California strawberry systems. Such information is still lacking.

5.4 Cool season vegetable –Lettuce

Lettuce is a cool-season crop that grows best with moderate daytime temperatures (22.8^{o}C) and cool nights (7.2^{o}C) (Turini *et al.*, 2011). California produces around 71% of the head lettuce in the U.S. with a total value of nearly $2 billion in 2013 (USDA ERS: Vegetable and Pulses Data 2015). According to the 2012 USDA Census of Agriculture, lettuce was produced on 131,000 hectares and the major production areas in California are the Central Coast (Monterey, San Benito, Santa Cruz, Santa Clara, and San Luis Obispo Counties), the southern coast (Santa Barbara and Ventura Counties), the Central Valley (Fresno, Kings, and Kern Counties), and the southern deserts (Imperial and Riverside Counties). Production is highest in Monterey County, which account for over 70% of the state’s supply and generates $1.48 billion, followed by Imperial at $158 million, and then Santa Barbara at $107 million. Lettuce grows best in lighter-textured soils because these soils provide good drainage during cold weather and warm up readily. In some areas such as the Central Coast and Central Valley, lettuce can be grown on heavy clay soils as long as there is good soil structure and adequate drainage.

5.4.1 Fertilization

Based on a survey carried out in the coastal valleys of central California, Hartz et al. (2007) reported that the average seasonal N fertilizer application rate was around 184 kg N ha^{-1}, ranging from 30 to 437 kg N ha^{-1}. Fall application of N fertilization is not recommended due to the risk of leaching by the winter rains. Generally, small quantities of N (about 22 kg N ha^{-1}) are applied before or at planting, and 56-90 kg ha^{-1} of N is side-dressed into the beds. One or more additional side-dressings are common, typically several weeks apart. Generally, 11 to 17 kg ha^{-1} of N is applied 7 to 10 days prior to harvest to ensure that the crop color and growth rate are acceptable. The fertilization practices usually vary among growers and locations. The N application rate recommended by the USDA National Agricultural Statistics Service is about 166 kg N ha^{-1} in California lettuce cropping systems (https://quickstats.nass.usda.gov/). Liquid fertilizers such as UAN-32 are most commonly used in drip irrigation systems. Composted manures and yard wastes are used to maintain soil structure in lettuce cropping systems by some growers. Application rates are typically around 9 tons ha^{-1}.

In a greenhouse study, Pereira (2014) observed that when N fertilizers were applied between 0 to 225 kg N ha^{-1}, measured growing seasonal N_2O emissions ranged from 0.16 to 1.15 kg N ha^{-1}. The scaled annual N_2O emissions ranged from 0.27 to 1.92 kg N ha^{-1}; as a consequence, the annual EFs (background corrected) of N input calculated using their data range from 0.34 to 0.56 % (Table 2). Burger and Horwath (2012) measured annual N_2O emissions under different N application rates which ranged from 84 to 336 kg N ha^{-1} in lettuce fields. They reported annual N_2O emissions between 0.64 to 1.47 kg N ha^{-1}, while the annual background uncorrected EFs of N inputs reported in their study ranged from 0.44 to 0.76 % and the annual background corrected EFs ranged from 0.41-0.65%. Averaging the EFs from these studies (9 observations) across different N fertilization rates, we acquire an averaged background uncorrected EF of 0.75±0.27 % and background corrected EF of 0.61±0.04 % from N inputs in lettuce cropping systems under no cover crops, standard tillage and subsurface irrigation practices. Assuming all lettuce systems are well-managed and growers use the recommended N application rates (166 kg N ha^{-1}), the total annual N_2O emissions from California lettuce systems is 76.3±27.5 Gg CO_2eq (Table 6) and the N inputs produce 62.1 ± 4.07 N_2O (calculated using corrected EF) under these practices.

5.4.2. Irrigation

The irrigation used in lettuce cropping systems varies widely. In the inland deserts, furrow irrigation is commonly used (Smith *et al.*, 2011), while on the Central Coast, growers mostly use sprinkler or drip irrigations. Since 2006, surface-placed drip irrigation has been increasing and now accounts for more than 30% of lettuce in the Salinas Valley. No studies are yet available to compare the GHGs emissions from lettuce cropping systems under different irrigation practices.

5.4.3. Tillage

In California lettuce cropping systems, tillage mostly occurs at land preparation, and includes disking, subsoiling, chiseling, leveling land, and preparing the seed beds (Tourte and Smith, 2010). No tillage is used during the growing season. No studies have been conducted to compare the effect of different tillage practices on GHGs emissions from lettuce cropping systems.

5.4.4 Cover crops

Growers often plant cover crops such as Merced rye on a portion of their acreage during each production year to build soil organic matter and improve soil structure. Unlike in tomato cropping systems where the application of cover crops dramatically increases N_2O emissions and therefore EFs (Kallenbach *et al.*, 2010), Suddick and Six (2013) reported an annual background uncorrected EF of N inputs in the lettuce systems under cover crops of 0.42%, comparable to the EF (0.44%) measured by Burger and Horwath (2012) under a similar N application rate (260 vs. 252 kg N ha^{-1}) but without cover crops in the lettuce cropping system. However, side-by-side trials to compare N_2O emissions between cover cropped and non-cover cropped systems are needed to address the influence of cover crop management on total annual GHG emissions from California lettuce systems. Assuming all lettuce systems are well-managed and growers use the recommended N application rates (166 kg N ha^{-1}), the total annual N_2O emissions will be 42.8 Gg CO_2eq, 32.6 Gg CO_2eq of which will come from the N inputs (calculated using corrected EF). This calculation assumes that these systems are managed using cover crops, standard tillage and subsurface irrigation practices (Table 6).

5.5 Other cool season vegetables, broccoli, cauliflower, and cabbage

The three most important cultivars of *Brassica oleracea* in the U.S. are broccoli, cauliflower and head cabbage. Together, they are named *cole crops*. California produces about 90% of the nation's broccoli and cauliflower (Starrs and Goin, 2010) and 20% of the nation's cabbage, with a total value of around $280 million in 2012 (National Agricultural Statistics Service, 2014). In California, broccoli is produced on 49,000 hectares, while cauliflower and cabbage are produced on 13,200 and 5,140 hectares, respectively. These three crops are grown in many locations around the Central Coast, San Joaquin Valley, the Central Valley, the southern coast, and the southern desert. Their optimal temperature ranges from 18.3-20ºC for cauliflower (Koike *et al.*, 2009) and 15.6-18.3ºC for broccoli and cabbage (Daugovish *et al.*, 2008; Le Strange *et al.*, 2010b). The amount of GHGs emitted from cole crop systems in California or areas with a Mediterranean climate has not yet been reported in the literature. We therefore only summarize the management practices that are commonly used in California cole crop systems. If necessary, the IPCC default factor is used to derive a base emission rate (Table 6).

5.5.1 Fertilization

Cole crops have high nutrient demand. In broccoli and cauliflower cropping systems, 22-34 kg N ha^{-1} of N fertilizer is usually applied before or at planting; 56-90 kg N ha^{-1} of N fertilizer is applied in the beds at the first sidedress (Koike *et al.*, 2009; Le Strange *et al.*, 2010a). One or more additional sidedressings are common. Fertilization practices usually vary widely. Liquid fertilizers are generally used in drip irrigation systems in these cropping systems. In cabbage cropping systems, growers in the southern desert usually broadcast ammonium phosphate at 25 kg N ha^{-1} before listing the beds and 67-90 kg N ha^{-1} fertilizer is applied as a sidedressing. However, growers in coastal areas usually apply 84 kg N ha^{-1} before planting. When the plants have five to six true leaves, 29 kg N ha^{-1} ammonium nitrate is directly sprayed and 12 kg N ha^{-1} ammonium nitrate or calcium nitrate is applied at midseason (Daugovish *et al.*, 2008). The N application rate recommended by the USDA National Agricultural Statistics Service is about 204, 228, and 196 kg N ha^{-1} in California broccoli, cabbage, cauliflower cropping systems, respectively (https://quickstats.nass.usda.gov/).

5.5.2. Irrigation

Water limitations are a major concern for cole crops in California. Broccoli and cauliflower are mostly irrigated with furrows and overhead sprinklers. Many growers use sprinkler irrigations through seed emergence or to set transplants, then switch to furrow or drip irrigation for the rest of growing period. Drip irrigation is not commonly used during the summer in the Central Valley, but in the Central Coast the usage of drip irrigation has been increasing (Koike *et al.*, 2009; Le Strange *et al.*, 2010b). In cabbage cropping systems, most growers use solid-set or hand-move sprinklers to germinate seed or establish transplants, then switch to furrow or surface drip irrigation (Daugovish *et al.*, 2008). All these crops are moderately salt-sensitive (Shannon and Grieve, 1998), so degradation of groundwater quality will become an increasing concern.

5.5.3. Tillage

In broccoli and cauliflower cropping systems, primary tillage, including disking, rolling, subsoiling, land leveling, and listing beds, occurs in June of the planting year. Usually, fields are disked and rolled two times, ripped, disked and rolled two more times, landplaned with three passes, chiseled three times, and disked two more times. Following the tillage operations, a custom operator lists the 38-inch beds (University of California Cooperative Extension, 2012). In cabbage cropping systems, the types of tillage used vary among fields and growers. For example, growers in Ventura County use a tillage regime that includes 15-18 land preparation operations, incorporates most crop residues and leaves less than 30% of the surface covered by residues (Mitchell, 2009).

6. Greenhouse gas emissions from California specialty woody perennials: Fruit and nut crops

California farmland in orchard and vineyards is approximately 34% of the state farmland (UCAIC, 2009). California leads the nation in the production of the following woody perennials: almonds, apricots, dates, figs, grapes, kiwifruit, kumquats, lemons, limes, olives, peaches, pears, persimmons, pistachios, plums, pluots, pomegranates, and walnuts (NASS, 2011). Agricultural statistics indicate that woody perennials have occupied an increasingly larger proportion of the California landscape over the past few decades. This trend suggests that woody perennials play an important role in GHG emissions from California. However, detailed inventories on GHG emissions from these systems are rare and difficult to accurately quantify (Williams *et al.*, 2011). This is because studies that have examined GHG emissions from California perennials or the effects of management on these systems have only been done in wine grapes (Smart *et al.*, 2006; Steenwerth and Belina, 2008; Garland *et al.*, 2011, 2014; Verhoeven and Six, 2014; Wolff, 2015; Yu *et al.*, 2017) and almonds (Smart *et al.*, 2011; Suddick *et al.*, 2011; Schellenberg *et al.*, 2012; Alsina *et al.*, 2013; CalRecycle, 2015; Wolff, 2015). Only one study on GHG emissions from walnuts (Pereira, 2014) is available; no studies have examined GHG emissions from California pistachios, stone fruit trees, and citrus systems.

6.1 Grapes

California produces over 90% of the wine grapes and 99% of the raisin grapes in the U.S., putting it among the top raisin producers and top wine producers in the world (Tolomeo *et al.*, 2012). California grows grapes on a total of 376,000 hectares, with wine grapes and raisin grapes occupying approximately 249,000 and 77,700 hectares, respectively (CDFA, 2015). Wine grapes are the most widely distributed, while table and raisin grapes are typically grown in the southern part of the Central Valley and in the Coachella Valley (Elias *et al.*, 2015). In 2014, a total of

4,142,934 tons of grapes were harvested for a total value of over $3 billion, with wine grapes accounting for 57% of the total and table grapes and raisin grapes following behind at 23% and 20%, respectively (Tolomeo et al., 2012).

Here, we focus mainly on the management practices that are commonly adopted in California grape cropping systems, summarize the N_2O EFs of wine grapes under these practices based on published studies, and assess the GHG inventory of wine grape cropping systems according to the EFs and acreage in California. The EFs are reported as background uncorrected EFs since no corrections can be made based on either crop residue N input or control (no N input) treatment. Presently, field studies with GHG measurements for raisins and table grapes do not exist.

6.1.1 Fertilization

Unlike vegetable crops and other fruit crops, grapes comparatively do not require intensive N input (Peacock *et al.*, 1998). For example, the N requirement for raisin grape production in the San Joaquin Valley was 84 kg ha^{-1}, with approximately 35 kg ha^{-1} removed by the crop (Williams, 1987). This suggests that the annual N demand is approximately 25 to 50 kg ha^{-1} depending on crop size. Fertilization may not be necessary when high levels of NO_3^- are present in irrigation water, or when legume cover crops are grown. Excess N supply for grapes can be detrimental to vine growth and production, especially wine grapes, and can increase the potential for NO_3^- pollution and N_2O emissions. In California wine grape cropping systems, the most commonly used fertilizer is UAN-32 (Wunderlich *et al.*, 2015). The average application rate recommended by the USDA National Agricultural Statistics Service is around 33 kg N ha^{-1} yr^{-1} in all types of grape cropping systems, with 47, 25, and 52 kg N ha^{-1} yr^{-1} in raisin, wine grape, and table grape cropping systems, respectively (https://quickstats.nass.usda.gov/).

The effects of fertilizer rate on N_2O emissions from California wine grape vineyards have been examined on numerous occasions. In Napa County, Smart et al (2006) reported that N_2O emissions increased from 0.03 to 0.09 kg N_2O-N ha^{-1} yr^{-1} when N rates increased from 0 to 45 kg N ha^{-1}. Contrary to this emission rate trend, however, a higher EF was found at the lower N rate, i.e. by using the flux reported in Smart et al. (2006), the EF was calculated as 0.96 % when N applied at 6 kg N ha^{-1}, but 0.20 % EF at 45 kg N ha^{-1}. In a two-year study, Wolff et al. (2015) observed that annual N_2O emission rate was 0.13 kg N_2O-N ha^{-1} yr^{-1} in the first year and 0.49 kg N_2O-N ha^{-1} yr^{-1} in the second year in the grape cropping systems under standard tillage and without cover crop management; the EFs we calculated from these different tillage practices treatments was 0.47 % when vineyards received 8.4 kg N ha^{-1} synthetic fertilizer plus organic N from alley crop residue in the first year and 0.49% when vineyards received 16.8 kg N ha^{-1} synthetic fertilizer plus organic N from alley crop residue in the second year. In Garland et al (2014)'s study, the EF was reported at 10.4% when 5.4 kg N ha^{-1} fertilizer was applied. Averaging the EFs reported from these studies (3 observations) across different N fertilization rates gives an EF of 3.79% from N fertilization in grape cropping systems under standard tillage, no cover crops, and drip irrigation (Table 3, 6). Assuming all grape growers use recommended N application rates (47, 25, and 52 kg N ha^{-1} yr^{-1}in raisin, wine grape, and table grape cropping systems, respectively) and manage grape cropping systems under standard tillage combined with drip irrigation, the annual N_2O emissions from California grape cropping systems will be 205 Gg CO_{2eq} (Table 6).

6.1.2 Irrigation

Grapes are not as sensitive as many crops to drought. In wine grape cropping systems, drip irrigation is widely adopted, but in raisin and table grapes systems, furrow irrigation is still used very rarely in certain areas (Peacock *et al.*, 2000). Efficiency gains have been possible with irrigated grapes by transitioning table and raisin grapes from furrow irrigation to drip irrigation, and by better quantifying effective regulated or sustained deficit irrigation regimes for different types of wine grapes in different locations. Water quality, such as salinity in groundwater is also a concern for winegrowers, especially growers in the coastal regions. The mitigation potential of alternative irrigation practice cannot be estimated as no field study has compared N_2O emissions from different irrigation practices in California grape cropping systems. The majority of wine grapes and raisin grapes are drip irrigated. Thus, the annual N_2O emission from California vineyard under standard tillage combined with drip irrigation is 205 Gg $CO_{2eq.}$

6.1.3 Tillage

In California grape vineyards, tillage practices include standard (complete) tillage and conservation tillage or no-till (Hirschfelt, 2000). Under standard tillage, the alleys are cultivated with standard disks and harrows. A French plow or spring-hoe weeder in late winter or early spring can be used to control weeds in the vine row instead of herbicide. No-till practices combined with cover crops recently have been promoted in California vineyards, but potential drawbacks need to be considered before selection such as potential impacts on residual-N in soil and vine balance (Steenwerth et al. 2016). Garland et al. (2011) reported that N_2O emissions from vineyards under no-till practices were greater than standard tillage, with 0.12 kg N_2O–N ha^{-1} growing $season^{-1}$ emitted from standard tillage compared to 0.18 kg N_2O–N ha^{-1} from no-till. However, Wolff (2015) observed higher N_2O emissions from a standard tillage wine grape vineyard than in a reduced tillage system. Assuming all grape growers use no-till practices, cover crops, and recommended N application rates (47, 25, and 52 kg N ha^{-1} yr^{-1} in raisin, wine grape, and table grape cropping systems, respectively), the annual N_2O emissions from California grape cropping systems will be 187 Gg CO_{2eq}. If these systems are managed under standard tillage with cover crops, 162 Gg CO_{2eq} N_2O will be released (Table 6).

6.1.4 Cover crops

Cover crops are commonly grown in California wine grape cropping systems, but are found less commonly in table grapes and raisins due to limited winter precipitation in their growing regions. Growing cover crops in a vineyard can regulate vine growth by improving water penetration and soil fertility, and may also play a role in pest management (Hirschfelt, 2000; Guerra and Steenwerth, 2011). Perennial cover crops cannot be used in raisin vineyards when they are sun-dried on the vineyard floor instead of using the 'dried on the vine' system. Cover crops are not commonly grown in table grape and raisin systems as they exist in regions with low winter rainfall. Commonly used cover crop species include barley, oats, triticale, winter peas, vetch, bell beans, daikon radish, clover, rye and resident vegetation. In tilled systems, cover crops are generally planted in the fall and mowed and tilled into the soil in the spring when the ground can be easily cultivated. In no-till systems, vineyards are seeded with species that will reseed themselves on an annual basis and replanted as needed with a no-till drill. Thereafter, the cover crops are mowed in spring and early summer, residues remain on the soil surface to decompose. In a California winegrape vineyard, Garland et al. (2014) observed that the annual N fixed by the leguminous cover crop (approximately 47 kg N ha^{-1} yr^{-1}) led to 3.92 kg N ha^{-1} N_2O from soil, whereas the bare soil emitted 0.56 kg N ha^{-1} N_2O. Steenwerth and Belina (2008) and Wolff (2015) also observed that N_2O emissions were significantly higher in the cover crop treatment than in the no cover crop treatment in California vineyards. Assuming all grape growers use standard tillage, cover crops, and recommended N application rates (47, 25, and 52 kg N ha^{-1} yr^{-1} in raisin, wine grape, and table grape cropping systems, respectively), the annual N_2O emissions from California grape cropping systems will be 162 Gg CO_{2eq}. If all these systems are managed under standard tillage with no cover crops, 205 Gg CO_{2eq} N_2O will be released (Table 6).

6.2 Nut tree crops- Almonds, pistachios, and walnuts

California currently produces 100% of the nation's commercial almonds, 99% of the nation's commercial pistachios, and 99% of the nation's walnuts. In 2015, California almonds were grown on approximately 450,000 hectares, with a value of $5.33 billion (NASS, 2015). Almost all California almonds are grown in the Central Valley (Sacramento and San Joaquin Valleys). Pistachios in California are planted on approximately 126,000 hectares with a total crop value of $1.3 billion in 2014, while walnuts are grown on 148,000 hectares with a total value of $1.9 billion in 2014 (NASS, 2015). Pistachio production occurs mainly in the San Joaquin Valley (Starrs and Goin, 2010), and walnut production is concentrated in both the Sacramento and San Joaquin Valleys. Almonds require 200-400 annual chill hours to reach optimal yields, while the relatively high chilling requirement of pistachios and walnuts (800-1000 hours) is a cause for concern in a warmer climate.

6.2.1 Fertilization

Compared to grapes, nut tree crops have higher nutrient requirements. Generally, mature almond trees use 80% of their total annual N requirement between March and mid-May-June to reach maximum yield. Applications of soluble N fertilizers are most commonly split throughout the annual production cycle. The types of N fertilizer used include urea, ammonium and nitrate based fertilizers. In a two-year California almond orchard study, Schellenberg et al. (2012) observed that when N fertilizers were applied at 224 kg N ha^{-1}, N_2O annual emissions from UAN tended to be higher than CAN though not significantly different, with background uncorrected EFs of 0.23% and 0.35% for UAN and CAN, respectively. Wolff (2015) reported that both the fertilization frequency and type significantly influenced N_2O emissions from an almond orchard, with higher N_2O emissions caused by a high frequency of fertilization (336 kg N ha^{-1} split into 20 applications) compared to the standard application frequency (336 kg N ha^{-1} split into 4 applications), and higher N_2O emissions from UAN than from KNO_3. The N application rates recommended by the USDA National Agricultural Statistics Service are around 150, 131, and 122 kg N ha^{-1} for almonds, pistachios, and walnuts, respectively (https://quickstats.nass.usda.gov/). Averaging the EFs from published studies focusing on N fertilization, we obtain a background uncorrected EF of 0.43% and background corrected EF of 0.31% from N inputs in California almond cropping systems under drip irrigation (Table 4). Assuming all California almond orchards are managed under drip irrigation and farmers use the recommended N application rate of 150 kg N ha^{-1}, the total annual N_2O emissions from these systems will be 136 ± 10.7 Gg CO_2eq, 98 ± 12.6 CO_2eq (calculated using corrected EF) of which will come from the N inputs (Table 6).

In an organic walnut orchard, Pereira et al. (2016) reported annual N_2O emissions of 1.15-1.18 kg N ha^{-1} from tree rows and 1.29-2.41 kg N ha^{-1} from tractor rows. The EF calculated based on the emission data reported in their study range from 0.93-1.55% (Table 5). However, these EFs were not used to calculate the station-wide emissions from walnut orchard because the majority of walnut orchards in California are not organically managed. Therefore, we use the EFs from almonds cropping systems to calculate N_2O emissions from California pistachio and walnut cropping systems. Assuming all California pistachio and walnut farmers use drip irrigation and recommended N application rates, the total annual N_2O emissions from these systems will be 33.2±11.6 and 36.3 ±12.7 Gg $CO_{2eq,}$ respectively(Table 6). The N inputs in these two systems will produce 24.0 and 26.2 CO_2eq N_2O (calculated using corrected EF), respectively.

6.2.2 Irrigation

The use of micro-irrigation systems has been adopted by a majority of California nut tree growers (Lopus *et al.*, 2010), while the use of flood irrigation has greatly declined as water availability for agriculture in California has become more restricted. The most common micro-irrigation systems for nut tree crops consist of aboveground drip (conventional drip) and stationary microjet sprinklers. Schellenberg et al. (2012) reported that in a California almond cropping system under microsprinkler irrigation, the annual N_2O emissions were 0.53 and 0.80 kg N ha^{-1}, corresponding to EFs of 0.23 and 0.35% (background uncorrected) when UAN and CAN were applied, respectively. Compared to microsprinkler irrigation, significantly higher N_2O emissions have been found in almond cropping systems under drip irrigation (Alsina *et al.*, 2013; CalRecycle, 2015; Wolff, 2015). Averaging the EFs from published studies (5 observations) focusing on microsprinkler irrigation practice in California almond orchards, we acquire a background uncorrected EF of 0.25% and background corrected EF of 0.19% in this cropping system. Assuming all California almond orchard are well-managed under microsprinkler irrigation and farmers use the recommended N application rate of 150 kg N ha^{-1}, the total annual N_2O emissions from these systems will be 79 ± 15.8 Gg CO_2eq, 60.1 Gg CO_2eq of which will come from the N inputs (calculated using corrected EF). If these systems are managed under drip irrigation, 136± 10.7 Gg CO_2eq N_2O will be released, 98.0 Gg CO_2eq N_2O of which will come from the N inputs (calculated using corrected EF). Since almonds have similar irrigation demands as pistachios and walnuts, we assume pistachio and walnut cropping systems have the same EFs under microsprinkler and drip irrigation as almonds. We also assume that farmers use the recommended N application rates. Therefore, if all California pistachio cropping systems were managed under microsprinkler, the total annual N_2O emissions from these systems would be 19.3±3.86 CO_2eq, 14.7 Gg CO_2eq of which would come from the N input (calculated using corrected EF). If all these systems were managed under drip irrigation, 33.2 ±11.6 Gg CO_{2eq} N_2O would be released, 24.0 Gg CO_2eq N_2O of which would come from the N inputs (calculated using corrected EF). If all California walnut cropping systems were managed under microsprinkler, the total annual N_2O emissions would be 21.1±4.23 Gg CO_{2eq} , 16.1 Gg CO_2eq of which would come from the N inputs (calculated using corrected EF). If these systems were managed under drip irrigation, 36.6±12.7 Gg CO_{2eq} N_2O would be released, 26.2 Gg CO_2eq of which would come from the N inputs (calculated using corrected EF) (Table 6).

6.2.3 Tillage and cover crops

Generally, tillage can be kept to a minimum or no-till in nut tree cropping systems. At one time cover crops were only used in organic orchards but recently many larger ranches are experimenting by planting different cover crops. The most common cover crops grown in nut tree orchards are vetch, Blando bromegrass, mustards, and clovers (Fred Thomas *et al.*, 2011). To date, no side-by-side studies have compared N_2O emissions from California nut tree cropping systems as affected by the adoption of cover crops, except one study which reported N_2O emissions from organic walnut orchard under cover cropping (Pereira *et al.*, 2016).

6.3 Stone Fruit

California produces about 20% of the nation's sweet cherries, 70% of peaches and 100% of nectarines, 95% of apricots, 95% of fresh plums, and 99% of dried plums (Starrs and Goin, 2010). All of California's stone fruits have a similar irrigation demand of about 90-100 cm per year, but significantly differ in temperature requirement (Baldocchi and Wong, 2008). In 2013, California peaches and nectarines were grown on 18,600 and 262,000 hectares, with values of $278 million and $117 million, respectively (USDA, 2014). To date, no studies have reported N_2O emissions from California stone fruit systems.

6.4 Citrus

California produces 30% of the nation's oranges, 90% of lemons, 48% of mandarins, and 30% of grapefruit (Starrs and Goin, 2010). In 2014-2015, California citrus were grown on 105,623 hectares, with a value of $1.9 billion. Most of California's citrus is located in the southern San Joaquin Valley. Citrus orchards have a moderately high annual irrigation demand of 34-36 inches per year in the San Joaquin Valley and somewhat less for orchards near the coast due to the cooler temperatures and fog. Citrus production can decrease with decreasing quantity and quality of irrigation water, but warmer temperatures are not likely of great concern. No studies have reported N_2O emissions from California citrus systems.

7. Development of the DayCent model and its application on California specialty crops

DayCent is an ecosystem model used to simulate C, N, P, and S dynamics and includes submodels for soil organic decomposition, plant productivity (crop/grass and tree submodels), and trace gas fluxes (Parton *et al.*, 2001). DayCent has been applied in many ecosystems around the globe to simulate SOC, N_2O emissions, NO_3 leaching, and plant productivity, including predicting SOC stock changes and N_2O emissions for the U.S. National GHG Inventory (EPA 2017) and estimating field-level SOC stock changes and N_2O emissions in the online COMET-Farm tool (www.comet-farm.com). The goal of the work presented here was to predict the GHG benefits of conservation practices in specialty crop systems and add a number of economically important specialty crops into the COMET-Farm system. Prior to this effort, about 65% of the acreage of California cropland could be modeled in the COMET-Farm system. After this effort, more than 85% of the acreage of California cropland now can be modeled. Several new woody and non-woody crops were modeled as part of this effort. In this section, we illustrate the process and results of modeling wine grapes, almonds, and lettuce, to demonstrate the variety of modeling challenges addressed to fulfill the project requirements.

Briefly, we describe the DayCent model development and testing process for specialty crops; documentation will be forthcoming in the scientific literature. The model parameterization effort typically involves developing model input parameters for plant production, resource partitioning, C to N ratios, and accumulated biomass C stocks based on published literature. We run the DayCent model for experimental sites with sufficient site and management information, and statistically compare the DayCent model C pool predictions against experimental observations. The goal is to achieve a 1:1 correspondence (slope of 1.0 with an intercept of 0) between measured and modeled plant production in each plant biomass C pool. Once we finish parameterizing a cropping system, we validate the new crop against an independent dataset containing some measure of plant production. As there are relatively few studies available for parameterizing the different C components of crops, we typically validate new crops against reported crop yields that represent the most geographically-diverse regions where crop yield data are available, to take into account regional differences in soils, climate, and practices. The most comprehensive datasets available for this work consist of yearly, county-average farmer reported yields from the National Agricultural Statistics Service (https://quickstats.nass.usda.gov/) and the California Department of Food and Agriculture (CDFA, 2017a).

The regional model validation process includes the following steps: 1) select a random, geographically diverse set of multiple point-based samples for each crop, using the NASS Cropland Data Layer (CDL) (https://quickstats.nass.usda.gov/) to determine crop locations; 2) construct DayCent model runs that correspond to typical, regional management practices for each crop; 3) run the DayCent model for each point using site-specific soil and weather data and regionally representative cropping systems; and 4) compare modeled crop yields for each point against the average crop yield reported for the particular year and for the county in which points were located. The mean crop yield of all DayCent-modeled points is compared to the mean yield reported in the independent dataset. The overall DayCent plant production parameter is adjusted somewhat up or down in order to achieve a 1:1 correspondence (or within 5%) of the two means. Any modifications to this general approach will be outlined in the sections for the modeled crops presented here.

7.1 Modeling woody crops in DayCent

For annual specialty crops (lettuce, cole crops, strawberries, etc.), no structural changes to the DayCent model were needed, but model changes were required to simulate the unique properties of orchards and vineyards. These model changes have previously been applied to CENTURY, the monthly time-step version of DayCent (Parton *et al.*, 1987; Parton *et al.*, 1998; Paustian *et al.*, 2012). The tree submodel in DayCent allocates C and N to five biomass pools:

leaves, fine roots, fine branches, large wood, and coarse roots. Because a large fraction of annual production is allocated to fruits and nuts (Lakso *et al.*, 2003), a new pool was added to the tree submodel for fruits/nuts, which is parameterized by C: N ratio, lignin content, death rates, time of flowering, time of harvest/removal, and growth allocation rate. Tree flowering date is specified by the user, after which a growing degree day model determines fruit/nut maturity. Additional model changes allow the user to harvest fruits/nuts, prune branches, thin fruit, and renew orchards/vineyards. Orchard/vineyard growers often maintain groundcover between tree rows. Tree crops and herbaceous groundcover can be grown simultaneously in DayCent and interact through competition for resources. In many modern orchard/vineyards systems, inputs, such as N fertilizer, are only applied to the tree rows. To better represent targeted placement of N fertilizer additions, we added a parameter that allows the user to apportion fertilizer N and soil available N according to areas of trees/vines and groundcover. No changes to the model were needed for other common management practices, such as irrigation, fertilization, organic matter amendments, and groundcover management (mowing, harvesting, etc.).

Several gaps in knowledge about biomass production and allocation in woody perennial systems presented difficulties for their parameterization in DayCent. Until recently, few researchers reported whole plant biomass measurements of orchards and/or vineyards. Worldwide, very few researchers have reported measurements of root biomass from orchard and vineyard systems. Developing and improving the plant growth submodels in ecosystem models like DayCent requires measurements of biomass accumulation and partitioning, and the quality of the models depend directly on the abundance and quality of the measurements used to derive input parameters for the models. Because of the relative lack of root biomass data in tree crops, we use root: shoot ratios documented by IPCC (2006) for woody systems.

In the section that follows we describe the effort to model three unique crops – wine grapes, almonds, and lettuce – as a snapshot of the larger effort.

7.2 Wine Grapes

Wine grapes systems present a unique challenge for DayCent model development due to the wide diversity of climates, cultural practices such as trellis systems, pruning styles, and floor management; and rootstocks and scion varieties, not to mention the wide selection of clonal material within a specific variety. To address this complexity, we leveraged the work by Amerine and Winkler (1944; 1974), who classified wine grapes into different production classes relative to their growing degree day requirements. We developed four wine production classes corresponding to this index, utilizing the overall DayCent biomass partitioning parameters for wine grapes developed from the Steenwerth dataset (see below). These classes are described in Table 7.

The DayCent model input parameterization for wine grapes was based on biomass accumulation and resource partitioning measurements. These consisted of multiple studies from the Napa, Lodi, North Coast, and Central Coast regions of California (K. Steenwerth, unpublished data; J. Williams, unpublished data; Williams et al. 2011) as well as a study by Morande et al. (2017). Together, these datasets provided the richest and most geographically and agronomically diverse biomass accumulation data of any crop in this effort. These data consisted of biomass measurements of grape stems, grape vines, and grape yield from twenty-one different wine production blocks and 7 different wine varieties. No measurements of root biomass were available from any studies evaluated in this effort. One measurement of leaf biomass was available for wine grapes, hence model performance for root and leaf C pools were not evaluated statistically.

Where trunk and cordon measurements were taken, we modeled biomass based on the diameter and height of the trunk combined with the diameter, length, and number of associated cordons (K. Steenwerth, unpublished data) as well as one destructive sample (Morandé *et al.*, 2017). We found no data or models that could help predict fruit or vine production relative to pruning strategy. Measurements were based on harvested fruit biomass and dormant-period vine prunings. DayCent slightly under-predicted trunk/cordon biomass (measured vs modeled comparison R^2 = 0.43, correlation coefficient of 0.62, and a slope of 0.93) (Figure 1). Grape growers employ innumerable vine and fruit thinning strategies to influence fruit quality, and pruning techniques in the dormant-period that influence yields for the next year. Also, management practices and weather of the preceding year can affect fruitfulness because clusters for the next year's crop form concurrently with leaf primordia in the compound bud (Williams 2000). These various strategies all influence grape yield and end-of-season vine biomass in different ways that are independent of ecosystem (soil quality, climate) and other management factors (irrigation, fertilization).

Vine biomass measurements examined in this study demonstrated the greatest variation in the C pools. DayCent slightly under-predicted the mean vine biomass accumulation. DayCent under-predicted fruit biomass by 15% (Figure 3). Measurements of fruit removal during thinning, when growers will adjust the crop load of the vine, and removal of suckers at the base of the vine and hedging to control canopy vigor were not available.

The relative performance of predictions into these C pools shed light into how management strategies can influence the ecosystem dynamics of orchard and vineyard systems. Biomass from suckering and fruit thinning during the growing season are typically left *in situ* (Steenwerth, personal communication). We know neither the significance nor the greenhouse gas consequences of these C and N inputs into the ecosystem. Their decomposition likely contributes a small amount of C and N into the soil and leads to minor nitrous oxide emissions. In the opinion of the authors, their influence is not likely to dominate soil C and N flux dynamics, however the potential dynamics involved may be worth investigating. Future measurements of their biomass and C: N ratios may help improve model performance.

The low fertilization rates of wine grapes compared to other commodities modeled in DayCent presented a challenge. Annual N fertilization in wine grapes is < 30 kg N ha^{-1} compared with 100-150 kg N ha^{-1} in table grapes, ~200 kg N ha^{-1} in almonds, and 50-250 kg N ha^{-1} in other commodity crops (CDFA, 2017b). Actual reported practices can be much higher, with some fertilizer application rates exceeding 400 kg N ha^{-1} (anonymous, personal communication). DayCent biomass predictions tend to correlate best in agricultural systems fertilized at 50 kg N ha^{-1} or higher (Paustian research group, unpublished data). We found no published biomass measurements from table or raisin grape systems.

To verify the DayCent model performance for crop production, we examined utilizing two independent measures of biomass production. The California Department of Food and Agriculture publishes yearly crop production data for the state of California through the California Agriculture Statistics Review (CDFA, 2017a). Unfortunately, the crop yield data published within this report was for wine grapes in total and was not separated by variety, and hence could not be used. A second dataset – identified in California as the "Crush Report" (https://www.nass.usda.gov/Statistics_by_State/California/Publications/Grape_Crush/Final/2016/201603gcbtb00.pdf)- contained information on wine grape yield by county and region within California, however the dataset focused on economic yield data. The data contained generally do not reflect the actual crop yield due to the dataset's structure and intended use by the wine grape industry.

These issues, combined with concerns about how vine and fruit manipulation affect crop yield, led to circumstances whereby we could not verify the grape plant production model parameters against a separate, independent dataset.

7.3 Almonds

Here, we present the work conducted on almonds, another widely grown woody perennial crop in California. Fewer datasets were available for parameterizing almonds compared with wine grapes. Four datasets had the required information available for modeling in this effort, including work in Kern County (Goldhamer *et al.* 2006; Esparza 1999; Esparza 2001; Hutmacher *et al.* 1994; Kendall *et al.* 2015; Marvinney *et al.* 2015). Model parameterization results for large wood, fine branches, leaves, and fruit are shown in Figures 4 through 7. The biomass partitioning measurements were from trees destructively sampled at the end of their agricultural production cycle, between 20 and 25 years of age, and before orchards were renewed. We found no biomass accumulation data from studies where younger trees were destructively sampled. Table 8 shows the statistics from the modeling effort. It is our opinion that the model parameterization effort would be improved with either destructive sampling or allometric-derived biomass measurements from younger orchards, ideally distributed in time throughout the age of a typical almond orchard (up to about 25 years). The dataset used for this analysis contained a sample size of just six measures for large wood biomass, six for small branches, and four for leaves.

We verified modeled plant production against almond kernel yield data derived from the California Agricultural Statistics Reviews from 2000 to 2015 (CDFA, 2017a). We modeled plant production and crop yield using DayCent at > 1,000 agricultural points in California. Each point was predicted by the Crop Data Layer (NASS 2017b) to have an almond orchard for at least 7 years. Soils data for each point were derived from the NRCS SSURGO soils database (NRCS 2017), and weather were derived from the PRISM daily weather model (PRISM 2017). UC-Davis Extension enterprise economic models (various sources) were used to set up the regional agricultural practices used in the simulations. . For the sixteen years modeled in this process (2000-2015), the mean DayCent-modeled yield was 397 g C m^{-2}, and the mean measured almond kernel yield as reported by CDFA was 387 g C m^{-2}. The RMSE of the comparison was 74.97, and the correlation coefficient was 10%.

7.4 Lettuce

The input parameters for lettuce production were developed from studies conducted at two locations: 'Hartnell, CA' (X. Zhu-Barker, unpublished data) and 'Cal-Core' (S. Kortman unpublished data). Both studies were conducted in coastal ecosystems. No published studies were located for winter lettuce production in the southern desert. Model parameterization results are shown in Figure 9.

Based on differences in plant production methods and yield, we developed three lettuce crops for use in COMET-Farm: romaine lettuce, leaf lettuce, and head lettuce. Crop yield data modeled by DayCent at points predicted by the NASS Cropland Data Layer to grow these crops in the California Central Coast, San Joaquin Valley, and Southern Desert were used in the model parameterization effort. The mean results of the verification modeling efforts are shown in Table 9.

During the model verification process for these three crops presented here, as well as other crops from previous work, we frequently find relatively ambiguous correlations between the measured yield and the DayCent modeled yield. There are a tremendous number of potential sources of variation in the process, few of which we can control. The yield measurements are averages at the county level for a particular year. Farmers occasionally have structural or process disincentives to report actual crop yields, which can lead to differences in the mean yield. These include crop losses (not reported as economic yield), crops rejected by processors due to quality or timing standards, or regulatory caps on the amount of product that may be sold. On the modeling side, we find it impossible to capture the broad suite of management practices that producers employ in crop production, including fertilizer rate and timing, irrigation amount and timing, timing of planting, timing of harvest, and other issues. Variations in model input datasets (soils, weather) can also lead to variation in model results. Because of these, we use the model verification process to identify regional variations in crop yield and overall plant production. We leave biomass partitioning alone and adjust only overall plant production up or down so that the DayCent-modeled crop yield fits the reported crop yield from the region the point falls within. The complexity of practices and diverse climates for these specialty crop systems, such as in strawberries (e.g. coastal climate, plastic mulching), lettuce (e.g. inland and coastal climates, fertilization regimes), and wine grapes, also presented challenges to DayCent performance, underscoring the need for comprehensive datasets on how these practices affect both crop biomass production, C and N partitioning within farming system, and greenhouse gas emissions.

8. Literature summary and gaps in GHG emission knowledge from California specialty crops

Although a number of studies have examined N_2O emissions from certain specialty crops, more robust estimates of GHG emissions from California specialty crops still require additional research, particularly in side-by-side comparisons of annual N_2O emissions across different soil types and practices. For example, in tomato cropping systems, Kallenbach et al. (2010) compared the effect of irrigation type and cover crop on N_2O emissions. In this study, however, only hourly fluxes were reported and 13 gas samples were collected over one year. As a result of standard interpolation methods used for these data, annual N_2O emissions and EFs calculated based on these hourly fluxes are much higher than in other studies (Table 1). Other issues related to the estimation of GHG emissions based on published studies include the overestimation of EFs under cover crop practices due to lack of information on the contribution of cover crop N to N_2O emissions and the effect of fertilizer placement and timing. Difficulties are also faced in estimating the GHG mitigation potential for alternative practices, including subsurface drip irrigation, conservation tillage, organic amendments like compost and manures, and cover crops due to lack of research on these practices and their impacts on N_2O emissions across California landscapes. Limited geographic extent of these measurements also presents challenges to drawing reliable conclusions about the consistency of observed phenomena.

To estimate N_2O fluxes that reflect an integration of multiple management practices, research is also needed to improve empirical quantification of soil N_2O emissions and therefore model (i.e. DayCent) development. For example, more data are needed to better quantify soil N_2O production from different sources as affected by climate change, management practices, and soil biophysics; greater understanding is needed of the mechanism of the effects of subsurface drip irrigation on soil N_2O emissions; development is needed of a set of geographically stratified test sites at which factors known to affect agronomic N_2O emissions are varied in order to provide a robust empirical data set for establishing Tier 2 and Tier 3 methods of IPCC. The development and improvement of the DayCent model for California specialty crops also requires information on how management practices influence nutrient cycling and water movement. For example, in woody perennials, better data on how floor management affects nutrient allocation is needed to more effectively model these crops. Other practices, such as plastic mulch used in strawberry cropping systems, subsurface drip irrigation in tomato systems, microsprinkler/drip irrigation in almond orchards, and the application of different synthetic fertilizers (i.e. alkaline-forming vs. acidifying N fertilizers) and organic amendments (i.e. fresh organics vs. compost), also warrant further characterization for use in the DayCent model.

Agricultural practices are being increasingly examined and employed for dual benefits for mitigation and adaptation. Understanding how various practices for reductions in greenhouse gas emissions will continue to improve as knowledge gaps are filled regarding the mechanistic understanding of soil N_2O production processes for the numerous specialty crops in California. A key need is the generation and curation of high quality data from these specialty crop systems that will be used to evaluate and refine predictive emissions models. Other benefits derived from addressing this need include identification of practices that enable more efficient on-farm N fertilizer use, engagement of land users through education and outreach, and delivery of new data to inform policy makers' efforts to design farseeing strategies that meet both agricultural and environmental goals.

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Appendix 1. Brief overview of biological pathways underlying nitrogen transformations and emissions

1.1 Ammonia oxidation pathways

Ammonia oxidation is the first step of traditional nitrification, by which ammonia is converted with ammonia monooxygenase to nitrite via hydroxylamine (Firestone and Davidson, 1989a). This biotic process, carried out under aerobic conditions by nitrifying microorganisms that use ammonia as an energy source, can lead to N_2O production under different conditions through several pathways. These pathways include nitrifier nitrification, nitrifier denitrification, and nitrification-coupled denitrification (Hooper and Terry, 1979; Wrage *et al.*, 2001; Zhu *et al.*, 2013a).

Factors that influence N_2O production through ammonia oxidation pathways include soil nitrifier activity, ammonium concentration and sorption, texture, pH, temperature, moisture, oxygen availability, and bioavailability of metals (e.g. iron and manganese). Examples of how these factors influence N_2O production are presented here. Nitrous oxide production has been reported to be greater from clay soils than from loams or sandy soils (Zhu *et al.*, 2013a). This is likely a result of clay surfaces protecting nitrifier populations from the effects of acidifying hydrogen ions (H^+) produced during NH_3 oxidation (Powell and Prosser, 1991). Increased clay content can lead to higher nitrifier populations and associated higher nitrification rates (Fortuna *et al.*, 2012), as well as more anaerobic microsites that enable denitrification. Short-term increases in soil pH and associated higher free ammonia limit nitrite oxidation by *Nitrobacter*, resulting in the accumulation of nitrite (Hawkins *et al.*, 2010; Venterea *et al.*, 2015; Breuillin-Sessoms *et al.*, 2017), which can also lead to N_2O production (Venterea and Rolston, 2000; Venterea, 2007). Generally, increased soil temperatures are associated with increases in ammonia oxidizer populations (Avrhami and Conrad, 2003; Avrahami and Bohannan, 2007; Szukics *et al.*, 2010) and activities (Avrahami *et al.*, 2003), resulting in increases in N_2O production through ammonia oxidation pathways. However, one study showed that when soil temperature increased from 25 to 37^{o}C, potential nitrification rate decreased but the rate of N_2O release increased monotonically (Avrahami *et al.*, 2003). This indicates that N_2O production through ammonia oxidation pathways can be independent from nitrification under certain conditions. It has been reported that oxygen availability in soil is also an important factor that influences N_2O production through ammonia oxidation pathways (Zhu *et al.*, 2013a). When oxygen concentrations decreased from 21% to 0.5%, N_2O production via ammonia oxidation pathways increased. Soil moisture influences N_2O production by controlling not only the diffusion of oxygen, but also substrate availability (Stark and Firestone, 1995) and microbial activity (Avrahami and Bohannan, 2007). The biological availability of iron is a key factor that regulates iron-dependent enzymes participating in N_2O production through ammonia oxidation pathways, such as ammonia monooxygenase, hydroxylamine oxidoreductase, and nitrite oxidoreductase (Meiklejohn, 1952; Godfrey and Glass, 2011; Stein, 2011; Glass and Orphan, 2012).

1.2 Heterotrophic denitrification

Heterotrophic denitrification is a stepwise reduction of nitrate or nitrite with dinitrogen (N_2) as the end product. This process is performed by denitrifiers, which are widely distributed across bacterial taxa and include *Pseudomonas, Bacillus, Thiobacillus, Propionibacterium* and others (Firestone *et al.*, 1980c; Firestone, 1982; Knowles, 1982). Some intermediates, such as NO and N_2O, can be released into the atmosphere under certain environmental conditions (Firestone *et al.*, 1979; Firestone *et al.*, 1980a; Firestone and Davidson, 1989a; Weier *et al.*, 1993; Mulvaney *et al.*, 1997; Van Cleemput, 1998a; Gillam *et al.*, 2008; Phillips, 2008). Enzymes catalyzing these reductions are nitrate reductase, nitrite reductase, nitric oxide reductase and N_2O reductase (Hochstein and Tomlinson, 1988).

Soil water content is one of the most important factors influencing the denitrification process, as it interacts with pore size distribution to control oxygen diffusion, soil NO_3^- concentration and C availability (Smid and Beauchamp, 1976; Firestone *et al.*, 1979; Firestone and Davidson, 1989a; Weier *et al.*, 1993). Denitrification-derived N_2O has been shown to increase as soil water content increases, especially when soil water content is higher than 70% of water-filled pore space (WFPS) (Bateman and Baggs, 2005). The effects of oxygen concentration on denitrification are mainly due to the effects on the activities of enzymes associated with denitrification. For example, nitrate reductase activity can be completely blocked at an oxygen concentration greater than 0.25%, compared with 0.13% and 0.02% for nitrite and N_2O reductases, respectively (Bonin *et al.*, 1989). Increases in nitrate concentration do not always increase the overall rate of denitrification, but can result in a higher proportion of N_2O as the product of denitrification (Cooper and Smith, 1963; Firestone *et al.*, 1979; Knowles, 1982; Weier *et al.*, 1993). This is likely because nitrate is a preferred electron acceptor over N_2O (Thauer *et al.*, 1977; Firestone *et al.*, 1980b; Firestone and Davidson, 1989b). Content of soil organic carbon (SOC), as energy source and electron donor for denitrifiers, is highly correlated with denitrification capacity (Myrold and Tiedje, 1985; Weier *et al.*, 1993; Gillam *et al.*, 2008). Both N_2O and N_2/N_2O ratio increase as SOC, especially decomposable carbon (C), increases (Burford and Bremner, 1975).

Temperature is also an important factor controlling emissions of N_2O from denitrification. For example, the rate of $N_2O + N_2$ evolved from denitrification has been shown to increase with an increase in soil temperature to 67^{o}C. Nitrous oxide also becomes an increasingly smaller component of the gaseous nitrogen (N) evolved and persists for shorter times with increasing temperature (Nömmik, 1956; Smid and Beauchamp, 1976; Keeney *et al.*, 1979).

1.3 Abiotic chemodenitrification

Chemodenitrification is the chemical decomposition of N intermediates in the ammonia oxidation (e.g. nitrite) or denitrification (e.g. nitrate, nitrite, or NO or N_2O) pathways, via coupled reactions involving oxidation of Fe(II) or other electron donors such as humic acids (Chao and Kroontje, 1966; Buresh and Moraghan, 1976; Chalk, 1983; Sørensen and Thorling, 1991; Van Cleemput, 1998a; Thorn and Mikita, 2000; Picardal, 2012). Soil N_2O emissions from chemodenitrification are stimulated by decreased oxygen (i.e. increased denitrification), increased Fe(II) and the presence of organic matter (Nelson and Bremner, 1970; Van Cleemput and Baert, 1984). Therefore, N_2O production from chemodenitrification is assumed to mostly exist in anoxic conditions with a high supply of organic substrates, such as in wetlands and waterlogged soils. In these environments, persistence of significant concentrations of Fe(II) and bacterial denitrification can lead to the accumulation of reactive NO_2^- and NO (Van Cleemput, 1998b), and subsequent N_2O production (Venterea and Rolston, 2000; Venterea, 2007).

Appendix 2. Overview of agricultural practices that influence soil organic carbon content

The intent of Appendix 2 is to provide updated information on the role of agricultural practices in soil organic carbon (SOC) dynamics. It also serves as a point of comparison for the specialty crop systems examined in the main body of this report.

2.1 Fertilization

Soil fertility affects the production of crop biomass, as well as the quality and quantity of crop residues that are returned to the soil (Rasmussen and Rohde, 1988; Lal *et al.*, 1998; Follett, 2001; Christopher and Lal, 2007; Halvorson *et al.*, 2009; Halvorson and Jantalia, 2011; Jantalia and Halvorson, 2011; Halvorson and Stewart, 2015). Halvorson et al. (1999) observed that in a barley-corn-winter wheat-oat-pea-hay rotation system, the amount of crop residue (aboveground residue plus belowground residue) returned to the soil increased with increasing nitrogen (N) rate and resulted in increased SOC in the 0 to 15 cm soil depth. For example, 140 and 182 kg ha^{-1} more SOC accumulated in plots receiving fertilizer N rates at 67 and 134 kg N ha^{-1} yr^{-1} compared with the unfertilized treatment, respectively. In comparison, Kong et al. (2005) reported that between 1993 and 2003 in California, SOC was sequestered at the rate of 40 kg carbon (C) ha^{-1}yr^{-1} in a fertilized wheat-fallow system while SOC decreased at the rate of 190 kg C ha^{-1}yr^{-1} in a unfertilized wheat-fallow system. In a review of SOC in U.S. cropping systems, Lal et al. (1998) concluded that 50-150 kg C ha^{-1}yr^{-1} SOC can be sequestered by fertility management practices.

Increasing SOC sequestration under N fertilization relies on crop residue C inputs, N fertilization rate, initial soil C content, and the soil's capacity for retention. Increases in SOC do not always correspond directly to increases in N fertilization if crop growth is optimized at a lower rate (McCarty et al. 1997). Initial background SOC can also be sufficiently high to mask increases in SOC, even in long-term studies that demonstrate consistent increases, in response to N fertilization, in the crop biomass returned to the soil (Halvorson et al. (2002). Some studies also report little or no increase in SOC sequestration with increasing N additions during crop growth, especially for inorganic N fertilizer (Campbell *et al.*, 1991; Huggins and Fuchs, 1997; Belanger *et* *al.*, 1999; Halvorson *et al.*, 2002). Many of crops grown where findings on the role of fertilization on crop growth and resulting increases in SOC are rainfed. In contrast, nearly all California's specialty crop systems are irrigated, suggesting that managing specialty crop growth to increase SOC sequestration could occur by careful optimization of fertilization with irrigation (see 4.3).

2.2 Tillage

Changes in tillage practices influence the total soil C stock and the vertical distribution of C in the soil profile (Paustian *et al.*, 1997). Long-term conventional tillage has resulted in the reduction of soil C stocks by up to 30% compared to pre-tillage levels, yet a change in tillage from conventional to long-term adoption of conservation, no-till or reduced tillage may reverse such losses of soil C and enhance C sequestration (Paustian *et al.*, 1997; Lal *et al.*, 1998; West and Marland, 2002; Lal *et al.*, 2004; Ogle and Paustian, 2005; Álvaro-Fuentes *et al.*, 2009), but the magnitude of soil C accumulation after the adoption of no-till also depends on soil texture, climate, and current soil C stocks (West and Six, 2007; Syswerda *et al.*, 2011; Collins *et al.*, 2012; Ogle *et al.*, 2012). For example, an estimated 233 kg C $ha^{-1}yr^{-1}$ was sequestered with no-till in an annual cropping rotation system in North Dakota, compared with 25 kg C $ha^{-1}yr^{-1}$ with conservation tillage and a loss of 141 kg C $ha^{-1}yr^{-1}$ with conventional tillage (Halvorson *et al.*, 2002). In some cases, crop production and corresponding C inputs decrease under no till, but any potential reduction in SOC stocks can be offset by a decrease in soil C decomposition rates, except in cases where C inputs are reduced by 15% or more (Ogle *et al.*, 2012). Still, efficacy of conservation or no till practices has been questioned in recent studies (Baker *et al.*, 2007; Christopher *et al.*, 2009). No-till practices can lead to accumulation of soil C in the upper soil profile, but with little to no change in the lower soil profile (Syswerda *et al.*, 2011). Furthermore, a shift from no-till to conventional tillage can cause a decrease in soil C in the upper soil profile, negating any increase in SOC in the upper soil profile under no-till (Ogle and Paustian, 2005; Christopher *et al.*, 2009). Yet, Alcántara et al. (2016) reported that tillage with deep plowing can increase SOC sequestration by translocating the SOC formed near the surface to the subsoil and enlarging the storage space for SOC-rich material.

As stated above, the response of SOC to tillage and associated benefits depends on the duration of tillage adoption. In a California tomato-cotton rotation system, for example, Veenstra et al. (2007) reported that compared to the initial SOC, the content of SOC in the top 30 cm of soil had decreased five years after the adoption of conservation tillage, whereas a net increase was found ten years after adoption (Mitchell *et al.*, 2015b). This is largely due to the dry climate in California, which delays decomposition of crop residues and consequently delays accumulation of SOC. No difference in SOC content (0-30 cm depth) was found between the conventional and conservation tillage ten years after the tillage adoption in this tomato-cotton rotation system (Veenstra *et al.*, 2007; Mitchell *et al.*, 2015b). However, the conversion of conventional tillage to no-till has led to a SOC sequestration rate of 0.3 Mg C $ha^{-1}yr^{-1}$ in annual crop systems under other climates such as temperate climate (West and Marland, 2002; Lal, 2003). The adoption of conservation tillage also influences yields and benefits soil and water conservation efforts (Unger, 1999; Baker *et al.*, 2005; Mitchell *et al.*, 2009; Mitchell and Miyao, 2012; Mitchell *et al.*, 2015b). For example, in a California cotton-tomato rotation system, Baker et al. (2005) reported that compared to conventional tillage, the adoption of conservation tillage reduced cumulative dust emissions by 67% throughout the two-year rotation. In the same rotation system, Mitchell et al. (2015b) found that tomato yields increased under conservation tillage compared to conventional tillage, whereas the opposite was found for cotton yields.

2.3 Irrigation

Irrigation is fundamental to maintaining or increasing crop yields in semiarid and arid lands where plant growth is limited by available water. Under irrigated systems, wetting of dry soils typically leads to short-term increases in CO_2 emissions as the labile soil C pool is mineralized (Kieft *et al.*, 1987; Fierer and Schimel, 2002; Ruser *et al.*, 2006). However, SOC in these systems can be maintained or increased due to C inputs as a result of increased plant growth and therefore increased inputs from crop residues (both above- and below-ground) and increased aggregate formation (Follett, 2001; De Gryze *et al.*, 2005; Kong *et al.*, 2005; Gillabel *et al.*, 2007; Collins *et al.*, 2012). Lal et al. (1998) estimated that rates of SOC sequestration in irrigated cropping system in the U.S. fall between 50 and 150 kg $ha^{-1}yr^{-1}$. However, Follett (2001) viewed these estimates as too conservative. The potential for C sequestration in intensively managed irrigated cropping systems can be increased considerably by the use of improved fertilization practices, tillage, and irrigation management. After considering the application of improved technology, Eve et al. (2002) documented that irrigation results in SOC sequestration ranging from 0.25 to 0.52 Mg C $ha^{-1}yr^{-1}$ in the western U.S. In a model projection study conducted in a semiarid Mediterranean agroecosystem, however, the shift from rain-fed conditions to irrigation resulted in an increase in C inputs but a decrease in the SOC sequestered during the 2010-2100 period due to the interactions of climate with system management that ultimately promoted greater C mineralization (Álvaro-Fuentes and Paustian, 2011). In California, almost all agricultural land is irrigated due to the prevailing Mediterranean climate. However, information on how different irrigation management practices affect soil C stock, especially the distribution of SOC in the soil profile, is still lacking.

2.4 Cover crops and organic amendments

The potential for cover crops and organic amendments to enhance SOC content, increase crop productivity, sequester C, and reduce atmospheric CO_2 concentrations makes them appealing for both climate change mitigation and land-use sustainability. Duval et al. (2016) observed that the increase in the mean annual C input by cover crops and their plant residue into the soil explained most of the variation in SOC in a soybean cropping system in Argentina. In a California tomato-cotton rotation cropping system, planting a cover crop over a 5-year period increased SOC sequestration rates by 0.9 and 0.77 Mg C ha^{-1} yr^{-1} compared to the absence of cover crops under conservation and conventional tillage practices, respectively (Veenstra *et al.*, 2007). After ten years of implementation n the tomato-cotton rotation, cover cropping resulted in sequestration of 0.63 and 0.46 Mg C ha^{-1} yr^{-1} more SOC across the soil depth of 0-30 cm than non-cover cropping under conservation and conventional tillage practices, respectively (Mitchell *et al.*, 2015b). Across U.S. cropland, adoption of cover crop practices has been projected to potentially convert an estimated 0.15-0.22 Mg C ha^{-1}yr-^{1} atmospheric CO_2 to SOC (Swan *et al.*, 2015; Chambers *et al.*, 2016). This SOC sequestration rate is comparable to the C sequestration potential (0.1- 0.3 kg C ha^{-1} yr^{-1}) of winter cover crops in the U.S. estimated by Lal (1998).

The application of organic amendments supplies organic C that can directly compensate for soil C loss due to cultivation. Many studies on agricultural soils have reported that organic amendment additions result in higher C sequestration compared to synthetic N fertilizer additions (Fortuna *et al.*, 2003; Jarecki *et al.*, 2005; Kong *et al.*, 2005; Marriott and Wander, 2006; Zhang *et al.*, 2009; Qiao *et al.*, 2014; Li and Han, 2016). For example, a long-term experiment in Oregon investigating the effects of various N additions on SOC sequestration in a winter wheat system showed that over the 56-yr period examined, SOC concentrations in the soil depth of 0-30 cm declined with time in all the treatments except the manure treatments, in which SOC concentrations increased with time (Rasmussen and Parton, 1994). Tian et al. (2009) also reported that the application of biosolids as an organic amendment has the potential to turn Midwest Corn Belt soils in the U.S. from C neutral into C sinks. The high nutrient availability in organic amendments like manure and compost adds to their value in crop production systems, and the corresponding increases in crop residue C inputs (Zhang *et al.*, 2009; Qiao *et al.*, 2014).

Under California's Mediterranean climate, a 10-year long-term experiment in maize-tomato rotation cropping systems examined the role of C input in SOC sequestration. Mean annual C input as composted manure and crop residue in an organic system was 89.6 Mg C ha^{-1} yr^{-1} whereas the C inputs by just crop residues in the system receiving inorganic fertilizer was 51.8 Mg C ha^{-1} yr^{-1}; this resulted in a higher SOC sequestration rate in the organic system than in the inorganic system (0.56 vs. 0.04 Mg C ha^{-1} yr^{-1}) (Kong *et al.*, 2005). Application of compost (add amount) over six years also led to increases in SOC content in organic vegetable systems in the Central Coast of California (Brennan and Acosta-Martinez, 2017). In Spain, another Mediterranean climate, Calleja-Cervantes et al. (2015) observed that the application of organic amendments to a vineyard over 13 years increased SOC at least 35% more compared to the control (no organic amendments added). GHG mitigation from increased SOC sequestration resulting from organic amendments may be negated by GHGs emitted by fuel consumption caused by transportation to the farm and application of organic amendments. GHG emissions from California agriculture, as with many other regions, have varied responses to organic amendments depending on organic amendment management, soil type and environmental conditions (Zhu *et al.*, 2013b; Zhu-Barker *et al.*, 2015b). A comprehensive assessment of GHG emissions and SOC in specialty cropping systems receiving organic amendments across multiple regions and farming systems will support development of strategies to improve GHG mitigation and SOC sequestration. Additional studies on the underlying mechanisms of soil C stabilization also will maximize the capacity for soil C retention under California's Mediterranean climate (Kong *et al.*, 2005; Brennan and Acosta-Martinez, 2017).

2.5 Cropping system conversion

Cropping system conversion practices, e.g. intensification, crop rotation diversification, and conversion of annual to perennial crops, have a significant impact on SOC sequestration (De Gryze *et al.*, 2004; Eagle *et al.*, 2011). The practices improve soil C storage through increases in crop biomass returning to the soils (Kroodsma and Field, 2006; Eagle *et al.*, 2011). Crop rotation diversification often involves changing from a continuously cropped cereal or simple rotation to multiple crops over multiple years of a crop rotation. This change is expected to increase residue biomass, residue quality (e.g. N content), and root exudates, and therefore soil C storage. Eagle et al. (2011) reviewed studies on soil C sequestration for diversifying crop rotations and summarized that in U.S. cropland, the diversification of annual crop rotations has the potential to increase soil C at a rate of 1.6 Mg C ha^{-1} yr^{-1}. In a 20-year study of crop rotations in Nebraska, no increase in SOC content from two-year rotations (corn-soybean and sorghum-soybean) was found over continuous mono-cropping. However, four-year rotations with oats and clover significantly increased SOC content at rates of 0.34, 0.29, and 0.24 Mg C ha^{-1} yr^{-1} after 10, 16, and 20 years of diverse crop rotations, respectively (Varvel, 2006). Increasing the number of crops per year can produce higher quantity and quality residues that increase SOC sequestration (Halvorson *et al.*, 2002; Ogle *et al.*, 2005). For example, reducing fallow periods by adopting winter cover crops can increase C input and therefore increase SOC storage (Ogle *et al.*, 2005; Liebig *et al.*, 2010). An additional month of cropping each year also increased SOC at a rate of 0.07 Mg C $ha^{-1}yr^{-1}$ in a 10-year cropping study in Texas, U.S. (Franzluebbers *et al.*, 1998). The adoption of cover crops can utilize the additional nutrients left over from the preceding crop and reduce nutrient losses. As discussed in section 4.4., cover crops also increase SOC sequestration through biomass C inputs.

In U.S. cropping systems, the conversion of an annual to a perennial crop has been suggested to sequester C in soil, with variable results depending on the crop type (Liebig *et al.*, 2005; Grandy and Robertson, 2007). SOC sequestration was estimated to increase by 0.6 Mg C ha^{-1} yr^{-1} after the conversion of an annual to a perennial crop, while changing from cropland to pasture land yielded a projected average net impact of 1.2 Mg C ha^{-1} yr^{-1} (Culman *et al.*, 2014). Using the CASA (Carnegie-Ames-Stanford Approach) model to estimate soil C sequestration potential in California agriculture, Kroodsma and Field (2006) suggested that conversion from annual crops to vineyards can sequester 0.68 Mg C ha^{-1} yr^{-1}, and switching from annual crops to orchards can sequester 0.85 Mg C ha^{-1} yr^{-1}.

Appendix 3. Calculations for Emission Factors

$$EF(SDI) = EF(furrow) * EF'(SDI)/EF'(furrow) \qquad \text{Equation 1}$$

Where EF´(SDI) and EF´(furrow) are acquired from Kallenbach et al (2010), and EF(furrow) is averaged from Kennedy et al. (2013) and Burger and Horwath (2012).

$$EF\ (cover\ crop) = EF\ (no\ cover\ crop) * EF'\ (cover\ crop)/EF'\ (no\ cover\ crop) \qquad \text{Equation 2}$$

Where EF´ (cover crop) and EF´ (no cover crop) are calculated based on the N_2O fluxes reported by Kallenbach et al (2010), and EF (no cover crop) is averaged from Kennedy et al. (2013) and Burger and Horwath (2012) (for furrow irrigation) or calculated from equation 1(for SDI).

Figures:

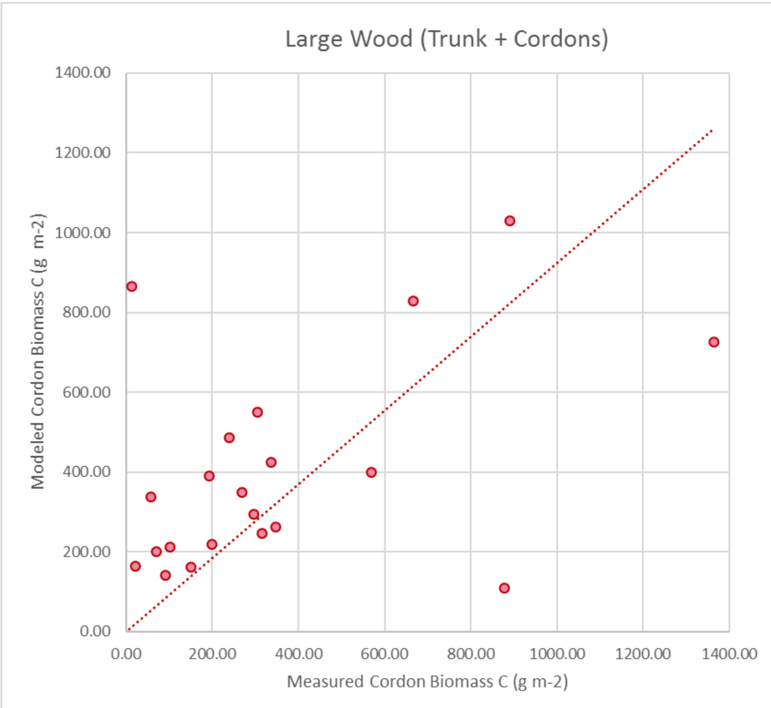

Figure 1. Measured vs modeled large wood biomass (diameter > 1cm), which corresponds to the vine trunk and woody cordons. R^2 = 0.42, slope = 0.93, correlation coefficient = 0.62.

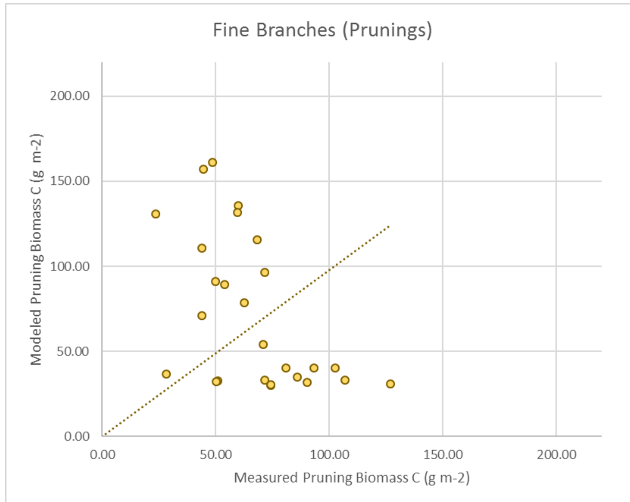

Figure 2. Measured vs modeled carbon in fine branches biomass (diameter <1 cm), which corresponds to woody vines pruned off the cordons during the dormant season. R^2 = 0.02, slope = 0.93, correlation coefficient = -0.14.

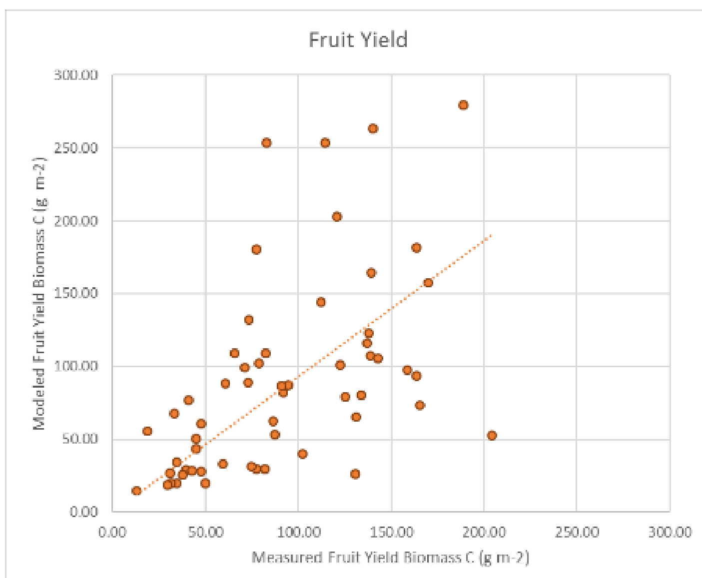

Figure 3. Measured vs modeled fruit carbon biomass in wine grapes. Fruit water content was estimated to be 85%. R^2 = 0.29, slope = 0.85, correlation coefficient = 0.54.

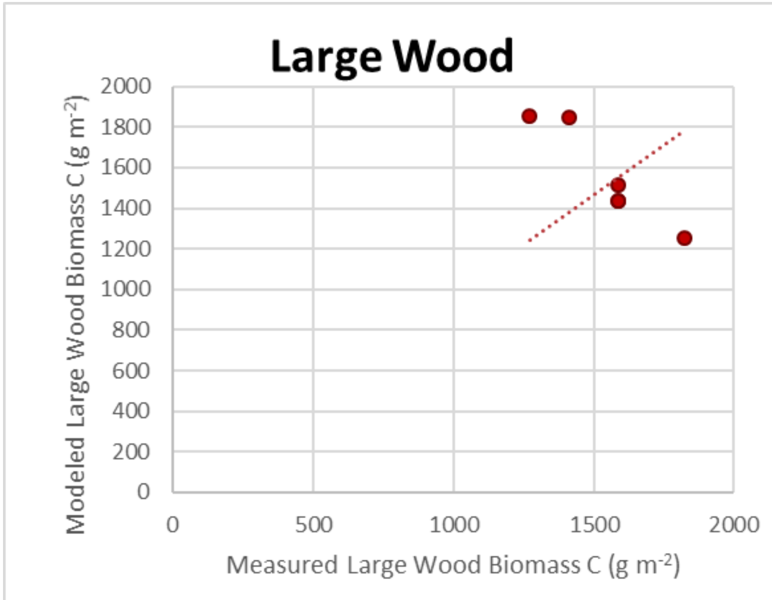

Figure 4. Measured vs modeled large wood biomass (diameter > 1cm) for almonds, which corresponds to the woody pool in trunks and branches approximately older than 1 year.

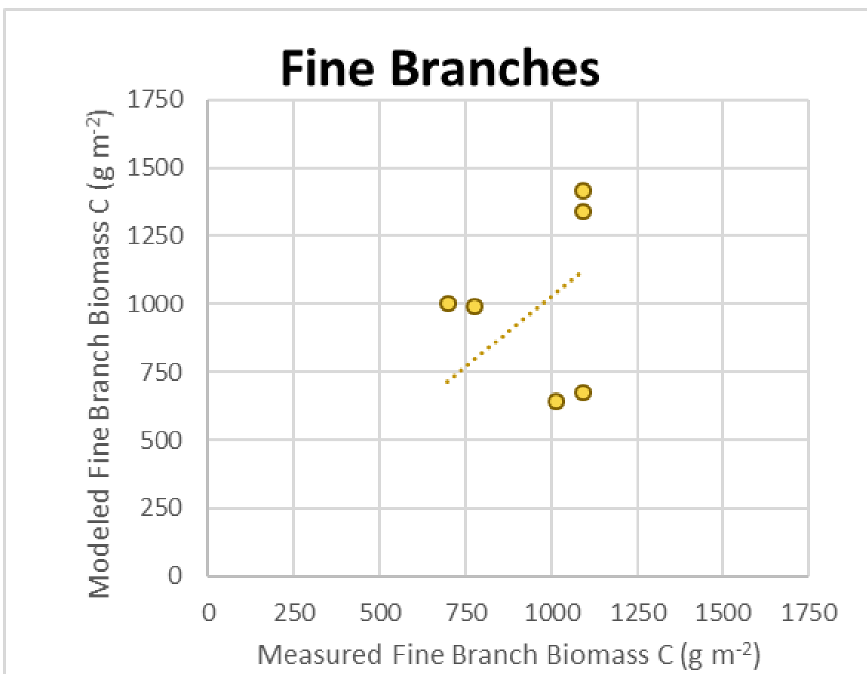

Figure 5. Measured vs modeled fine branches biomass (diameter < 1cm) for almonds, which corresponds to the woody biomass in branches that are approximately less than 1 year old.

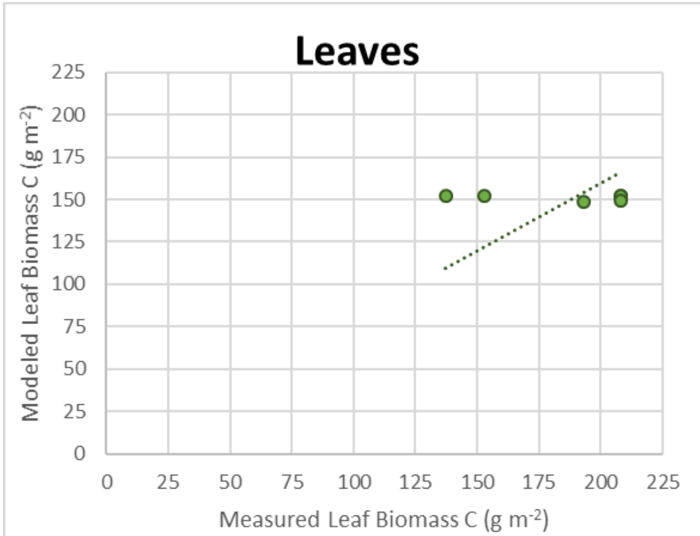

Figure 6. Measured vs modeled leaf pool for the almond studies.

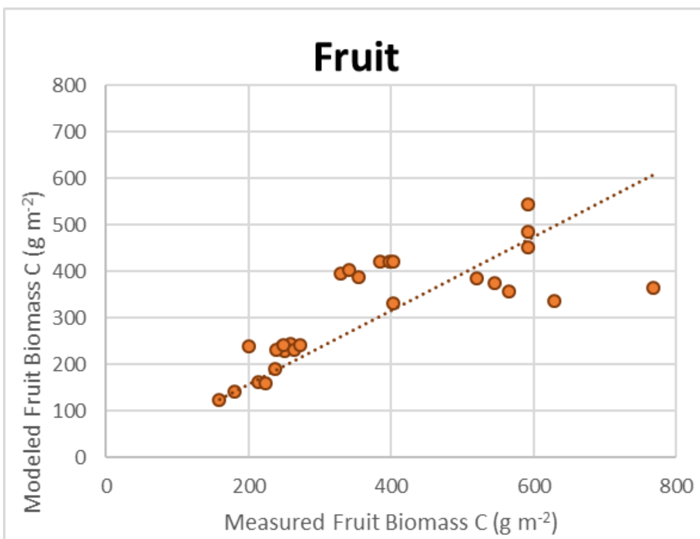

Figure 7. Measured vs modeled fruit pool for the almond studies.

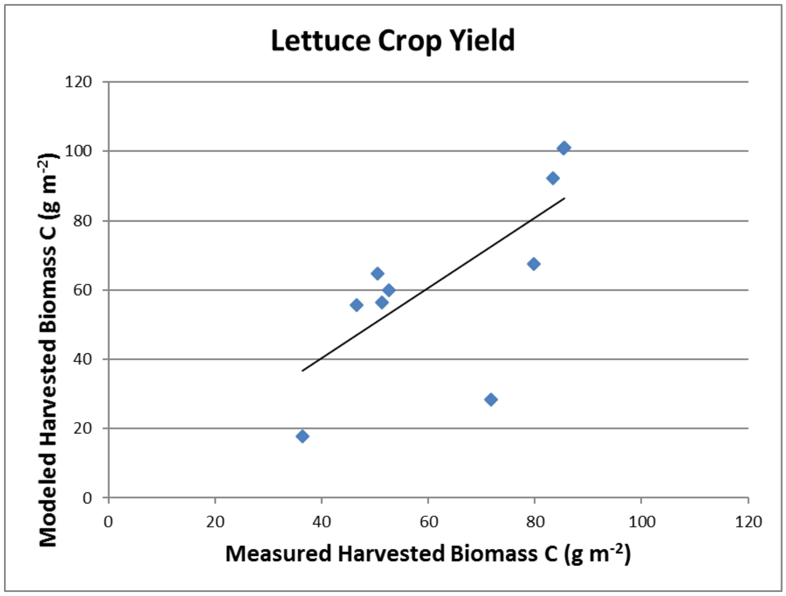

Figure 8. Results of the DayCent model input parameterizations for lettuce. Slope of the regression was 1.0, RMSE was 18.4, and the correlation coefficient was 0.73.

Tables

Table 1. Nitrous Oxide Emission Studies in California Tomato Cropland

Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha^{-1})	Crop residual N (kg N ha-1)	N_2O Emissions			Emission factors (% of applied N emitted as N_2O, uncorrected for background flux)		Emission factors (% of applied N emitted as N_2O, corrected for background flux)*		Study scale	References
							Growing season (kg N ha^{-1})	Annual (kg N ha^{-1} yr^{-1})	Scaling factor (annual/season)	Seasonal	Annual	Seasonal	Annual		
Clay loam	Furrow	Standard	N	8-24-6; 28-0-5; CAN-17	402[a]	73	2.01 ± 0.19	3.06 ± 0.19	1.52	0.85± 0.08	0.76 ± 0.05	0.65± 0.06	0.64 ± 0.04	Field	Kennedy et al., 2013
	SDI	Reduced	N	8-24-6; UAN-32	205	111	0.58 ± 0.06	0.95 ± 0.05	1.64	0.28 ± 0.03	0.46 ± 0.02	0.28 ± 0.03	0.31 ± 0.02	Field	Kennedy et al., 2013
Yolo silt loam: 22% sand, 47% silt, 31% clay	Furrow	Standard	N	15-15-15; Ammonium sulfate; UAN-32	0			1.00 ± 0.16[b]		na	na	na	na	Field	Burger and Horwath, 2012
					75		na	1.23 ± 0.01[b]		na	1.63 ± 0.01	na	0.31 ± 0.01	Field	Burger and Horwath, 2012
					162		na	1.81 ± 0.18[b]		na	1.11± 0.11	na	0.50± 0.07	Field	Burger and Horwath, 2012
					225		na	4.06 ± 0.49[b]		na	1.80 ± 0.22	na	1.36 ± 0.20	Field	Burger and Horwath, 2012
					300		na	4.34 ± 0.87[b]		na	1.45 ± 0.29	na	1.12± 0.24	Field	Burger and Horwath, 2012
Reiff loam and Yolo silt loam	Furrow	Standard	N	Ammoniun nitrate	120	74	na	4.04[d]		na	3.37[d]	na	2.08[d]	Field	Kallenbach et al., 2010
	Furrow	Standard	Y	Ammoniun nitrate	227[c]	74	na	7.99[d]		na	3.52[d]	na	2.65[d]	Field	Kallenbach et al., 2010
	SDI	Standard	N	Ammoniun nitrate	120	74	na	2.09[d]		na	1.74[d]	na	1.07[d]	Field	Kallenbach et al., 2010
	SDI	Standard	Y	Ammoniun nitrate	227[c]	74	na	5.68[d]		na	2.50[d]	na	1.89[d]	Field	Kallenbach et al., 2010

*Emission factors were corrected based on considering either crop residue N input or N_2O emissions from zero synthetic N input treatment

[a] 237 kg N ha^{-1} was applied during tomato growing season and 165 kg N ha^{-1} was applied during winter wheat growing season.

[b] Tomato growing season.

[c] 120 kg N ha^{-1} synthetic N plus 107 kg N ha^{-1} biomass N from cover crop residue.

[d] Data based on only 13 hourly flux measurements.

na: No data available; SDI: subsurface drip irrigation.

Table 2. Nitrous Oxide Emission Studies in California Lettuce Cropland

Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha^{-1})	Crop residual N (kg N ha^{-}	N_2O Emissions			Emission factors (% of applied N emitted as N_2O, uncorrected for		Emission factors (% of applied N emitted as N_2O, corrected for		Study scale	References
							Growing season (kg N ha^{-1})	Annual (kg N ha^{-1} yr^{-1})	Scaling factor (annual/season)	Seasonal	Annual	Seasonal	Annual		
Yolo silt loam: 46% sand, 32 silt, 22% clay	SDI [a]	Standard [a]	N	Commercial organic fertilizer	190	0.02	0.55±0.09	0.92 [c]	1.67 [b]	0.30±0.05	0.48	0.30±0.05	0.48	Greenhouse	Pereira, 2014
Yolo silt clay loam: 19% sand; 48% silt; 34% clay			N	Commercial organic fertilizer	0		0.16±0.12	0.27 [c]	1.67	na	na	na	na	Greenhouse	Pereira, 2014
	SDI [a]	Standard [a]			56		0.35±0.05	0.58 [c]	1.67	0.63±0.08	1.04	0.34±0.12	0.55	Greenhouse	Pereira, 2014
	SDI [a]	Standard [a]			112		0.79±0.06	1.32 [c]	1.67	0.71±0.05	1.18	0.56±0.07	0.94	Greenhouse	Pereira, 2014
	SDI [a]	Standard [a]			168		1.00±0.21	1.67 [c]	1.67	0.60±0.13	0.99	0.50±0.15	0.83	Greenhouse	Pereira, 2014
	SDI [a]	Standard [a]			225		1.15±0.13	1.92 [c]	1.67	0.51±0.06	0.85	0.44±0.08	0.73	Greenhouse	Pereira, 2014
Silt loam: 49% sand, 29 silt, 22% clay	SDI	Standard	Y	organic fish pellet fertilizer	260	na		1.09			0.42		0.32 [d]	Field	Suddick and Six, 2013
Loam: 54% sand; 29% silt; 17% clay	SDI	Standard	N	UAN-32	84	15	0.34±0.02	0.64±0.05	1.89	0.4	0.76±0.06	0.34	0.65±0.03	Field	Burger and Horwath, 2012
					168	19.5	0.50±0.04	0.91±0.14	1.82	0.3	0.54±0.08	0.27	0.49±0.04	Field	Burger and Horwath, 2012
					252	21.5	0.74±0.04	1.12±0.11	1.52	0.29	0.44±0.04	0.27	0.41±0.04	Field	Burger and Horwath, 2012
					336	21.5	1.00±0.08	1.47±0.25	1.47	0.3	0.44±0.07	0.28	0.41±0.06	Field	Burger and Horwath, 2012

[a]Simulated practice

[b]Value averaged from the scaling factors obtained from Burger and Horwath 2012

[c] Calculated based on scaling factor

[d] Background corrected using N_2O emission background reported in Pereira, 2014

Table 3. Nitrous Oxide Emission Studies in California Vineyard

Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha^{-1})	N_2O Emissions: Growing season (kg N ha^{-1})	N_2O Emissions: Annual (kg N ha^{-1} yr^{-1})	N_2O Emissions: Scaling factor (annual/season)	Emission factors (% of applied N emitted as N_2O): Seasonal	Emission factors (% of applied N emitted as N_2O): Annual	Study scale	References
Loam: 48% sand, 33% silt, 19% clay	Drip	No till	Y	UAN-32	52[a]	0.20±0.02[b]	5.04[c]	28	0.38±0.04	9.69	Field	Garland et al., 2011
		Standard				0.14±0.04[b]	3.92[c]	28	0.27±0.08	7.54		
Loam: 48% sand, 33% silt, 19% clay	Drip	Standard	Y	UAN-32	52[a]	0.14	3.92	28	0.27	7.54	Field	Garland et al., 2014
			N		5.4	0.3	0.56	1.87	5.56	10.4		
Sandy clay loam: 50% sand, 27% silt, 23% clay	Drip	Standard	Y	UAN-32	129[d]	0.76	2	2.63	0.59	1.55	Field	Verhoeven and Six, 2014
					275[e]	0.37	1.6	4.32	0.13	0.58		
Loam	Drip	Standard	N	na	na	na	0.47			na	Field	Steenwerth et al., 2008
			Y			na	0.57			na		
			Y			na	0.69			na		
Loam: 33% sand, 42% silt, 25% clay	Drip	Reduced	Y	Potassium nitrate	43.4[f]	na	0.14±0.03			0.32±0.07	Field	Wolff, 2015
		Standard	Y		42[f]	na	0.17±0.01			0.40±0.02		
		Standard	N		27.6[f]	na	0.13±0.03			0.47±0.11		
Loam: 33% sand, 42% silt, 25% clay	Drip	Reduced	Y	Potassium nitrate	38.8[g]	na	0.15±0.04			0.39±0.10	Field	Wolff, 2015
		Standard	Y		44[g]	na	0.20±0.04			0.45±0.09		
		Standard	N		38.6[g]	na	0.19±0.04			0.49±0.10		
unknown	Drip	Standard	unknown	unknown	0	na	0.03[h]			na	Field	Smart et al., 2006
					6	na	0.05[h]			0.96		
					45	na	0.09[h]			0.2		

[a]5 kg N ha^{-1} synthetic N plus 47 kg N ha^{-1} organic N from cover crop residue

[b]Value calculated from berm (contributes 26%) and row emissions (contributes 74%

[c]Value calculated based on scaling factor

[d]9.6 kg N ha^{-1} synthetic N plus 119.6 kg N ha^{-1} organic N from cover crop residue

[e]66.4 kg N ha^{-1} synthetic N plus 208.8 kg N ha^{-1} organic N from cover crop residue

[f]8.4 kg N ha-1 synthetic N plus organic N from alley crop residue

[g]16.8 kg N ha-1 synthetic N plus organic N from alley crop residue

[h]Value calculated from the fluxes reported in the literature and were not included in the statewide emission calculation

Table 4. Nitrous Oxide Emission Studies in California Almond Orchard

Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha^{-1})	Returned N (kg N ha^{-1})*	N_2O Emissions			Emission factors (% of applied N emitted as N_2O, uncorrected for background flux)		Emission factors (% of applied N emitted as N_2O, corrected for background flux)†		Study scale	References
							Growing season (kg N ha^{-1})	Annual (kg N ha^{-1} yr^{-1})	Scaling factor (annual/season)	Seasonal	Annual	Seasonal	Annual		
Sandy loam: 64% sand, 17% silt, 19% clay	Microsprinkler	No till	N	UAN: 4 times application	224	105		0.80±0.19			0.35±0.08		0.24±0.06	Field	Schellenberg et al., 2012
	Microsprinkler	No till		CAN: 4 times application	224	102		0.53±0.11			0.23±0.05		0.16±0.03		
Sandy loam: 64% sand, 17% silt, 19% clay	Drip	No till	N	UAN: 4 times application	336	105[a]	0.781	1.17	1.5	0.23	0.35	0.18	0.27	Field	Wolff, 2015
	Drip	No till		UAN: 8 times application	336	105[a]	1.036	1.55	1.5	0.31	0.46	0.23	0.35		
	Drip	No till		CAN: 8 times application	336	102[a]	0.511	0.77	1.5	0.15	0.23	0.12	0.17		
Sandy loam	Drip	No till	N	unknown	236	80[b]		1.61±0.68			0.68		0.47	Field	Alsina et al., 2013
	Microsprinkler	No till		unknown	236	80[b]		0.6±0.25			0.25		0.18		
Sandy loam: 67% sand, 19% silt, 14% clay	Drip	No till	N	UAN32	225	80[b]		1.3±0.6			0.58		0.43	Field	Calrecycle, 2015
					252	80[b]		0.70±0.02			0.3		0.21		
Sandy loam: 60% sand, 27% silt, 13% clay	Microsprinkler	No till	N	UAN32	258	78		0.65±0.07			0.25±0.03		0.19	Field	Decock et al, 2017
					280	82		0.53±0.19			0.19±0.07		0.18		

[*]N returned in hull and shall

[†]Emission factors were corrected based on including crop residue N in N input

[a]Value was adopted from Schellenberg et al., 2012

[b]Value was adopted from Decock et al., 2017

Table 5. Nitrous Oxide Emission Studies in California Walnut Orchard

Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha^{-1})	N_2O Emissions			Emission factors (% of applied N emitted as N_2O)		Study scale	References
						Growing season (kg N ha^{-1})	Annual (kg N ha^{-1} yr^{-1})	Scaling factor (annual/season)	Seasonal	Annual		
Silty clay loam: 18.8% sand, 47.6% silt, 33.6% clay	Microsprinkler	No till	Y	Cover crop biomass-N	Tree row: 56 kg N ha^{-1}; Tractor row: 103 kg N ha^{-1}		Tree row: 1.15 kg N ha^{-1}; Tractor row: 1.29 kg N ha^{-1}			1.51[a]	Field	Pereira et al., 2016
	Microsprinkler	No till		Feather meal; Cover crop biomass-N	Tree row: 179 kg N ha^{-1}; Tractor row: 226 kg N ha^{-1}		Tree row: 1.18 kg N ha^{-1}; Tractor row: 2.41 kg N ha^{-1}			0.93[a]		

[a]Value calculated from the emission factors from tree row (contributes 33%) and tractor row (contributes 74%); value was not included in the statewide emission calculation

Table 6. California Specialty Crops Planting Acreage, Management Practices, N_2O Emission Factors, and Statewide N_2O emissions

Crop	Planted area (1,000 ha)	N input (kg N ha^{-1})	Irrigation type	Tillage	Cover crop	Uncorrected for background flux: Annual emission factors (% of applied N emitted as N_2O-N)	Uncorrected for background flux: Statewide annual N_2O-N emissions, Mg	Uncorrected for background flux: Statewide annual N_2O-N emissions, Gg CO_2 eq	Corrected for background flux: Annual emission factors (% of applied N emitted as N_2O-N)	Corrected for background flux: Statewide annual N_2O-N emissions, Mg	Corrected for background flux: Statewide annual N_2O-N emissions, Gg CO_2 eq	Number of observations	IPCC Tier 1 method derived statewide N_2O-N emissions, Gg CO_2 eq
Tomato	Fresh-market tomato: 12.14; Processing tomato: 121.4	Fresh-market tomato: 108; Processing tomato: 178	Furrow	Standard	N	1.35 ± 0.24[a]	309 ± 55	149 ± 25.8	0.79 ± 0.11	181 ± 25.2	84.8 ± 11.8	5	107
Tomato	Fresh-market tomato: 12.14; Processing tomato: 121.4	Fresh-market tomato: 108; Processing tomato: 178	SDI	Reduced	N	0.46 ± 0.02	105 ± 4.34	49 ± 2.03	0.31 ± 0.02	71.1 ± 4.58	33.3 ± 2.15	1	107
Tomato	Fresh-market tomato: 12.14; Processing tomato: 121.4	Fresh-market tomato: 108; Processing tomato: 178	SDI	Standard	N	0.70[b]	160	75.1	0.41b	94	44	1	107
Tomato	Fresh-market tomato: 12.14; Processing tomato: 121.4	Fresh-market tomato: 108; Processing tomato: 178	Furrow	Standard	Y	1.41[c]	323	151	1.01c	231	108	1	107
Tomato	Fresh-market tomato: 12.14; Processing tomato: 121.4	Fresh-market tomato: 108; Processing tomato: 178	SDI	Standard	Y	1.01[c]	231	108	0.72c	165	77.3	1	107
Strawberry	13.4	110	Drip	Standard	N							0	6.9
Lettuce	131	166	Sprinkler/Drip	Standard	N	0.75 ± 0.27	163±59	76.3±27.5	0.61 ± 0.04	133 ± 8.70	62.1 ± 4.07	9	102
Lettuce	131	166	Sprinkler/Drip	Standard	Y	0.42	91.3	42.8	0.32	69.6	32.6	1	102
Broccoli	49	204	Sprinkler/Furrow/Drip	Standard	na	na	na	na	na	na	na	0	46.8
Cauliflowe	13.2	196	Sprinkler/Furrow/Drip	Standard	na	na	na	na	na	na	na	0	12.1
Cabbage	5.14	228	Sprinkler/Furrow/Drip	Standard	na	na	na	na	na	na	na	0	5.49
Grape	Wine grapes: 249; raisin grapes: 77.7; table grapes: 49.3	Wine grapes: 25; raisin grapes: 47; table grapes: 42	Drip	Standard	N	3.79 ± 2.67	437 ± 308	205 ± 144	na	na	na	3	55.9
Grape	Wine grapes: 249; raisin grapes: 77.7; table grapes: 49.3	Wine grapes: 25; raisin grapes: 47; table grapes: 42	Drip	Standard	Y	3.01 ± 1.32	347 ± 152	162 ± 71.3	na	na	na	6	55.9
Grape	Wine grapes: 249; raisin grapes: 77.7; table grapes: 49.3	Wine grapes: 25; raisin grapes: 47; table grapes: 42	Drip	Reduced /No till	Y	3.47 ± 2.54	400 ± 293	187 ± 137	na	na	na	3	55.9
Almond	450	150	Drip	No till	N	0.43±0.15	290±22.8	136±10.7	0.31 ± 0.04	209 ± 27.0	98.0 ± 12.6	6	316
Almond	450	150	Mircrosprinkler	No till	N	0.25±0.05	169±33.8	79±15.8	0.19 ± 0.01	128 ± 6.75	60.1 ± 3.16	5	316
Pistachio	126	131	Drip	No till	N	0.43±0.15*	71.0±24.8	33.2±11.6	0.31 ± 0.04*	51.2 ± 6.60	24.0 ± 3.09	0	77.3
Pistachio	126	131	Mircrosprinkler	No till	N	0.25±0.05*	41.3±8.25	19.3±3.86	0.19 ± 0.01*	31.4 ± 1.65	14.7 ± 0.77	0	77.3
Walnut	148	122	Drip	No till	N	0.43±0.15*	77.6±27.1	36.3±12.7	0.31 ± 0.04*	56.0 ± 7.22	26.2 ± 3.38	0	84.6
Walnut	148	122	Mircrosprinkler	No till	N	0.25±0.05*	45.1±9.03	21.1±4.23	0.19 ± 0.01*	34.3 ± 1.81	16.1 ± 0.85	0	84.6

[a] Value is averaged from Kennedy et al (2013) and Burger and Horwath (2012)
[b] Value is calculated from equation 1
[c] Value is calculated from equation 2
* Value is adopted from almond system
Note: the EFs present in this table were calculated based on limited studies.

Table 7. Wine grape production classes, stratefied by growing degree day requirements

Wine Class	Minimum Growing Degree Days	Maximum Growing Degree Days
R1	1,111	1,389
R2	1,390	1,667
R3	1,668	1,944
R4	1,945	2,222

Table 8. DayCent model input parameterization statistics for almonds

Biomass Fraction	Slope	RMSE	Correlation
Large Wood	0.98	33.2	-0.95
Fine Branches	1.03	29.26	0.14
Leaves	0.8	39.26	-0.41
Fruit	0.79	86.76	0.77

Table 9. Model verification results for lettuce

Crop	Mean Measured Yield from CDFA (g C m^{-2})	Mean DayCent Model Yield (g C m^{-2})	RMSE	Correlation Coefficient
Head Lettuce	90	90	17.28	-26%
Leaf Lettuce	60	60	16.85	-4%
Romaine Lettuce	72	71	17.06	9%