White Paper

6 July 2018

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}

^aDepartment of Land, Air & Water Resource, University of California, Davis, CA 95616 ^bNatural Resource Ecology Laboratory, Colorado State University, Ft. Collins, CO ^cUSDA-ARS, Crops Pathology and Genetics Research Unit, Davis, CA 95616 ^dUSDA California Climate Hub, Davis, CA 95616 Contact: kerri.steenwerth@ars.usdsa.gov

This paper is the intellectual contriution of the above named authors and shared by the USDA Climate Hub as a service to parties interested in assessment, research and evaluation of the topic. It has been intependently peer-reviewed by three qualified scientists. It is the intension of the authors to publish this content in several peer-reviewed artiles. All queries on this contenct should be directed to the authors.

Suggested citation: Zhu-Barker, X., Carlson, M., Easter, M., Swan, A., Thompson, L., Horwath, W.R., Paustian, K., Steenwerth, K.L. 2018. Identifying Greenhouse Gas Emission Sources and Mitigation Opportunities in California Specialty Cropping Systems. USDA California Climate Hub. 6 July 2018.

1

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.

Table of Contents

Excutive summary	3
1. Introduction	6
2. Overview of nitrous oxide sources and producton pathways in cropping systems	6
3. General effects of management practices on greenhouse gas emissions	6
3.1 Fertilization	7
3.1.1 Fertilizer rate	8
3.1.2 Fertilizer type	10
3.1.3 Fertilizer placement	12
3.1.4 Fertilizer application timing	13
3.1.5 Fertilizer efficiency enhancers	14
3.2 Irrigation	16
3.3 Tillage	19
3.4 Cover crops	21
4. Contribution of changes in soil carbon stocks to greenhouse gas emissions	23
5. Greenhouse gas emissions from California specialty annual crops	24
5.1 Method for estimating EF and state-wide emissions	25
5.2 Tomatoes	26
5.2.1 Fertilization	26
5.2.2 Irrigation	28
5.2.3 Tillage	30
5.2.4 Cover crops	31
5.3 Strawberries	33
5.3.1 Fertilization	34
5.3.2 Irrigation	35
5.3.3 Tillage	35
5.3.4 Cover crops	36
5.4 Cool season vegetable – Lettuce	36
5.4.1 Fertilization	37
5.4.2. Irrigation	38
5.4.3. Tillage	38
5.4.4 Cover crops	39
5.5 Other cool season vegetables, broccoli, cauliflower, and cabbage	39
5.5.1 Fertilization	40
5.5.2. Irrigation	41
5.5.3. Tillage	41
6. Greenhouse gas emissions from California specialty woody perennials: Fruit and nut crops4	42
6.1 Grapes	42
6.1.1 Fertilization	43
6.1.2 Irrigation	44
6.1.3 Tillage	45
6.1.4 Cover crops	46
6.2 Nut tree crops- Almonds, pistachios, and walnuts	47
6.2.1 Fertilization	47
6.2.2 Irrigation	49

6.2.3 Tillage and cover crops	50
6.3 Stone Fruit	51
6.4 Citrus	51
7. The development of DayCent model and its application on California specialty crops	51
7.1 Modeling woody crops in DayCent	53
7.2 Wine Grapes	55
7.3 Almonds	58
7.4 Lettuce	59
8. Literature summary and gaps in GHG emission knowledge from California specialty crop	ps60
Acknowledgements	63
References	64
Appendix 1. Overview of nitrous oxide sources and production pathways in cro	opping
systems	100
1.1 Ammonia oxidation pathways	100
1.2 Heterotrophic denitrification	102
1.3 Abiotic chemodenitrification	104
Appendix 2. Contribution of changes in soil carbon stocks to greenhouse gas emissions	104
2.1 Fertilization	105
2.2 Tillage	106
2.3 Irrigation	108
2.4 Cover crops and organic amendments	109
2.5 Cropping system conversion	111
Appendix 3. Calculations for Emissions Factors	112

1. Introduction

Agriculture contributes about 8.3% to total U.S. greenhouse gas (GHG) emissions, mainly in the form of nitrous oxide (N₂O) and methane (CH₄), and adds approximately 80% to total U.S. N₂O 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 CO₂ (IPCC, 2013). As such, development of practices to reduce N₂O 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 CH₄ 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 N₂O 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 N₂O emissions so that such factors can be adjusted by agricultural management practices to reduce N₂O 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 N₂O 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 N₂O 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.*, 2015c). California's Mediterranean climate influences the adoption of these management practices and associated soil N₂O 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 N₂O 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 N₂O 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

8

drive N₂O 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 N₂O emissions is also soil- and climate-specific since variations in temperature and soil water content under different climates can significantly affect N₂O 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 N₂O emissions from agricultural systems.

3.1.1 Fertilizer rate

The application rate of N fertilizer strongly influences N₂O emissions (Grant *et al.*, 2006; Zebarth *et al.*, 2008b; Burger and Horwath, 2012; Zhu *et al.*, 2014). Generally, the relationship between N rate and N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O accounting method adopted by IPCC, whereby N₂O emissions are assumed to be a simple fraction of N inputs, has the potential to underestimate or overestimate fertilizer derived N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O emissions from soils is influenced by the form of available N substrates, i.e. NH4⁺, NO3⁻ or organic N, and short- and long-term changes in soil pH after application. In California, commonly used synthetic N fertilizers include ureaammonium-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 N₂O emissions during denitrification (Koskinen and Keeney, 1982; Mulvaney et al., 1997), likely because the substrate affinity of nitrate reductase is higher than that of N₂O 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 N₂O 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 N₂O emissions after applications of alkaline-forming fertilizers is difficult to ascertain because DOC can also promote the consumption of N₂O to N₂. In comparison to alkaline-forming fertilizers, acidifying fertilizers tend to promote higher N₂O 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 N₂O emissions are varied. In almond orchards, higher N₂O 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 N₂O emissions compared to ammonium sulfate (Zhu-Barker *et al.* 2015c). Higher N₂O 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 N₂O emissions from different fertilizer types and identify consistency of fertilizer effects are needed.

3.1.3 Fertilizer placement

Crop yield and N₂O 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 N₂O 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 N₂O production (Mulvaney et al., 1997). In a greenhouse and field study in Montana, N₂O 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 N₂O emissions than a surface broadcast application. In Minnesota, Maharjan and Venterea (2013) also reported significantly higher N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O emissions, in part due to the interactive effects of climate and crop conditions. Changing fertilizer application to sidedress 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 N₂O emissions (Zebarth et al. 2008a). Venterea and Coulter (2015) also showed that later in-season N application does not always reduce N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O emissions were higher and less effective under lower background emission rates (Burger and Horwath, 2012). In contrast, no change in N₂O 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 N₂O emissions (data not published), most likely because of low background N₂O emissions from these systems. To date, no field trials have been conducted to investigate the effect of urease inhibitors on N₂O 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 N₂O 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 N₂O emissions is examined briefly. During irrigation, soil moisture approaches saturation and oxygen diffusion is limited. This results in anoxic conditions that promote N₂O production (Zhu *et al.*, 2013a), but once the soil is completely flooded only trace amounts of N₂O can be emitted to the atmosphere. At that point, the produced N₂O is trapped in the water and is further reduced to N₂ before diffusing to the atmosphere (Davidson, 1991; Dunfield *et al.*, 1995). Optimal moisture conditions for N₂O 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 N₂O 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 N₂O 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 N₂O emissions as extremely anoxic conditions can lead to complete denitrification (Firestone *et al.*, 1979; Bonin *et al.*, 1989). For example, less N₂O 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 N₂O and CO₂ 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 N₂O emissions from furrow irrigation systems can be very high. For example, significantly higher N₂O 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 N₂O 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 N₂O 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 N₂O by 1.95 kg N ha⁻¹yr⁻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 N₂O 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 N₂O fluxes are expected to be similar. However, N₂O 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 N₂O 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 N₂O 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

20

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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 μ m (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

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 N2O 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 N₂O emissions from the soil surface, Basche et al. (2014) reported that over half of the studies observed higher N₂O emissions from soils under cover crops than without cover crops. The incorporation of cover crops into the soil also promoted N₂O 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 N₂O 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 N₂O emissions observed during cover crops decreased emissions, underscoring the importance of long-term (i.e. entire year) monitoring to quantify the effect of cover crops on N₂O emissions.

Poor synchronization of mineralized N availability from cover crop decomposition and uptake by subsequent crops can lead to increases in soil N₂O production. Excess N, combined with available soil C, can increase N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O EFs (both uncorrected and corrected for background flux) and estimated state level emissions for each specialty cropping system under different management practices.

1 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 (%)= (N₂O-N / input N) $\times 100$

- In this report, if the N₂O background (no fertilizer input) emission is available in the literature, the EF calculation is adjusted as follows: adjusted EFs (%)= [(N₂O-N_{fertilizer treatment} N₂O-N_{background}) / input N] ×100.
- 3. If a study does not investigate N₂O 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 (%)= [N₂O-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 N₂O emission baseline for each cropping system is defined as N₂O 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.
- 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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 wellmanaged and growers use the recommended N application rates (108 and 178 kg N ha⁻¹ for freshmarket and processing tomatoes, respectively), the total annual N₂O emission is 149 ± 25.8 Gg CO₂eq (calculated using uncorrected EF), while 84.8 ± 11.8 Gg CO₂eq N₂O 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 N₂O flux from the furrow irrigation treatment in rainy seasons tended to be higher than from SDI. When a cover crop was planted, the mean N₂O flux from furrow irrigation was two times higher than from SDI, suggesting that SDI offers the potential to decrease N₂O emissions from tomato systems under cover crop management. Using the mean value of N₂O 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 N₂O 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 N₂O 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 N₂O 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 N₂O emission from California tomato systems will be approximately 75 Gg CO₂eq, 44 Gg CO₂eq 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 CO₂eq N₂O will be released, 84.8 Gg CO₂eq N₂O 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, N₂O emissions tended to be reduced in SDI compared to furrow irrigation (Kallenbach *et al.*, 2010), meriting further investigation of SDI's potential to reduce N₂O 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 N₂O fluxes

compared to standard tillage (12 to 15 tractor passes) across conventional, low-input, and organic systems. However, robust annual N₂O 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 N₂O emissions from tomato systems under minimum tillage practices by adopting the background uncorrected EF ($0.46 \pm 0.02\%$) reported by Kennedy et al. (2013). Therefore, if all California tomato systems were managed under minimum tillage combined with SDI practices, the total annual N₂O emission from these systems would be approximately 49±2.03 Gg CO₂eq, 33.3 Gg CO₂eq N₂O 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 CO₂eq N₂O would be released, 44 Gg CO₂eq N₂O of which would come from the N inputs (calculated using corrected EF) (Table 6). No study has been conducted yet to compare N₂O 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 N₂O emissions. For example, Kallenbach et al. (2010) found significantly higher N₂O 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 N₂O emissions from tomato systems under cover cropping management. Assuming that the difference in N₂O 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 N₂O emissions

33

from these systems would be approximately 108 Gg CO₂eq (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 N₂O 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 CO₂ ha⁻¹ yr⁻¹ in tomato systems under cover cropping practices (Veenstra et al., 2007), the global warming potential is -285 Gg CO₂eq when SDI and standard tillage practices are adopted and -242 Gg CO₂eq 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 (17-0-8Ca) 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°C) and cool nights (7.2°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 N₂O emissions ranged from 0.16 to 1.15 kg N ha⁻¹. The scaled annual N₂O 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 N₂O emissions under different N application rates which ranged from 84 to 336 kg N ha⁻¹ in lettuce fields. They reported annual N₂O 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 N₂O emissions from California lettuce systems is 76.3±27.5 Gg CO₂eq (Table 6) and the N inputs produce 62.1 ± 4.07 N₂O (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 N₂O 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 N₂O 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 N₂O emissions will be 42.8 Gg CO₂eq, 32.6 Gg CO₂eq 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 18.3-20°C for cauliflower (Koike *et al.*, 2009) and 15.6-18.3°C 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 N₂O 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 NO₃⁻ 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 NO₃⁻ pollution and N₂O 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 N₂O emissions from California wine grape vineyards have been examined on numerous occasions. In Napa County, Smart et al (2006) reported that N₂O emissions increased from 0.03 to 0.09 kg N₂O-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 N₂O emission rate was 0.13 kg N₂O-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⁻¹in raisin, wine grape, and table grape cropping systems, respectively) and manage grape cropping systems under standard tillage combined with drip irrigation, the annual N2O 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 N₂O emissions from different irrigation practices in California grape cropping systems. The majority of wine grapes and raisin grapes are drip irrigated. Thus, the annual N₂O 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 N₂O emissions from vineyards under no-till practices were greater than standard tillage, with 0.12 kg N₂O–N ha⁻¹ growing season⁻¹ emitted from standard tillage compared to 0.18 kg N₂O–N ha⁻¹ from no-till. However, Wolff (2015) observed higher N₂O 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 N₂O 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} N₂O 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 N₂O from soil, whereas the bare soil emitted 0.56 kg N ha⁻¹ N₂O. Steenwerth and Belina (2008) and Wolff (2015) also observed that N₂O 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 N₂O 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} N₂O 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, N₂O 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 N₂O emissions from an almond orchard, with higher N₂O 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 N₂O emissions from UAN than from KNO₃. 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 N₂O emissions from these systems will be 136 ± 10.7 Gg CO₂eq, 98 ± 12.6 CO₂eq (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 N₂O 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 N₂O 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 N₂O 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 CO₂eq N₂O (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 microirrigation 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 N₂O 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 N₂O 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 N₂O emissions from these systems will be 79 ± 15.8 Gg CO₂eq, 60.1 Gg CO₂eq of which will come from the N inputs (calculated using corrected EF). If these systems are managed under drip irrigation, 136± 10.7 Gg CO₂eq N₂O will be released, 98.0 Gg CO₂eq N₂O 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 N₂O emissions from these systems would be 19.3 ± 3.86 CO₂eq, 14.7 Gg CO₂eq 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 CO₂eq N₂O would be released, 24.0 Gg CO₂eq N₂O 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 N₂O emissions would be 21.1 ± 4.23 Gg CO₂eq , 16.1 Gg CO₂eq 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 CO₂eq N₂O would be released, 26.2 Gg CO₂eq 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 CO₂eq N₂O would be released, 26.2 Gg CO₂eq 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 N₂O emissions from California nut tree cropping systems as affected by the adoption of cover crops, except one study which reported N₂O 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 N₂O 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 N₂O 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, N₂O emissions, NO₃ leaching, and plant productivity, including predicting SOC stock changes and N₂O emissions for the U.S. National GHG Inventory (EPA 2017) and estimating field-level SOC stock changes and N₂O 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 nonwoody 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 (<u>https://quickstats.nass.usda.gov/</u>) 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

57

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 $< 30 \text{ kg N ha}^{-1}$ 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 identified California "Crush Report" dataset ___ in as the (https://www.nass.usda.gov/Statistics_by_State/California/Publications/Grape_Crush/Final/2016 <u>/201603gcbtb00.pdf)</u>- 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 DayCentmodeled 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 N₂O 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 N₂O 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 N2O 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 N₂O 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 N₂O 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 N₂O 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 N₂O 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.

Acknowledgements

We thank the California Department of Food and Agriculture, USDA-OCE Climate Change Programs Office, USDA Natural Resource Conservation Service, USDA Forest Service, USDA Agricultural Research Service, USDA Farm Services Agency, USDA California Climate Hub and the University of California at Davis for providing funding to support this work. We are indebted to the many extension agents, researchers and growers who provided information and data needed to parameterize the DayCent model. Finally, we are grateful to USDA-ARS research scientists, Dr. Steven Del Grosso, Dr. Michel Cavigelli and Dr. Rod Venterea, for their careful and critical reviews of this work.

References

- Abdalla, M., Osborne, B., Lanigan, G., Forristal, D., Williams, M., Smith, P., Jones, M., 2013. Conservation tillage systems: a review of its consequences for greenhouse gas emissions. Soil Use Manage 29, 199-209.
- Agricultural Marketing Service, U., 2016. USDA Definition of Specialty Crops. Retrieved June 1, 2016, from http://www.ams.usda.gov/AMSv1.0/scbgpdefinitions.
- Aguilera, E., Lassaletta, L., Sanz-Cobena, A., Garnier, J., Vallejo, A., 2013. The potential of organic fertilizers and water management to reduce N 2 O emissions in Mediterranean climate cropping systems. A review. Agriculture, ecosystems & environment 164, 32-52.
- Akiyama, H., Yan, X.Y., Yagi, K., 2010. Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for N2O and NO emissions from agricultural soils: meta-analysis. Global Change Biol 16, 1837-1846.
- Alcántara, V., Don, A., Well, R., Nieder, R., 2016. Deep ploughing increases agricultural soil organic matter stocks. Global Change Biol.
- Almaraz, J.J., Mabood, F., Zhou, X.M., Madramootoo, C., Rochette, P., Ma, B.L., Smith, D.L., 2009. Carbon Dioxide and Nitrous Oxide Fluxes in Corn Grown under Two Tillage Systems in Southwestern Quebec. Soil Sci Soc Am J 73, 113-119.
- Alsina, M.M., Fanton-Borges, A.C., Smart, D.R., 2013. Spatiotemporal variation of event related N2O and CH4 emissions during fertigation in a California almond orchard. Ecosphere 4, 1-21.
- Álvaro-Fuentes, J., Lopez, M., Arrue, J., Moret, D., Paustian, K., 2009. Tillage and cropping effects on soil organic carbon in Mediterranean semiarid agroecosystems: Testing the Century model. Agriculture, ecosystems & environment 134, 211-217.

- Álvaro-Fuentes, J., Paustian, K., 2011. Potential soil carbon sequestration in a semiarid Mediterranean agroecosystem under climate change: Quantifying management and climate effects. Plant and soil 338, 261-272.
- Amerine, M., Winkler, A., 1944. Composition and quality of musts and wines of California grapes. California Agriculture 15, 493-675.
- Aulakh, M., Doran, J., Walters, D., Power, J., 1991. Legume residue and soil water effects on denitrification in soils of different textures. Soil Biology and Biochemistry 23, 1161-1167.
- Avrahami, S., Bohannan, B.J.A., 2007. Response of Nitrosospira sp strain AF-Like ammonia oxidizers to changes in temperature, soil moisture content, and fertilizer concentration. Appl Environ Microb 73, 1166-1173.
- Avrahami, S., Conrad, R., Braker, G., 2002. Effect of soil ammonium concentration on N2O release and on the community structure of ammonia oxidizers and denitrifiers. Appl Environ Microb 68, 5685-5692.
- Avrahami, S., Liesack, W., Conrad, R., 2003. Effects of temperature and fertilizer on activity and community structure of soil ammonia oxidizers. Environ Microbiol 5, 691-705.
- Avrhami, S., Conrad, R., 2003. Patterns of community change among ammonia oxidizers in meadow soils upon long-term incubation at different temperatures. Appl Environ Microb 69, 6152-6164.
- Bai, E., Houlton, B., Wang, Y., 2012. Isotopic identification of nitrogen hotspots across natural terrestrial ecosystems. Biogeosciences 9, 3287-3304.
- Baker, J., Southard, R., Mitchell, J., 2005. Agricultural dust production in standard and conservation tillage systems in the San Joaquin Valley. J Environ Qual 34, 1260-1269.

- Baker, J.M., Ochsner, T.E., Venterea, R.T., Griffis, T.J., 2007. Tillage and soil carbon sequestration—What do we really know? Agriculture, Ecosystems & Environment 118, 1-5.
- Baldocchi, D., Wong, S., 2008. Accumulated winter chill is decreasing in the fruit growing regions of California. Climatic Change 87, 153-166.
- Basche, A.D., Miguez, F.E., Kaspar, T.C., Castellano, M.J., 2014. Do cover crops increase or decrease nitrous oxide emissions? A meta-analysis. Journal of Soil and Water Conservation 69, 471-482.
- Bateman, E.J., Baggs, E.M., 2005. Contributions of nitrification and denitrification to N₂O emissions from soils at different water-filled pore space. Biol Fert Soils 41, 379-388.
- Belanger, G., Richards, J., Angers, D., 1999. Long-term fertilization effects on soil carbon under permanent swards. Can J Soil Sci 79, 99-102.
- Belmonte, S., R.J. Stahel, E. Bonifacio, L. Celi, V. Novello, E. Zanini, K.L. Steenwerth . Effects of Long-term Soil Management on the Mutual Interaction among Soil Organic Matter, Microbial Activity and Aggregate Stability within a Vineyard. Pedosphere. In press.
- Betlach, M.R., Tiedje, J.M., 1981. Kinetic explanation for accumulation of nitrite, nitric oxide, and nitrous oxide during bacterial denitrification. Appl Environ Microb 42, 1074-1084.
- Bonin, P., Gilewicz, M., Bertrand, J.C., 1989. Effects of Oxygen on Each Step of Denitrification on Pseudomonas-Nautica. Can J Microbiol 35, 1061-1064.
- Bouman, O., Curtin, D., Campbell, C., Biederbeck, V., Ukrainetz, H., 1995. Soil acidification from long-term use of anhydrous ammonia and urea. Soil Sci Soc Am J 59, 1488-1494.
- Bouwman, A., Boumans, L., Batjes, N., 2002a. Emissions of N2O and NO from fertilized fields: Summary of available measurement data. Global Biogeochemical Cycles 16.

- Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002b. Modeling global annual N2O and NO emissions from fertilized fields. Global Biogeochemical Cycles 16, 1-8.
- Breemen, N.v., Feijtel, T., Bouwman, A., 1990. Soil processes and properties involved in the production of greenhouse gases, with special relevance to soil taxonomic systems. Soils and the greenhouse effect. John Wiley and Sons Ltd, pp. 195-223.
- Breitenbeck, G.A., Bremner, J.M., 1986. Effects of Rate and Depth of Fertilizer Application on Emission of Nitrous-Oxide from Soil Fertilized with Anhydrous Ammonia. Biol Fert Soils 2, 201-204.
- Bremner, J.M., Blackmer, A.M., Waring, S.A., 1980. Formation of Nitrous-Oxide and Dinitrogen by Chemical Decomposition of Hydroxylamine in Soils. Soil Biol Biochem 12, 263-269.
- Brennan, E.B., Acosta-Martinez, V., 2017. Cover cropping frequency is the main driver of soil microbial changes during six years of organic vegetable production. Soil Biology and Biochemistry 109, 188-204.
- Brennan, E.B., Boyd, N.S., 2012. Winter cover crop seeding rate and variety affects during eight years of organic vegetables: I. Cover crop biomass production. Agron J 104, 684-698.
- Brennan, E.B., Boyd, N.S., Smith, R.F., 2013. Winter cover crop seeding rate and variety effects during eight years of organic vegetables: III. Cover crop residue quality and nitrogen mineralization. Agron J 105, 171-182.
- Breuillin-Sessoms, F., Venterea, R.T., Sadowsky, M.J., Coulter, J.A., Clough, T.J., Wang, P., 2017. Nitrification gene ratio and free ammonia explain nitrite and nitrous oxide production in urea-amended soils. Soil Biology and Biochemistry 111, 143-153.

- Broder, M., Doran, J., Peterson, G., Fenster, C., 1984. Fallow tillage influence on spring populations of soil nitrifiers, denitrifiers, and available nitrogen. Soil Sci Soc Am J 48, 1060-1067.
- Buresh, R.J., Moraghan, J.T., 1976. Chemical Reduction of Nitrate by Ferrous Iron. J Environ Qual 5, 320-325.
- Burford, J., Bremner, J., 1975. Relationships between the denitrification capacities of soils and total, water-soluble and readily decomposable soil organic matter. Soil biology and biochemistry 7, 389-394.
- Burger, M., Horwath, W., 2012. Assessment of baseline nitrous oxide emissions in California cropping systems. California Air Resources Board, c.-.(Ed.).
- Burton, D., Li, X., Grant, C., 2008. Influence of fertilizer nitrogen source and management practice on N2O emissions from two Black Chernozemic soils. Can J Soil Sci 88, 219-227.
- Cabrera, M., M. Cavanaugh, Wil, N., 2014. California Drought Forces Farmers To Adapt. in KPBS Radio.
- Cahn, M., 2011. Irrigation management of strawberries. University of California Division of Agriculture and Natural Resources, Editor.
- Calleja-Cervantes, M.E., Fernández-González, A.J., Irigoyen, I., Fernández-López, M., Aparicio-Tejo, P.M., Menéndez, S., 2015. Thirteen years of continued application of composted organic wastes in a vineyard modify soil quality characteristics. Soil Biology and Biochemistry 90, 241-254.
- CalRecycle, 2015. Research to Evaluate Nitrous Oxide (N2O) Emissions from Compost in SupportofAB32ScopingPlanCompostingMeasure.

http://www.calrecycle.ca.gov/Publications/Documents/1544%5C201501544.pdf, Publication #DRRR 2015-01544.

- Campbell, C., Biederbeck, V., Zentner, R., Lafond, G., 1991. Effect of crop rotations and cultural practices on soil organic matter, microbial biomass and respiration in a thin Black Chernozem. Can J Soil Sci 71, 363-376.
- CARB, 2015. California Greenhouse Gas Emissions Inventory. Sacramento, CA: California Air Resources Board, California Environmental Protection Agency. <u>http://www.arb.ca.gov/cc/inventory/doc/docs4/4b_solidwastetreatment_composting_feedstoc_kprocessed_ch4_2012.htm</u>.
- Cayuela, M.L., Aguilera, E., Sanz-Cobena, A., Adams, D.C., Abalos, D., Barton, L., Ryals, R.,
 Silver, W.L., Alfaro, M.A., Pappa, V.A., 2017. Direct nitrous oxide emissions in
 Mediterranean climate cropping systems: emission factors based on a meta-analysis of
 available measurement data. Agriculture, Ecosystems & Environment 238, 25-35.
- CDFA, 2012. Fertilizer Tonnage Reports Sacramento, CA: California Department of Food and Agriculture.
- CDFA, 2015. California Grape Acreage Report, 2014 Summary. https://www.nass.usda.gov/Statistics_by_State/California/Publications/Fruits_and_Nuts/2015 /201503grpac.pdf.

 CDFA,
 2016.
 California
 Processing
 Tomato
 Report.

 https://www.nass.usda.gov/Statistics_by_State/California/Publications/Vegetables/2016/2016
 05ptom.pdf.

CDFA, 2017a. California Agricultural Statistics Review 2015-2016. https://www.cdfa.ca.gov/statistics/PDFs/2016Report.pdf, viewed on 8/15/2017.

- CDFA, 2017b. Fertilizer Research and Education Program. Viewed 8/2/2017 at https://apps1.cdfa.ca.gov/fertilizerresearch/docs/guidelines.html.
- Celette, F., Findeling, A., Gary, C., 2009. Competition for nitrogen in an unfertilized intercropping system: The case of an association of grapevine and grass cover in a Mediterranean climate. Eur J Agron 30, 41-51.
- Chalk, P., Keeney, D., Walsh, L., 1975. Crop recovery and nitrification of fall and spring applied anhydrous ammonia. Agron J 67, 33-37.
- Chalk, P.M., Smith,C.J., 1983. Chemodenitrification. In: Freney, J.R., Simpson, J.R. (Eds),Gaseous loss of nitrogen from plant-soil systems. Development in Plant and Soils Sciences 9,25.
- Chambers, A., Lal, R., Paustian, K., 2016. Soil carbon sequestration potential of US croplands and grasslands: Implementing the 4 per Thousand Initiative. Journal of Soil and Water Conservation 71, 68A-74A.
- Chao, T.-T., Kroontje, W., 1966. Inorganic nitrogen transformations through the oxidation and reduction of iron. Soil Sci Soc Am J 30, 193-196.
- Christensen, L.P., Bianchi, M.L., Peacock, W.L., Hirschfelt, D.J., 1994. Effect of nitrogen fertilizer timing and rate on inorganic nitrogen status, fruit composition, and yield of grapevines. American Journal of Enology and Viticulture 45, 377-387.
- Christopher, S.F., Lal, R., 2007. Nitrogen management affects carbon sequestration in North American cropland soils. Critical Reviews in Plant Sciences 26, 45-64.
- Christopher, S.F., Lal, R., Mishra, U., 2009. Regional study of no-till effects on carbon sequestration in the Midwestern United States. Soil Sci Soc Am J 73, 207-216.

- Collins, H.P., Mikha, M.M., Brown, T.T., Smith, J.L., Huggins, D., Sainju, U.M., 2012. Agricultural Management and Soil Carbon Dynamics: Western U.S. Croplands.
- Commission, C.S., 1999. Crop Profile for Strawberries in California. USDA Regional IPM Centers. Online: http://www/. ipmcenters. org/CropProfiles/docs/c astrawberries. pdf.
- Cooper, G.S., Smith, R., 1963. Sequence of products formed during denitrification in some diverse western soils. Soil Sci Soc Am J 27, 659-662.
- Crews, T., Peoples, M., 2004. Legume versus fertilizer sources of nitrogen: ecological tradeoffs and human needs. Agriculture, Ecosystems & Environment 102, 279-297.
- Culman, S.W., Haden, V.R., Maxwell, T., Waterhouse, H., Horwath, W., 2014. Greenhouse Gas Mitigation Opportunities in California Agriculture: Review of California Cropland Emissions and Mitigation Potential.
- Dalal, R.C., Wang, W.J., Robertson, G.P., Parton, W.J., 2003. Nitrous oxide emission from Australian agricultural lands and mitigation options: a review. Aust J Soil Res 41, 165-195.
- Daugovish, O., Cahn, M., Koike, S., Natwick, E., Cantwell, M., Takele, E., 2008. Cabbage Production in California. University of California Division of Agriculture and Natural Resources.
- Davidson, E.A., 1991. Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems. In:
 Rogers, J.E., Whitman, W.B. (Eds.), Microbial Production and Consumption of Greenhouse
 Gases: Methane, Nitrogen Oxides and Halomethanes. American Society of Microbiology,
 Washington, DC, pp. 219–235.
- Davidson, E.A., 1992. Sources of nitric oxide and nitrous oxide following wetting of dry soil. Soil Sci Soc Am J 56, 95-102.
- Davidson, E.A., 2009. The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. Nat Geosci 2, 659-662.
- de C. Carmello, Q., Anti, G., 2004. Accumulation of nutrients and growth of processing tomato. IX International Symposium on the Processing Tomato 724, pp. 85-90.
- De Gryze, S., Albarracin, M., Catala-Luque, R., Howitt, R., Six, J., 2009. Modeling shows that alternative soil management can decrease greenhouse gases. California agriculture 63, 84-90.
- De Gryze, S., Lee, J., Ogle, S., Paustian, K., Six, J., 2011. Assessing the potential for greenhouse gas mitigation in intensively managed annual cropping systems at the regional scale. Agriculture, Ecosystems and Environment 144, 150-158.
- De Gryze, S., Six, J., Brits, C., Merckx, R., 2005. A quantification of short-term macroaggregate dynamics: influences of wheat residue input and texture. Soil Biology and Biochemistry 37, 55-66.
- De Gryze, S., Six, J., Paustian, K., Morris, S.J., Paul, E.A., Merckx, R., 2004. Soil organic carbon pool changes following land-use conversions. Global Change Biol 10, 1120-1132.
- de Klein, C., Novoa, R.S., Ogle, S., Smith, K., Rochette, P., Wirth, T., McConkey, B., Mosier, A., Rypdal, K., Walsh, M., 2006. N2O emissions from managed soils, and CO2 emissions from lime and urea application. IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme 4, 1-54.
- DeLonge, M.S., Owen, J.J., Silver, W.L., 2014. Greenhouse Gas Mitigation Opportunities in California Agriculture.
- Deschenes, O., Kolstad, C., 2011. Economic impacts of climate change on California agriculture. Climatic Change 109, 365-386.

- Dobbie, K.E., McTaggart, I.P., Smith, K.A., 1999. Nitrous oxide emissions from intensive agricultural systems: Variations between crops and seasons, key driving variables, and mean emission factors. J Geophys Res-Atmos 104, 26891-26899.
- Drinkwater, L.E., Snapp, S., 2007. Nutrients in agroecosystems: rethinking the management paradigm. Advances in Agronomy 92, 163-186.
- Drury, C., Reynolds, W., Tan, C., Welacky, T., Calder, W., McLaughlin, N., 2006. Emissions of nitrous oxide and carbon dioxide. Soil Sci Soc Am J 70, 570-581.
- Dunfield, P.F., Topp, E., Archambault, C., Knowles, R., 1995. Effect of nitrogen fertilizers and moisture content on CH4 and N2O fluxes in a humisol: measurements in the field and intact soil cores. Biogeochemistry 29, 199-222.
- Duval, M.E., Galantini, J.A., Capurro, J.E., Martinez, J.M., 2016. Winter cover crops in soybean monoculture: Effects on soil organic carbon and its fractions. Soil and Tillage Research 161, 95-105.
- Eagle, J., Henry, L.R., Olander, L.P., Haugen-Kozyra, K., Millar, N., Robertson, G.P., 2011. Greenhouse gas mitigation potential of agricultural land management in the United States: A Synthesis of the Literature.A Synthesis of the Literature. Technical Working Group on Agricultural Greenhouse Gases (T-AGG) Report.
- Elias, E., Steele, C., Havstad, K., Steenwerth, K., Chambers, J., Deswood, H., Kerr, A., Rango,A., Schwartz, M., Stine, P., 2015. Southwest regional climate hub and California subsidiaryhub assessment of climate change vulnerability and adaptation and mitigation strategies.
- Elmi, A.A., Madramootoo, C., Hamel, C., Liu, A., 2003. Denitrification and nitrous oxide to nitrous oxide plus dinitrogen ratios in the soil profile under three tillage systems. Biol Fert Soils 38, 340-348.

- Engel, R., Liang, D.L., Wallander, R., Bembenek, A., 2010. Influence of Urea Fertilizer Placement on Nitrous Oxide Production from a Silt Loam Soil. J Environ Qual 39, 115-125.
- ERS, 2016. U.S. Fertilizer Use and Price USDA Economic Research Service. http://www.ers.usda.gov/Data/FertilizerUse/. Accessed on June 22, 2016.
- Eve, M., Sperow, M., Howerton, K., Paustian, K., Follett, R., 2002. Predicted impact of management changes on soil carbon storage for each cropland region of the conterminous United States. Journal of Soil and Water Conservation 57, 196-204.
- Fierer, N., Schimel, J.P., 2002. Effects of drying–rewetting frequency on soil carbon and nitrogen transformations. Soil Biology and Biochemistry 34, 777-787.
- Firestone, M., 1982. Biological denitrification. *In:* Stevenson F J et al. (eds) Nitrogen in Agricultural Soils. Soil Science Society of America, Madison, WI, USA, 38.
- Firestone, M.K., Davidson, E.A., 1989a. Microbial basis of NO and N2O production and consumption in soil. In: Andreae MO, Schimel DS, editors. Exchange of trace gases between terrestrial ecosystems and the atmosphere. Life Sciences Research Reports. Chichester (UK): Wiley. p 7–21.
- Firestone, M.K., Davidson, E.A., 1989b. Microbiological Basis of NO and N2O Production and Consumption. In: Andreae, M.O., Schimel, D.S. (Eds.), Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere. Wiley, New York, pp. 7-21.
- Firestone, M.K., Firestone, R.B., Tiedje, J.M., 1980a. Nitrous oxide from soil denitrification: factors controlling its biological production. American Association for the Advancement of Science.
- Firestone, M.K., Firestone, R.B., Tiedje, J.M., 1980b. Nitrous oxide from soil denitrification: Factors controlling its biological production. Science (Washington DC) 208, 749-751.

- Firestone, M.K., Firestone, R.B., Tiedje, J.M., 1980c. Nitrous-Oxide from Soil Denitrification -Factors Controlling Its Biological Production. Science 208, 749-751.
- Firestone, M.K., Smith, M.S., Firestone, R.B., Tiedje, J.M., 1979. Influence of Nitrate, Nitrite, and Oxygen on the Composition of the Gaseous Products of Denitrification in Soil. Soil Sci Soc Am J 43, 1140-1144.
- Follett, R., 2001. Soil management concepts and carbon sequestration in cropland soils. Soil and Tillage Research 61, 77-92.
- Fortuna, A., Harwood, R., Kizilkaya, K., Paul, E., 2003. Optimizing nutrient availability and potential carbon sequestration in an agroecosystem. Soil Biology and Biochemistry 35, 1005-1013.
- Fortuna, A.M., Honeycutt, C.W., Vandemark, G., Griffin, T.S., Larkin, R.P., He, Z.Q., Wienhold,
 B.J., Sistani, K.R., Albrecht, S.L., Woodbury, B.L., Torbert, H.A., Powell, J.M., Hubbard,
 R.K., Eigenberg, R.A., Wright, R.J., Alldredge, J.R., Harsh, J.B., 2012. Links among
 Nitrification, Nitrifier Communities, and Edaphic Properties in Contrasting Soils Receiving
 Dairy Slurry. J Environ Qual 41, 262-272.
- Franzluebbers, A., Hons, F., Zuberer, D., 1998. In situ and potential CO 2 evolution from a Fluventic Ustochrept in southcentral Texas as affected by tillage and cropping intensity. Soil and Tillage Research 47, 303-308.
- Fred Thomas, Ann Mayse, Chaney, D., 2011. Cover cropping in orchards and vineyards. http://asi.ucdavis.edu/programs/sarep/publications/covercropping-orchardsandvineyards.pdf/.
- Fujinuma, R., Venterea, R.T., Rosen, C., 2011. Broadcast Urea Reduces N2O but Increases NO Emissions Compared with Conventional and Shallow-Applied Anhydrous Ammonia in a Coarse-Textured Soil. J Environ Qual 40, 1806-1815.

- Gagnon, B., Ziadi, N., 2010. Grain corn and soil nitrogen responses to sidedress nitrogen sources and applications. Agron J 102, 1014-1022.
- Gagnon, B., Ziadi, N., Rochette, P., Chantigny, M.H., Angers, D.A., 2011. Fertilizer source influenced nitrous oxide emissions from a clay soil under corn. Soil Sci Soc Am J 75, 595-604.
- Garland, G.M., Suddick, E., Burger, M., Horwath, W., Six, J., 2011. Direct N 2 O emissions following transition from conventional till to no-till in a cover cropped Mediterranean vineyard (Vitis vinifera). Agriculture, ecosystems & environment 144, 423-428.
- Garland, G.M., Suddick, E., Burger, M., Horwath, W., Six, J., 2014. Direct N 2 O emissions from a Mediterranean vineyard: Event-related baseline measurements. Agriculture, Ecosystems & Environment 195, 44-52.
- Gillabel, J., Denef, K., Brenner, J., Merckx, R., Paustian, K., 2007. Carbon sequestration and soil aggregation in center-pivot irrigated and dryland cultivated farming systems. Soil Sci Soc Am J 71, 1020-1028.
- Gillam, K., Zebarth, B., Burton, D., 2008. Nitrous oxide emissions from denitrification and the partitioning of gaseous losses as affected by nitrate and carbon addition and soil aeration. Can J Soil Sci 88, 133-143.
- Glass, J.B., Orphan, V.J., 2012. Trace metal requirements for microbial enzymes involved in the production and consumption of methane and nitrous oxide. Frontiers in microbiology 3, 61.
- Godfrey, L.V., Glass, J.B., 2011. The geochemical record of the ancient nitrogen cycle, nitrogen isotopes, and metal cofactors. Methods Enzymol 486, 483-506.
- Goldhamer, D.A., Viveros, M., Salinas, M., 2006. Regulated deficit irrigation in almonds: effects of variations in applied water and stress timing on yield and yield components. Irrigation Sci 24, 101-114.

- Grandy, A.S., Loecke, T.D., Parr, S., Robertson, G.P., 2006. Long-term trends in nitrous oxide emissions, soil nitrogen, and crop yields of till and no-till cropping systems. J Environ Qual 35, 1487-1495.
- Grandy, A.S., Robertson, G.P., 2007. Land-use intensity effects on soil organic carbon accumulation rates and mechanisms. Ecosystems 10, 59-74.
- Grant, R.F., Pattey, E., Goddard, T.W., Kryzanowski, L.M., Puurveen, H., 2006. Modeling the effects of fertilizer application rate on nitrous oxide emissions. Soil Sci Soc Am J 70, 235-248.
- Gryze, S.D., Wolf, A., Kaffka, S.R., Mitchell, J., Rolston, D.E., Temple, S.R., Lee, J., Six, J., 2010. Simulating greenhouse gas budgets of four California cropping systems under conventional and alternative management. Ecol Appl 20, 1805-1819.
- Guerra, B., Steenwerth, K., 2011. Influence of floor management technique on grapevine growth, disease pressure, and juice and wine composition: a review. American Journal of Enology and Viticulture, ajev. 2011.10001.
- Halvorson, A.D., Del Grosso, S.J., 2013. Nitrogen placement and source effects on nitrous oxide emissions and yields of irrigated corn. J Environ Qual 42, 312-322.
- Halvorson, A.D., Follett, R.F., Reule, C.A., Del Grosso, S., 2009. Soil organic carbon and nitrogen sequestration in irrigated cropping systems of the Central Great Plains. Soil Carbon Sequestration and the Greenhouse Effect, 141-157.
- Halvorson, A.D., Jantalia, C.P., 2011. Nitrogen fertilization effects on irrigated no-till corn production and soil carbon and nitrogen. Agron J 103, 1423-1431.
- Halvorson, A.D., Reule, C.A., Follett, R.F., 1999. Nitrogen fertilization effects on soil carbon and nitrogen in a dryland cropping system. Soil Sci Soc Am J 63, 912-917.

- Halvorson, A.D., Stewart, C.E., 2015. Stover removal affects no-till irrigated corn yields, soil carbon, and nitrogen. Agron J 107, 1504-1512.
- Halvorson, A.D., Wienhold, B.J., Black, A.L., 2002. Tillage, nitrogen, and cropping system effects on soil carbon sequestration. Soil Sci Soc Am J 66, 906-912.
- Hanson, B., Bendixen, W., May, D., 2000. Patterns of soil moisture, soil salinity, and soil nitrate under drip irrigation of row crops. Proceedings of the 2000 National Conference and Exhibition: Water–Essential for Life. Irrigation Association of Australia, Melbourne, AU.
- Hanson, B., May, D., 2007. The effect of drip line placement on yield and quality of drip-irrigated processing tomatoes. Irrigation and Drainage Systems 21, 109-118.
- Hao, X., Chang, C., Carefoot, J., Janzen, H., Ellert, B., 2001. Nitrous oxide emissions from an irrigated soil as affected by fertilizer and straw management. Nutr Cycl Agroecosys 60, 1-8.
- Harrison, R.M., Yamulki, S., Goulding, K.W.T., Webster, C.P., 1995. Effect of fertilizer application on NO and N2O fluxes from agricultural fields. J Geophys Res-Atmos 100, 25923-25931.
- Hart, J.F., 2003. Specialty cropland in California. Geographical Review, 153-170.
- Hartz, T., Miyao, G., Mickler, J., Le Strange, M., Stoddard, S., Nunez, J., Aegerter, B., 2008.Processing tomato production in California. Univ. Calif. Publ. 7228.
- Hartz, T.K., Bottoms, T.G., 2009. Nitrogen requirements of drip-irrigated processing tomatoes. Hortscience 44, 1988-1993.
- Hartz, T.K., Johnstone, P., Williams, E., Smith, R., 2007. Establishing lettuce leaf nutrient optimum ranges through DRIS analysis. Hortscience 42, 143-146.
- Hauck, R., Stephenson, H., 1965. Nitrogen sources, nitrification of nitrogen fertilizers. Effect of nitrogen source, size and pH of granule, and concentration. J Agr Food Chem 13, 486-492.

- Hawkins, S., Robinson, K., Layton, A., Sayler, G., 2010. Limited impact of free ammonia on Nitrobacter spp. inhibition assessed by chemical and molecular techniques. Bioresource Technology 101, 4513-4519.
- Hirschfelt, D.J., 2000. Vineyard floor management. Raisin production manual. Univ. California, Div. Agr. Natural Resources, Publ 3393, 134-138.
- Hoben, J.P., Gehl, R.J., Millar, N., Grace, P.R., Robertson, G.P., 2011. Nonlinear nitrous oxide (N2O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. Global Change Biol 17, 1140-1152.
- Hochmuth, G.J., Albregts, E.E., Chandler, C.C., Cornell, J., Harrison, J., 1996. Nitrogen fertigation requirements of drip-irrigated strawberries. J Am Soc Hortic Sci 121, 660-665.
- Hochstein, L.I., Tomlinson, G.A., 1988. The enzymes associated with denitrification. Annual Reviews in Microbiology 42, 231-261.
- Hooper, A.B., Terry, K.R., 1979. Hydroxylamine Oxidoreductase of Nitrosomonas Production of Nitric-Oxide from Hydroxylamine. Biochim Biophys Acta 571, 12-20.
- Howitt, R., Medellín-Azuara, J., MacEwan, D., Lund, J., Sumner, D., 2014. Economic analysis of the 2014 drought for California agriculture. Davis, CA: UC–Davis Center for Watershed
 Sciences. Online at <u>https://watershed/</u>. ucdavis. edu/files/biblio/DroughtReport_23July2014_0. pdf.
- Huggins, D., Fuchs, D., 1997. Long-term N management effects on corn yield and soil C of an Aquic Haplustoll in Minnesota. Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America, 121-128.
- Hultgreen, G., 2003. effect of nitrogen fertilizer placement, formulation, timing, and rate on greenhouse gas emissions and agronomic performance.

- Hutmacher, R., Nightingale, H., Rolston, D., Biggar, J., Dale, F., Vail, S., Peters, D., 1994. Growth and yield responses of almond (Prunus amygdalus) to trickle irrigation. Irrigation Sci 14, 117-126.
- Hyatt, C.R., Venterea, R.T., Rosen, C.J., McNearney, M., Wilson, M.L., Dolan, M.S., 2010. Polymer-coated urea maintains potato yields and reduces nitrous oxide emissions in a Minnesota loamy sand. Soil Sci Soc Am J 74, 419-428.

Institute, A., Sciences, C.o.A.a.E., University of California, D., 2012.

- IPCC, 2006. Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan.
- IPCC, 2007. Mitigation of climate change. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (Eds.), Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.Cambridge Univ Press, Cambridge, UK. Climate Change 2007.
- IPCC, 2013. Anthropogenic and natural radiative forcing. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- Jantalia, C.P., Halvorson, A.D., 2011. Nitrogen fertilizer effects on irrigated conventional tillage corn yields and soil carbon and nitrogen pools. Agron J 103, 871-878.
- Jarecki, M.K., Lal, R., 2003. Crop management for soil carbon sequestration. Critical Reviews in Plant Sciences 22, 471-502.

- Jarecki, M.K., Lal, R., James, R., 2005. Crop management effects on soil carbon sequestration on selected farmers' fields in northeastern Ohio. Soil and tillage research 81, 265-276.
- Kallenbach, C.M., Rolston, D.E., Horwath, W.R., 2010. Cover cropping affects soil N2O and CO2 emissions differently depending on type of irrigation. Agr Ecosyst Environ 137, 251-260.
- Katayanagi, N., Furukawa, Y., Fumoto, T., Hosen, Y., 2012. Validation of the DNDC-rice model by using CH4 and N2O flux data from rice cultivated in pots under alternate wetting and drying irrigation management. Soil Sci Plant Nutr 58, 360-372.
- Keeney, D., Fillery, I., Marx, G., 1979. Effect of temperature on the gaseous nitrogen products of denitrification in a silt loam soil. Soil Sci Soc Am J 43, 1124-1128.
- Kendall, A., Marvinney, E., Brodt, S., Zhu, W., 2015. Life cycle–based assessment of energy use and greenhouse gas emissions in almond production, part I: analytical framework and baseline results. Journal of Industrial Ecology 19, 1008-1018.
- Kennedy, T.L., 2011. Nitrous Oxide Emissions from California Tomato Cropping Systems under Conventional and Alternative Management. University of California, Davis.
- Kennedy, T.L., Suddick, E.C., Six, J., 2013. Reduced nitrous oxide emissions and increased yields in California tomato cropping systems under drip irrigation and fertigation. Agriculture, ecosystems & environment 170, 16-27.
- Kessavalou, A., Mosier, A.R., Doran, J.W., Drijber, R.A., Lyon, D.J., Heinemeyer, O., 1998. Fluxes of carbon dioxide, nitrous oxide, and methane in grass sod and winter wheat-fallow tillage management. J Environ Qual 27, 1094-1104.
- Kessel, C., Venterea, R., Six, J., Adviento-Borbe, M.A., Linquist, B., Groenigen, K.J., 2013. Climate, duration, and N placement determine N2O emissions in reduced tillage systems: a meta-analysis. Global Change Biol 19, 33-44.

- Kieft, T.L., Soroker, E., Firestone, M.K., 1987. Microbial biomass response to a rapid increase in water potential when dry soil is wetted. Soil biology & biochemistry [0038-0717] 19, 119-126.
- Kirschbaum, D.S., Borquez, A.M., Quipildor, S.L., Correa, M., Magen, H., Imas, P., 2006. Nitrogen requirements of drip irrigated strawberries grown in subtropical environments. Acta Horticulturae 708, 93.
- Knowles, R., 1982. Denitrification. Microbiol Rev 46, 43-70.
- Koike, S., Schulbach, K., Chaney, W., 2009. Cauliflower production in California. University of California Division of Agriculture and Natural Resources.
- Kong, A.Y., Fonte, S.J., van Kessel, C., Six, J., 2009. Transitioning from standard to minimum tillage: trade-offs between soil organic matter stabilization, nitrous oxide emissions, and N availability in irrigated cropping systems. Soil and Tillage Research 104, 256-262.
- Kong, A.Y., Six, J., Bryant, D.C., Denison, R.F., Van Kessel, C., 2005. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. Soil Sci Soc Am J 69, 1078-1085.
- Koskinen, W., Keeney, D., 1982. Effect of pH on the rate of gaseous products of denitrification in a silt loam soil. Soil Sci Soc Am J 46, 1165-1167.
- Kroodsma, D.A., Field, C.B., 2006. Carbon sequestration in California agriculture, 1980–2000.Ecol Appl 16, 1975-1985.
- Krusekopf, H., Mitchell, J., Hartz, T., May, D., Miyao, E., Cahn, M., 2002. Pre-sidedress soil nitrate testing identifies processing tomato fields not requiring sidedress N fertilizer. HortScience 37, 520-524.
- Lakso, A., Flore, J., Singha, S., Baugher, T., 2003. Carbohydrate partitioning and plant growth. Concise encyclopedia of temperate tree fruit. Food Products Press, New York, NY, 21-30.

- Lal, R., 2003. Soil erosion and the global carbon budget. Environment international 29, 437-450.
- Lal, R., Griffin, M., Apt, J., Lave, L., Morgan, M.G., 2004. Managing soil carbon. Science 304, 393-393.
- Lal, R., Kimble, J.M., Follett, R.F., Cole, C.V., 1998. The potential of US cropland to sequester carbon and mitigate the greenhouse effect. CRC Press.
- Le Strange, M., Cahn, M., Koike, S., SMITH, R., 2010a. Broccoli Production in Caifornia. UCANR Publications.
- Le Strange, M., Cahn, M., Koike, S., Smith, R., Daugovish, O., Fennimore, S., Natwick, E., Dara,K., Takele, E., Cantwell, M., 2010b. Broccoli Production in california. University of CaliforniaDivision of Agriculture and Natural Resources.
- Le Strange, M., Schrader, W., Hartz, T., 2000. Fresh-market tomato production in California. UCANR Publications.
- Levy, M., Christian-Smith, J., 2011. Groundwater Management in the Pajaro Valley. The Pacific Institute, Oakland, Californie, États-Unis.
- Li, L.-J., Han, X.-Z., 2016. Changes of soil properties and carbon fractions after long-term application of organic amendments in Mollisols. Catena 143, 140-144.
- Liebig, M., Gross, J., Kronberg, S., Phillips, R., 2010. Grazing management contributions to net global warming potential: A long-term evaluation in the Northern Great Plains. J Environ Qual 39, 799-809.
- Liebig, M., Johnson, H., Hanson, J., Frank, A., 2005. Soil carbon under switchgrass stands and cultivated cropland. Biomass and Bioenergy 28, 347-354.
- Lobell, D.B., Schlenker, W., Costa-Roberts, J., 2011. Climate trends and global crop production since 1980. Science 333, 616-620.

- Lopus, S., Santibáñez, M., Beede, R., Duncan, R., Edstrom, J., Niederholzer, F., Trexler, C., Brown, P., 2010. Survey examines the adoption of perceived best management practices for almond nutrition. California agriculture 64, 149-154.
- Ma, B., Wu, T., Tremblay, N., Deen, W., Morrison, M., McLaughlin, N., Gregorich, E., Stewart, G., 2010. Nitrous oxide fluxes from corn fields: on-farm assessment of the amount and timing of nitrogen fertilizer. Global Change Biol 16, 156-170.
- Maharjan, B., Venterea, R.T., 2013. Nitrite intensity explains N management effects on N 2 O emissions in maize. Soil Biology and Biochemistry 66, 229-238.
- Maharjan, B., Venterea, R.T., Rosen, C., 2014. Fertilizer and irrigation management effects on nitrous oxide emissions and nitrate leaching. Agron J 106, 703-714.
- Marriott, E.E., Wander, M.M., 2006. Total and labile soil organic matter in organic and conventional farming systems. Soil Sci Soc Am J 70, 950-959.
- Marvinney, E., Kendall, A., Brodt, S., 2015. Life Cycle–based Assessment of Energy Use and Greenhouse Gas Emissions in Almond Production, Part II: Uncertainty Analysis through Sensitivity Analysis and Scenario Testing. Journal of Industrial Ecology 19, 1019-1029.
- Matson, P.A., Naylor, R., Ortiz-Monasterio, I., 1998. Integration of environmental, agronomic, and economic aspects of fertilizer management. Science 280, 112-115.
- Matthiessen, J.N., Kirkegaard, J.A., 2006. Biofumigation and enhanced biodegradation: opportunity and challenge in soilborne pest and disease management. Critical Reviews in Plant Sciences 25, 235-265.
- McCarty, G., Meisinger, J., 1997. Effects of N fertilizer treatments on biologically active N pools in soils under plow and no tillage. Biol Fert Soils 24, 406-412.

- McSwiney, C.P., Robertson, G.P., 2005. Nonlinear response of N2O flux to incremental fertilizer addition in a continuous maize (Zea mays L.) cropping system. Global Change Biol 11, 1712-1719.
- Meiklejohn, J., 1952. Iron and the nitrifying bacteria. J Gen Microbiol 7, R5-R5.
- Miner, G., Poling, E., Carroll, D., Nelson, L., Campbell, C., 1997. Influence of Fall Nitrogen and Spring Nitrogen—Potassium Applications on Yield and Fruit Quality of Chandler'Strawberry.
 J Am Soc Hortic Sci 122, 290-295.
- Mitchell, J., 2009. Classification of conservation tillage practices in California irrigated row crop systems. UCANR Publications.
- Mitchell, J., Klonsky, K., Miyao, E., Hembree, K., 2009. Conservation tillage tomato production in California's San Joaquin Valley. UCANR Publications.
- Mitchell, J., Klonsky, K., Shrestha, A., Fry, R., DuSault, A., Beyer, J., Harben, R., 2007. Adoption of conservation tillage in California: Current status and future perspectives. Animal Production Science 47, 1383-1388.
- Mitchell, J., Shrestha, A., Irmak, S., 2015a. Trade-offs between winter cover crop production and soil water depletion in the San Joaquin Valley, California. Journal of Soil and Water Conservation 70, 430-440.
- Mitchell, J.P., Klonsky, K.M., Miyao, E.M., Aegerter, B.J., Shrestha, A., Munk, D.S., Hembree,
 K., Madden, N.M., Turini, T.A., 2012. Evolution of conservation tillage systems for processing tomato in California's Central Valley. Horttechnology 22, 617-626.
- Mitchell, J.P., Miyao, G., 2012. Cover Cropping and Conservation Tillage in California Processing Tomatoes. UCANR Publications.

- Mitchell, J.P., Shrestha, A., Horwath, W.R., Southard, R.J., Madden, N., Veenstra, J., Munk, D.S., 2015b. Tillage and cover cropping affect crop yields and soil carbon in the San Joaquin Valley, California. Agron J 107, 588-596.
- Miyao, G., Brenna Aegerter, Karen Klonsky, Don Stewart, 2014a. Sample Costs to Produce Processing Tomatoes Furrow Irrigated In the Sacramento Valley & northern Delta University of California-Cooperative Extension <u>http://coststudies.ucdavis.edu/current/</u>.
- Miyao, G., Brenna Aegerter, Karen Klonsky, Don Stewart, 2014b. Sample Costs to Produce Processing Tomatoes Sub-surface, Drip Irrigated In the Sacramento Valley & northern Delta University of California-Cooperative Extension <u>http://coststudies.ucdavis.edu/current/</u>.
- Miyao, G., Klonsky, K., Livingston, P., 2008. Sample Costs to Produce Processing Tomatoes Transplanted in the Sacramento Valley. University of California Cooperative Extension. Available at: coststudies. ucdavis. edu/files/tomatoessv1_.
- Morandé, J.A., Stockert, C.M., Liles, G.C., Williams, J.N., Smart, D.R., Viers, J.H., 2017. From berries to blocks: carbon stock quantification of a California vineyard. Carbon balance and management 12, 5.
- Mosier, A.R., Halvorson, A.D., Reule, C.A., Liu, X.J., 2006. Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. J Environ Qual 35, 1584-1598.
- Mulvaney, R.L., Khan, S.A., Mulvaney, C.S., 1997. Nitrogen fertilizers promote denitrification. Biol Fert Soils 24, 211-220.
- Munkholm, L.J., Heck, R.J., Deen, B., Zidar, T., 2016. Relationship between soil aggregate strength, shape and porosity for soils under different long-term management. Geoderma 268, 52-59.

- Munoz, F., Mylavarapu, R., Hutchinson, C., 2005. Environmentally responsible potato production systems: a review. J Plant Nutr 28, 1287-1309.
- Murray, B.C., Baker, J.S., 2011. An output-based intensity approach for crediting greenhouse gas mitigation in agriculture: Explanation and policy implications. Greenhouse Gas Measurement and Management 1, 27-36.
- Myrold, D.D., Tiedje, J.M., 1985. Establishment of denitrification capacity in soil: effects of carbon, nitrate and moisture. Soil Biology and Biochemistry 17, 819-822.
- Nägele, W., Conrad, R., 1990. Influence of soil pH on the nitrate-reducing microbial populations and their potential to reduce nitrate to NO and N2O. Fems Microbiol Lett 74, 49-57.
- NASS, N.A.S.S., 2011. California Agricultural Statistics 2010 Crop Year. Washington, DC: National Agricultural Statistics Service, USDA. Available from http://www.nass.usda.gov/ca.

NASS, N.A.S.S., 2015. 2015 California Walnut Acreage Report.

- National Agricultural Statistics Service, 2014. Vegetables: Final Estimates 2008-2012. US Department of Agriculture: Washington, DC., 12-29.
- Nelson, D.W., Bremner, J.M., 1970. Role of soil minerals and metallic cations in nitrite decomposition and chemodenitrification in soils. Soil Biology and Biochemistry 2, 1-8.
- Nelson, S.D., Terry, R.E., 1996. THE EFFECTS OF SOIL PHYSICAL PROPERTIES AND IRRIGATION METHOD ON DENITRIFICATION. Soil Sci 161, 242-249.
- Nömmik, H., 1956. Investigations on denitrification in soil. Acta Agriculturae Scandinavica 6, 195-228.
- Norman, R., Kurtz, L., Stevenson, F., 1987. Solubilization of soil organic matter by liquid anhydrous ammonia. Soil Sci Soc Am J 51, 809-812.

- Oenema, O., Velthof, G., Kuikman, P., 2001. Technical and policy aspects of strategies to decrease greenhouse gas emissions from agriculture. Nutr Cycl Agroecosys 60, 301-315.
- Ogle, S.M., Breidt, F.J., Paustian, K., 2005. Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. Biogeochemistry 72, 87-121.
- Ogle, S.M., Paustian, K., 2005. Soil organic carbon as an indicator of environmental quality at the national scale: Inventory monitoring methods and policy relevance. Can J Soil Sci 85, 531-540.
- Ogle, S.M., Swan, A., Paustian, K., 2012. No-till management impacts on crop productivity, carbon input and soil carbon sequestration. Agriculture, Ecosystems & Environment 149, 37-49.
- Ottow, J., Burth-Gebauer, I., El Demerdash, M., 1985. Influence of pH and partial oxygen pressure on the N2O-N to N2 ratio of denitrification. Denitrification in the Nitrogen Cycle. Springer, pp. 101-120.
- Owen, J.J., Kebreab, E., Silver, W., 2014. Greenhouse Gas Mitigation Opportunities in California Agriculture.
- Park, S., Bae, W., 2009. Modeling kinetics of ammonium oxidation and nitrite oxidation under simultaneous inhibition by free ammonia and free nitrous acid. Process Biochem 44, 631-640.
- Parton, W., Holland, E., Del Grosso, S., Hartman, M., Martin, R., Mosier, A., Ojima, D., Schimel, D., 2001. Generalized model for NO x and N2O emissions from soils. Journal of Geophysical Research: Atmospheres 106, 17403-17419.
- Parton, W.J., Hartman, M., Ojima, D., Schimel, D., 1998. DAYCENT and its land surface submodel: description and testing. Global and planetary Change 19, 35-48.

- Parton, W.J., Schimel, D.S., Cole, C., Ojima, D., 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. Soil Sci Soc Am J 51, 1173-1179.
- Paustian, K., Andrén, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tiessen, H., Noordwijk, M.v.,Woomer, P.L., 1997. Agricultural soils as a sink to mitigate CO2 emissions. Soil Use Manage 13, 230-244.
- Paustian, K., Schuler, J., Killian, K., Chambers, A., Del Grosso, S., Easter, M., Alvaro-Fuentes, J., Gurung, R., Johnson, G., Merwin, M., 2012. COMET2. 0-Decision Support System for Agricultural Greenhouse Gas Accounting.
- Peacock, B., Williams, L.E., Christensen, L.P., 2000. Water management and irrigation scheduling. Raisin Production in California. University of California Division of Agricultural and Natural Resources, 127-133.
- Peacock, W.L., Christensen, L., Hirschfelt, D., 1998. Best Management Practices for Nitrogen Fertilization of Grapevines. Tulare County UCCE, Pub NG4-96.
- Peacock, W.L., Christensen, L.P., Hirschfelt, D.J., 1991. Influence of timing of nitrogen fertilizer application on grapevines in the San Joaquin Valley. American Journal of Enology and Viticulture 42, 322-326.
- Pereira, E.I.P., 2014. Can biochar mitigate nitrogen losses in organic farming systems.
- Pereira, E.I.P., Suddick, E.C., Six, J., 2016. Carbon Abatement and Emissions Associated with the Gasification of Walnut Shells for Bioenergy and Biochar Production. PloS one 11, e0150837.
- Phillips, R.L., 2008. Denitrification in cropping systems at sub-zero soil temperatures. A review. Agron Sustain Dev 28, 87-93.
- Picardal, F., 2012. Abiotic and Microbial Interactions during Anaerobic Transformations of Fe (II) and. Frontiers in microbiology 3.

- Powell, S.J., Prosser, J.I., 1991. Protection of Nitrosomonas-Europaea colonizing clay-minerals from inhibition by nitrapyrin. J Gen Microbiol 137, 1923-1929.
- Qiao, Y., Miao, S., Han, X., You, M., Zhu, X., Horwath, W.R., 2014. The effect of fertilizer practices on N balance and global warming potential of maize–soybean–wheat rotations in Northeastern China. Field Crop Res 161, 98-106.
- Rasmussen, P., Parton, W., 1994. Long-term effects of residue management in wheat-fallow: I. Inputs, yield, and soil organic matter. Soil Sci Soc Am J 58, 523-530.
- Rasmussen, P., Rohde, C., 1988. Long-term tillage and nitrogen fertilization effects on organic nitrogen and carbon in a semiarid soil. Soil Sci Soc Am J 52, 1114-1117.
- Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. Science 289, 1922-1925.
- Rochette, P., 2008. No-till only increases N 2 O emissions in poorly-aerated soils. Soil and Tillage Research 101, 97-100.
- Rosenstock, T., Liptzin, D., Six, J., Tomich, T., 2013. Nitrogen fertilizer use in California: Assessing the data, trends and a way forward. California agriculture 67, 68-79.
- Rudaz, A.O., Davidson, E.A., Firestone, M.K., 1991. Sources of Nitrous-Oxide Production Following Wetting of Dry Soil. Fems Microbiol Ecol 85, 117-124.
- Ruser, R., Flessa, H., Russow, R., Schmidt, G., Buegger, F., Munch, J.C., 2006. Emission of N(2)O, N(2) and CO(2) from soil fertilized with nitrate: Effect of compaction, soil moisture and rewetting. Soil Biol Biochem 38, 263-274.

- Sainju, U.M., Schomberg, H.H., Singh, B.P., Whitehead, W.F., Tillman, P.G., Lachnicht-Weyers,
 S.L., 2007. Cover crop effect on soil carbon fractions under conservation tillage cotton. Soil and Tillage Research 96, 205-218.
- Schellenberg, D.L., Alsina, M.M., Muhammad, S., Stockert, C.M., Wolff, M.W., Sanden, B.L., Brown, P.H., Smart, D.R., 2012. Yield-scaled global warming potential from N2O emissions and CH4 oxidation for almond (Prunus dulcis) irrigated with nitrogen fertilizers on arid land. Agr Ecosyst Environ 155, 7-15.
- Schipanski, M.E., Barbercheck, M., Douglas, M.R., Finney, D.M., Haider, K., Kaye, J.P., Kemanian, A.R., Mortensen, D.A., Ryan, M.R., Tooker, J., 2014. A framework for evaluating ecosystem services provided by cover crops in agroecosystems. Agr Syst 125, 12-22.
- Schlesinger, W.H., 2013. An estimate of the global sink for nitrous oxide in soils. Global Change Biol 19, 2929-2931.
- Seiter, S., Horwath, W.R., 2004. Strategies for managing soil organic matter to supply plant nutrients. Soil organic matter in sustainable agriculture. CRC Press. Boca Raton, FL, 269-294.
- Serrano, L., Carbonell, X., Save, R., Marfà, O., Peñuelas, J., 1992. Effects of irrigation regimes on the yield and water use of strawberry. Irrigation Sci 13, 45-48.
- Shannon, M., Grieve, C., 1998. Tolerance of vegetable crops to salinity. Sci Hortic-Amsterdam 78, 5-38.
- Shaviv, A., 2001. Advances in controlled-release fertilizers. Advances in agronomy 71, 1-49.
- Shcherbak, J., 2014. Global Meta analysis of the Nonlinear Response of Soil Nitrous Oxide Emissions to Fertilizer Nitrogen. P Natl Acad Sci USA.
- Shipley, P.R., Messinger, J., Decker, A., 1992. Conserving residual corn fertilizer nitrogen with winter cover crops. Agron J 84, 869-876.

- Six, J., Ogle, S.M., Conant, R.T., Mosier, A.R., Paustian, K., 2004. The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. Global Change Biol 10, 155-160.
- Smart, D.R., Alsina, M.M., Wolff, M.W., Matiasek, M.G., Schellenberg, D.L., Edstrom, J.P., Brown, P.H., Scow, K.M., 2011. N2O emissions and water management in California perennial crops. Understanding Greenhouse Gas Emissions from Agricultural Management, American Chemical Society, Baltimore MD USA, 227-255.
- Smart, D.R., Suddick, E., Pritchard, T., 2006. Control of greenhouse gas emissions from California vineyards by soil carbon and water and its policy implications. final report, Kearney Foundation of Soil Science.
- Smid, A., Beauchamp, E., 1976. Effects of temperature and organic matter on denitrification in soil. Can J Soil Sci 56, 385-391.
- Smith, J., Wagner-Riddle, C., Dunfield, K., 2010. Season and management related changes in the diversity of nitrifying and denitrifying bacteria over winter and spring. Appl Soil Ecol 44, 138-146.
- Smith, K.A., McTaggart, I.P., Dobbie, K.E., Conen, F., 1998. Emissions of N2O from Scottish agricultural soils, as a function of fertilizer N. Nutr Cycl Agroecosys 52, 123-130.
- Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara,
 F., Rice, C., 2008. Greenhouse gas mitigation in agriculture. Philosophical Transactions of the
 Royal Society of London B: Biological Sciences 363, 789-813.
- Smith, R., Brennan, E., Bolda, M., 2016. Managing Cover Crops on Strawberry Furrow Bottoms. SAT 19, 18.

- Smith, R., Cahn, M., Daugovish, O., Koike, S., 2011. Leaf lettuce production in California. UCANR Publications.
- Smukler, S., O'Geen, A., Jackson, L., 2012. Assessment of best management practices for nutrient cycling: A case study on an organic farm in a Mediterranean-type climate. Journal of Soil and Water Conservation 67, 16-31.
- Snapp, S., Swinton, S., Labarta, R., Mutch, D., Black, J., Leep, R., Nyiraneza, J., O'Neil, K., 2005. Evaluating cover crops for benefits, costs and performance within cropping system niches. Agron J 97, 322-332.
- Sørensen, J., Thorling, L., 1991. Stimulation by lepidocrocite (7-FeOOH) of Fe (II)-dependent nitrite reduction. Geochimica et Cosmochimica Acta 55, 1289-1294.
- Stark, J.M., Firestone, M.K., 1995. Mechanisms for Soil-Moisture Effects on Activity of Nitrifying Bacteria. Appl Environ Microb 61, 218-221.
- Starrs, P.F., Goin, P., 2010. Field Guide to California Agriculture. University of California Press.
- Steenwerth, K., Belina, K., 2008. Cover crops and cultivation: Impacts on soil N dynamics and microbiological function in a Mediterranean vineyard agroecosystem. Appl Soil Ecol 40, 370-380.
- Steenwerth, K.L., McElrone, A.J., Calderón-Orellana, A., Hanifin, R.C., Storm, C., Collatz, W., Manuck, C., 2013. Cover crops and tillage in a mature Merlot vineyard show few effects on grapevines. American Journal of Enology and Viticulture, ajev. 2013.12119.
- Steenwerth, K.L., Orellana-Calderón, A., Hanifin, R.C., Storm, C., McElrone, A.J., 2016. Effects of Various Vineyard Floor Management Techniques on Weed Community Shifts and Grapevine Water Relations. American Journal of Enology and Viticulture, ajev. 2015.15050.

- Stehfest, E., Bouwman, L., 2006. N₂O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutr Cycl Agroecosys 74, 207-228.
- Stein, L.Y., 2011. Surveying N2o-Producing Pathways in Bacteria. Methods in Enzymology: Research on Nitrification and Related Processes, Vol 486, Part A 486, 131-152.
- Suddick, E., Steenwerth, K., Garland, G., Smart, D., Six, J., 2011. Discerning agricultural management effects on nitrous oxide emissions from conventional and alternative cropping systems: a California case study. Understanding greenhouse gas emissions from agricultural management 1072, 203-226.
- Suddick, E.C., Six, J., 2013. An estimation of annual nitrous oxide emissions and soil quality following the amendment of high temperature walnut shell biochar and compost to a small scale vegetable crop rotation. Science of the Total Environment 465, 298-307.
- Sumner, D., 2014. Greenhouse Gas Mitigation Opportunities in California Agriculture.
- Sutton, K.F., Thomas Lanini, W., Mitchell, J.P., Miyao, E.M., Shrestha, A., 2006. Weed Control, Yield, and Quality of Processing Tomato Production under Different Irrigation, Tillage, and Herbicide Systems 1. Weed Technol 20, 831-838.
- Swan, A., S.A. Williams, K. Brown, A. Chambers, Creque, J., Wick, J., Paustian, K., 2015. COMETPlanner. Carbon and greenhouse gas evaluation for NRCS conservation practice planning. A companion report to <u>http://www.comet-planner.com/</u>. <u>http://cometplanner.nrel.colostate.edu/</u> COMET-Planner_Report_Final.pdf.
- Syswerda, S., Corbin, A., Mokma, D., Kravchenko, A., Robertson, G., 2011. Agricultural management and soil carbon storage in surface vs. deep layers. Soil Sci Soc Am J 75, 92-101.

- Szukics, U., Abell, G.C., Hödl, V., Mitter, B., Sessitsch, A., Hackl, E., Zechmeister-Boltenstern,S., 2010. Nitrifiers and denitrifiers respond rapidly to changed moisture and increasing temperature in a pristine forest soil. Fems Microbiol Ecol 72, 395-406.
- Tenuta, M., Beauchamp, E.G., 2003. Nitrous oxide production from granular nitrogen fertilizers applied to a silt loam. Can J Soil Sci 83, 521-532.
- Thauer, R.K., Jungermann, K., Decker, K., 1977. Energy conservation in chemotropic anaerobic bacteria. Bacteriol Rev 41, 100-180.
- Thorn, K., Mikita, M., 2000. Nitrite fixation by humic substances nitrogen-15 nuclear magnetic resonance evidence for potential intermediates in chemodenitrification. Soil Sci Soc Am J 64, 568-582.
- Tian, G., Granato, T.C., Cox, A.E., Pietz, R.I., Carlson, C.R., Abedin, Z., 2009. Soil Carbon Sequestration Resulting from Long-Term Application of Biosolids for Land Reclamation. J Environ Qual 38, 61-74.
- Tolomeo, V., Krug, K., DeWalt, D., 2012. California Agricultural Statistics, 2012 Crop Year. United States Department of Agriculture, National Agricultural Statistics Service, Pacific Regional Office--California, Sacramento, California.
- Tourte, L., Smith, R., 2010. Sample Production Costs for Wrapped Iceberg Lettuce Sprinkler Irrigated – 40-inch Beds. University of California Cooperative Extension. Available at: coststudies. ucdavis. edu/files/tomatoessv1_.
- Turini, T., Cahn, M., Cantwell, M., Jackson, L., Koike, S., Natwick, E., Smith, R., Subbarao, K., Takele, E., 2011. Iceberg lettuce production in California. University of California Division of Agriculture and Natural Resources (Ed.) 7215.

- UCAIC, 2009. The Measure of California Agriculture. University of California Agricultural Issues Center.
- Unger, P.W., 1999. Conversion of conservation reserve program (CRP) grassland for dryland crops in a semiarid region. Agron J 91, 753-760.

University of California Cooperative Extension, 2011. Sample costs to produce strawberries.

- University of California Cooperative Extension, 2012. Sample costs to produce fresh market broccoli
- USDA, 2014a. Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory.

USDA, 2014b. Tomatoes. http://www.ers.usda.gov/topics/crops/vegetables-pulses/tomatoes.aspx.

- USEPA, 2016. Inventory of U.S. greenhouse gas emissions and sinks: 1990-2014. U.S. Environmental Protection Agency, 1200 Pennsylvania Ave. N.W. Washington, DC 20460.
 EPA 430-R-10-006. <u>http://epa.gov/climatechange/emissions/usinventoryreport.html</u> (verified May 27, 2016).
- Van Cleemput, O., 1998a. Subsoils: chemo- and biological denitrification, N2O and N-2 emissions. Nutr Cycl Agroecosys 52, 187-194.
- Van Cleemput, O., 1998b. Subsoils: chemo-and biological denitrification, N2O and N2 emissions. Nutrient Cycling in Agroecosystems 52, 187-194.
- Van Cleemput, O., Baert, L., 1984. Nitrite a Key Compound in N-Loss Processes under Acid Conditions. Plant and Soil 76, 233-241.
- Van Cleemput, O., Samater, A.H., 1996. Nitrite in soils: Accumulation and role in the formation of gaseous N compounds. Fert Res 45, 81-89.

- Van Groenigen, J.W., Velthof, G.L., Oenema, O., Van Groenigen, K.J., Van Kessel, C., 2010. Towards an agronomic assessment of N2O emissions: a case study for arable crops. European Journal of Soil Science 61, 903-913.
- Varco, J., Frye, W., Smith, M., Grove, J., 1987. Legume N transformation and recovery by corn as influenced by tillage. The Role of Legumes in Conservation Tillage Systems. Soil Conservation Society of America, University of Georgia, Athens.
- Varvel, G.E., 2006. Soil organic carbon changes in diversified rotations of the western Corn Belt. Soil Sci Soc Am J 70, 426-433.
- Veenstra, J.J., Horwath, W.R., Mitchell, J., 2007. Tillage and cover cropping effects on aggregateprotected carbon in cotton and tomato. Soil Sci Soc Am J 71, 362-371.
- Venterea, R.T., 2007. Nitrite-driven nitrous oxide production under aerobic soil conditions: kinetics and biochemical controls. Global Change Biol 13, 1798-1809.
- Venterea, R.T., Clough, T.J., Coulter, J.A., Breuillin-Sessoms, F., 2015. Ammonium sorption and ammonia inhibition of nitrite-oxidizing bacteria explain contrasting soil N2O production. Sci Rep-Uk 5.
- Venterea, R.T., Coulter, J.A., 2015. Split application of urea does not decrease and may increase N2O emissions in rainfed Corn. Agronomy, Soils & Environmental Quality 107, 337-348.
- Venterea, R.T., Coulter, J.A., Dolan, M.S., 2016. Evaluation of intensive "4R" strategies for decreasing nitrous oxide emissions and nitrogen surplus in rainfed corn. J Environ Qual 45, 1186-1195.
- Venterea, R.T., Dolan, M.S., Ochsner, T.E., 2010. Urea decreases nitrous oxide emissions compared with anhydrous ammonia in a Minnesota corn cropping system. Soil Sci Soc Am J 74, 407-418.

- Venterea, R.T., Maharjan, B., Dolan, M.S., 2011. Fertilizer Source and Tillage Effects on Yield-Scaled Nitrous Oxide Emissions in a Corn Cropping System. J Environ Qual 40, 1521-1531.
- Venterea, R.T., Rolston, D.E., 2000. Nitric and nitrous oxide emissions following fertilizer application to agricultural soil: Biotic and abiotic mechanisms and kinetics. J Geophys Res-Atmos 105, 15117-15129.
- Verhoeven, E., Six, J., 2014. Biochar does not mitigate field-scale N 2 O emissions in a Northern California vineyard: an assessment across two years. Agriculture, Ecosystems & Environment 191, 27-38.
- Watson, C., Atkinson, D., Gosling, P., Jackson, L., Rayns, F., 2002. Managing soil fertility in organic farming systems. Soil Use Manage 18, 239-247.
- Weier, K.L., Doran, J.W., Power, J.F., Walters, D.T., 1993. Denitrification and the dinitrogen nitrous oxide ratio as affected by soil-water, available carbon, and nitrate. Soil Sci Soc Am J 57, 7.
- West, T.O., Marland, G., 2002. A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: comparing tillage practices in the United States. Agriculture, Ecosystems & Environment 91, 217-232.
- West, T.O., Six, J., 2007. Considering the influence of sequestration duration and carbon saturation on estimates of soil carbon capacity. Climatic Change 80, 25-41.
- Williams, J.N., Hollander, A.D., O'Geen, A.T., Thrupp, L.A., Hanifin, R., Steenwerth, K., McGourty, G., Jackson, L.E., 2011. Assessment of carbon in woody plants and soil across a vineyard-woodland landscape. Carbon balance and management 6, 1.
- Williams, L., 1987. Growth of Thompson seedless grapevines. II: Nitrogen distribution. J Am Soc Hortic Sci 112, 330-333.

- Williams, L.E. 2000. Bud development and fruitfulness of grapevines, *In* Raisin Production Manual. L.P. Christensen (Ed.), pp. 24-29. University of California Division of Agriculture and Natural Resources, Oakland.
- Winkler, A., 1974. General viticulture. Univ of California Press.
- Wolfe, D., 1997. Soil compaction: crop response and remediation. Report.
- Wolff, M.W., 2015. Net Global Warming Potential and Nitrogen Fertigation in Orchards and Vineyards: Opportunities for Mitigation. UNIVERSITY OF CALIFORNIA, DAVIS.
- Wolff, M.W., Hopmans, J.W., Stockert, C.M., Burger, M., Sanden, B.L., Smart, D.R., 2016.Effects of drip fertigation frequency and N-source on soil N 2 O production in almonds.Agriculture, Ecosystems & Environment.
- Wrage, N., Velthof, G.L., van Beusichem, M.L., Oenema, O., 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol Biochem 33, 1723-1732.
- Wunderlich, L., Klonsky, K., Stewart, D., 2015. Sample costs to establish a vineyard and produce wine grapes. University of California-Cooperative Extension.
- Xu, J.-Z., Peng, S.-Z., Hou, H.-J., Yang, S.-H., Luo, Y.-F., Wang, W.-G., 2013. Gaseous losses of nitrogen by ammonia volatilization and nitrous oxide emissions from rice paddies with different irrigation management. Irrigation Sci 31, 983-994.
- Yu, O.T., Greenhut, R.F., O'Geen, A.T., Mackey, B., Horwath, W.R., Steenwerth, K.L., 2017.Precipitation Events and Management Practices Affect Greenhouse Gas Emissions from Vineyards in a Mediterranean Climate. Soil Sci Soc Am J 81, 138-152.
- Zebarth, B., Rochette, P., Burton, D., Price, M., 2008a. Effect of fertilizer nitrogen management on N2O emissions in commercial corn fields. Can J Soil Sci 88, 189-195.

- Zebarth, B.J., Rochette, P., Burton, D.L., 2008b. N2O emissions from spring barley production as influenced by fertilizer nitrogen rate. Can J Soil Sci 88, 197-205.
- Zhang, W., Xu, M., Wang, B., Wang, X., 2009. Soil organic carbon, total nitrogen and grain yields under long-term fertilizations in the upland red soil of southern China. Nutr Cycl Agroecosys 84, 59-69.
- Zhu, X., Burger, M., Doane, T.A., Horwath, W.R., 2013a. Ammonia oxidation pathways and nitrifier denitrification are significant sources of N₂O and NO under low oxygen availability.
 P Natl Acad Sci USA 110, 6328-6333.
- Zhu, X., Burger, M., Waterhouse, H., Horwath, W.R., 2014. The Effect of Ammonical N Fertilizer Concentration Soil O2 Consumption and N2O Production Pathways. ASA, CSSA and SSSA International Annual Meetings
- Zhu, X., Silva, L.C.R., Doane, T.A., Wu, N., Horwath, W.R., 2013b. Quantifying the effects of green waste compost application, water content and nitrogen fertilization on nitrous oxide emissions in 10 agricultural soils. J Environ Qual 42, 912-918.
- Zhu-Barker, X., Cavazos, A.R., Ostrom, N.E., Horwath, W.R., Glass, J.B., 2015a. The importance of abiotic reactions for nitrous oxide production. Biogeochemistry 126, 251-267.
- Zhu-Barker, X., Doane, T.A., Horwath, W.R., 2015b. Role of green waste compost in the production of N 2 O from agricultural soils. Soil Biology and Biochemistry.
- Zhu-Barker, X., Horwath, W.R., Burger, M., 2015c. Knife-injected anhydrous ammonia increases yield-scaled N 2 O emissions compared to broadcast or band-applied ammonium sulfate in wheat. Agriculture, Ecosystems & Environment 212, 148-157.

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

3

4 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 N₂O 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).

12 Factors that influence N₂O production through ammonia oxidation pathways include soil 13 nitrifier activity, ammonium concentration and sorption, texture, pH, temperature, moisture, 14 oxygen availability, and bioavailability of metals (e.g. iron and manganese). Examples of how 15 these factors influence N_2O production are presented here. Nitrous oxide production has been 16 reported to be greater from clay soils than from loams or sandy soils (Zhu et al., 2013a). This is 17 likely a result of clay surfaces protecting nitrifier populations from the effects of acidifying 18 hydrogen ions (H⁺) produced during NH₃ oxidation (Powell and Prosser, 1991). Increased clay 19 content can lead to higher nitrifier populations and associated higher nitrification rates (Fortuna et 20 al., 2012), as well as more anaerobic microsites that enable denitrification. Short-term increases in 21 soil pH and associated higher free ammonia limit nitrite oxidation by *Nitrobacter*, resulting in the 22 accumulation of nitrite (Hawkins et al., 2010; Venterea et al., 2015; Breuillin-Sessoms et al., 2017), which can also lead to N₂O production (Venterea and Rolston, 2000; Venterea, 2007). Generally, 23

24 increased soil temperatures are associated with increases in ammonia oxidizer populations 25 (Avrhami and Conrad, 2003; Avrahami and Bohannan, 2007; Szukics et al., 2010) and activities 26 (Avrahami *et al.*, 2003), resulting in increases in N_2O production through ammonia oxidation 27 pathways. However, one study showed that when soil temperature increased from 25 to 37°C, 28 potential nitrification rate decreased but the rate of N₂O release increased monotonically 29 (Avrahami et al., 2003). This indicates that N₂O production through ammonia oxidation pathways 30 can be independent from nitrification under certain conditions. It has been reported that oxygen 31 availability in soil is also an important factor that influences N_2O production through ammonia 32 oxidation pathways (Zhu *et al.*, 2013a). When oxygen concentrations decreased from 21% to 0.5%, 33 N_2O production via ammonia oxidation pathways increased. Soil moisture influences N_2O 34 production by controlling not only the diffusion of oxygen, but also substrate availability (Stark 35 and Firestone, 1995) and microbial activity (Avrahami and Bohannan, 2007). The biological 36 availability of iron is a key factor that regulates iron-dependent enzymes participating in N₂O 37 production through ammonia oxidation pathways, such as ammonia monooxygenase, 38 hydroxylamine oxidoreductase, and nitrite oxidoreductase (Meiklejohn, 1952; Godfrey and Glass, 39 2011; Stein, 2011; Glass and Orphan, 2012).

40

41 1.2 Heterotrophic denitrification

Heterotrophic denitrification is a stepwise reduction of nitrate or nitrite with dinitrogen (N₂)
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
N₂O, can be released into the atmosphere under certain environmental conditions (Firestone *et al.*,

47 1979; Firestone *et al.*, 1980a; Firestone and Davidson, 1989a; Weier *et al.*, 1993; Mulvaney *et al.*,
48 1997; Van Cleemput, 1998a; Gillam *et al.*, 2008; Phillips, 2008). Enzymes catalyzing these
49 reductions are nitrate reductase, nitrite reductase, nitric oxide reductase and N₂O reductase
50 (Hochstein and Tomlinson, 1988).

51 Soil water content is one of the most important factors influencing the denitrification 52 process, as it interacts with pore size distribution to control oxygen diffusion, soil NO_3^{-1} 53 concentration and C availability (Smid and Beauchamp, 1976; Firestone et al., 1979; Firestone 54 and Davidson, 1989a; Weier et al., 1993). Denitrification-derived N₂O has been shown to increase 55 as soil water content increases, especially when soil water content is higher than 70% of water-56 filled pore space (WFPS) (Bateman and Baggs, 2005). The effects of oxygen concentration on 57 denitrification are mainly due to the effects on the activities of enzymes associated with 58 denitrification. For example, nitrate reductase activity can be completely blocked at an oxygen 59 concentration greater than 0.25%, compared with 0.13% and 0.02% for nitrite and N₂O reductases, 60 respectively (Bonin *et al.*, 1989). Increases in nitrate concentration do not always increase the 61 overall rate of denitrification, but can result in a higher proportion of N_2O as the product of 62 denitrification (Cooper and Smith, 1963; Firestone et al., 1979; Knowles, 1982; Weier et al., 1993). 63 This is likely because nitrate is a preferred electron acceptor over N_2O (Thauer *et al.*, 1977; 64 Firestone et al., 1980b; Firestone and Davidson, 1989b). Content of soil organic carbon (SOC), as 65 energy source and electron donor for denitrifiers, is highly correlated with denitrification capacity 66 (Myrold and Tiedje, 1985; Weier et al., 1993; Gillam et al., 2008). Both N₂O and N₂/N₂O ratio 67 increase as SOC, especially decomposable carbon (C), increases (Burford and Bremner, 1975). 68 Temperature is also an important factor controlling emissions of N₂O from denitrification.

70	an increase in soil temperature to 67°C. Nitrous oxide also becomes an increasingly smaller
71	component of the gaseous nitrogen (N) evolved and persists for shorter times with increasing
72	temperature (Nömmik, 1956; Smid and Beauchamp, 1976; Keeney et al., 1979).

73

74 1.3 Abiotic chemodenitrification

75 Chemodenitrification is the chemical decomposition of N intermediates in the ammonia 76 oxidation (e.g. nitrite) or denitrification (e.g. nitrate, nitrite, or NO or N₂O) pathways, via coupled 77 reactions involving oxidation of Fe(II) or other electron donors such as humic acids (Chao and 78 Kroontje, 1966; Buresh and Moraghan, 1976; Chalk, 1983; Sørensen and Thorling, 1991; Van 79 Cleemput, 1998a; Thorn and Mikita, 2000; Picardal, 2012). Soil N₂O emissions from 80 chemodenitrification are stimulated by decreased oxygen (i.e. increased denitrification), increased 81 Fe(II) and the presence of organic matter (Nelson and Bremner, 1970; Van Cleemput and Baert, 82 1984). Therefore, N₂O production from chemodenitrification is assumed to mostly exist in anoxic 83 conditions with a high supply of organic substrates, such as in wetlands and waterlogged soils. In 84 these environments, persistence of significant concentrations of Fe(II) and bacterial denitrification 85 can lead to the accumulation of reactive NO₂⁻ and NO (Van Cleemput, 1998b), and subsequent 86 N₂O production (Venterea and Rolston, 2000; Venterea, 2007).

87

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

89

90 The intent of Appendix 2 is to provide updated information on the role of agricultural practices in 91 soil organic carbon (SOC) dynamics. It also serves as a point of comparison for the specialty crop 92 systems examined in the main body of this report. 93

94 2.1 Fertilization

95 Soil fertility affects the production of crop biomass, as well as the quality and quantity of 96 crop residues that are returned to the soil (Rasmussen and Rohde, 1988; Lal et al., 1998; Follett, 97 2001; Christopher and Lal, 2007; Halvorson et al., 2009; Halvorson and Jantalia, 2011; Jantalia 98 and Halvorson, 2011; Halvorson and Stewart, 2015). Halvorson et al. (1999) observed that in a 99 barley-corn-winter wheat-oat-pea-hay rotation system, the amount of crop residue (aboveground 100 residue plus belowground residue) returned to the soil increased with increasing nitrogen (N) rate 101 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 102 103 the unfertilized treatment, respectively. In comparison, Kong et al. (2005) reported that between 104 1993 and 2003 in California, SOC was sequestered at the rate of 40 kg carbon (C) ha⁻¹yr⁻¹ in a fertilized wheat-fallow system while SOC decreased at the rate of 190 kg C ha⁻¹yr⁻¹ in a unfertilized 105 106 wheat-fallow system. In a review of SOC in U.S. cropping systems, Lal et al. (1998) concluded that 50-150 kg C ha⁻¹yr⁻¹ SOC can be sequestered by fertility management practices. 107

108 Increasing SOC sequestration under N fertilization relies on crop residue C inputs, N 109 fertilization rate, initial soil C content, and the soil's capacity for retention. Increases in SOC do 110 not always correspond directly to increases in N fertilization if crop growth is optimized at a lower 111 rate (McCarty et al. 1997). Initial background SOC can also be sufficiently high to mask increases 112 in SOC, even in long-term studies that demonstrate consistent increases, in response to N 113 fertilization, in the crop biomass returned to the soil (Halvorson et al. (2002). Some studies also 114 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 115

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).

120

121 2.2 Tillage

122 Changes in tillage practices influence the total soil C stock and the vertical distribution of 123 C in the soil profile (Paustian et al., 1997). Long-term conventional tillage has resulted in the 124 reduction of soil C stocks by up to 30% compared to pre-tillage levels, yet a change in tillage from 125 conventional to long-term adoption of conservation, no-till or reduced tillage may reverse such 126 losses of soil C and enhance C sequestration (Paustian et al., 1997; Lal et al., 1998; West and 127 Marland, 2002; Lal et al., 2004; Ogle and Paustian, 2005; Álvaro-Fuentes et al., 2009), but the 128 magnitude of soil C accumulation after the adoption of no-till also depends on soil texture, climate, 129 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⁻¹yr⁻¹ was sequestered with no-till in an annual 130 cropping rotation system in North Dakota, compared with 25 kg C ha⁻¹yr⁻¹ with conservation tillage 131 and a loss of 141 kg C ha⁻¹yr⁻¹ with conventional tillage (Halvorson *et al.*, 2002). In some cases, 132 133 crop production and corresponding C inputs decrease under no till, but any potential reduction in 134 SOC stocks can be offset by a decrease in soil C decomposition rates, except in cases where C 135 inputs are reduced by 15% or more (Ogle et al., 2012). Still, efficacy of conservation or no till 136 practices has been questioned in recent studies (Baker et al., 2007; Christopher et al., 2009). No-137 till practices can lead to accumulation of soil C in the upper soil profile, but with little to no change 138 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.

144 As stated above, the response of SOC to tillage and associated benefits depends on the 145 duration of tillage adoption. In a California tomato-cotton rotation system, for example, Veenstra 146 et al. (2007) reported that compared to the initial SOC, the content of SOC in the top 30 cm of soil 147 had decreased five years after the adoption of conservation tillage, whereas a net increase was 148 found ten years after adoption (Mitchell et al., 2015b). This is largely due to the dry climate in 149 California, which delays decomposition of crop residues and consequently delays accumulation of 150 SOC. No difference in SOC content (0-30 cm depth) was found between the conventional and 151 conservation tillage ten years after the tillage adoption in this tomato-cotton rotation system 152 (Veenstra et al., 2007; Mitchell et al., 2015b). However, the conversion of conventional tillage to 153 no-till has led to a SOC sequestration rate of 0.3 Mg C ha⁻¹yr⁻¹ in annual crop systems under other 154 climates such as temperate climate (West and Marland, 2002; Lal, 2003). The adoption of 155 conservation tillage also influences yields and benefits soil and water conservation efforts (Unger, 156 1999; Baker et al., 2005; Mitchell et al., 2009; Mitchell and Miyao, 2012; Mitchell et al., 2015b). 157 For example, in a California cotton-tomato rotation system, Baker et al. (2005) reported that 158 compared to conventional tillage, the adoption of conservation tillage reduced cumulative dust 159 emissions by 67% throughout the two-year rotation. In the same rotation system, Mitchell et al. 160 (2015b) found that tomato yields increased under conservation tillage compared to conventional 161 tillage, whereas the opposite was found for cotton yields.
163 2.3 Irrigation

164 Irrigation is fundamental to maintaining or increasing crop yields in semiarid and arid lands 165 where plant growth is limited by available water. Under irrigated systems, wetting of dry soils 166 typically leads to short-term increases in CO_2 emissions as the labile soil C pool is mineralized 167 (Kieft et al., 1987; Fierer and Schimel, 2002; Ruser et al., 2006). However, SOC in these systems 168 can be maintained or increased due to C inputs as a result of increased plant growth and therefore 169 increased inputs from crop residues (both above- and below-ground) and increased aggregate 170 formation (Follett, 2001; De Gryze et al., 2005; Kong et al., 2005; Gillabel et al., 2007; Collins et 171 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⁻¹yr⁻¹. However, Follett (2001) viewed these estimates as 172 173 too conservative. The potential for C sequestration in intensively managed irrigated cropping 174 systems can be increased considerably by the use of improved fertilization practices, tillage, and 175 irrigation management. After considering the application of improved technology, Eve et al. 176 (2002) documented that irrigation results in SOC sequestration ranging from 0.25 to 0.52 Mg C ha⁻¹yr⁻¹ in the western U.S. In a model projection study conducted in a semiarid Mediterranean 177 178 agroecosystem, however, the shift from rain-fed conditions to irrigation resulted in an increase in 179 C inputs but a decrease in the SOC sequestered during the 2010-2100 period due to the interactions 180 of climate with system management that ultimately promoted greater C mineralization (Álvaro-181 Fuentes and Paustian, 2011). In California, almost all agricultural land is irrigated due to the 182 prevailing Mediterranean climate. However, information on how different irrigation management 183 practices affect soil C stock, especially the distribution of SOC in the soil profile, is still lacking.

185 2.4 Cover crops and organic amendments

186 The potential for cover crops and organic amendments to enhance SOC content, increase 187 crop productivity, sequester C, and reduce atmospheric CO₂ concentrations makes them appealing 188 for both climate change mitigation and land-use sustainability. Duval et al. (2016) observed that 189 the increase in the mean annual C input by cover crops and their plant residue into the soil 190 explained most of the variation in SOC in a soybean cropping system in Argentina. In a California 191 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 192 193 conservation and conventional tillage practices, respectively (Veenstra et al., 2007). After ten 194 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 195 196 under conservation and conventional tillage practices, respectively (Mitchell et al., 2015b). Across 197 U.S. cropland, adoption of cover crop practices has been projected to potentially convert an estimated 0.15-0.22 Mg C ha⁻¹yr-¹ atmospheric CO₂ to SOC (Swan et al., 2015; Chambers et al., 198 199 2016). This SOC sequestration rate is comparable to the C sequestration potential (0.1- 0.3 kg C 200 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).

214 Under California's Mediterranean climate, a 10-year long-term experiment in maize-215 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⁻¹ 216 217 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 218 inorganic system (0.56 vs. 0.04 Mg C ha⁻¹ yr⁻¹) (Kong et al., 2005). Application of compost (add 219 220 amount) over six years also led to increases in SOC content in organic vegetable systems in the 221 Central Coast of California (Brennan and Acosta-Martinez, 2017). In Spain, another 222 Mediterranean climate, Calleja-Cervantes et al. (2015) observed that the application of organic 223 amendments to a vineyard over 13 years increased SOC at least 35% more compared to the control 224 (no organic amendments added). GHG mitigation from increased SOC sequestration resulting 225 from organic amendments may be negated by GHGs emitted by fuel consumption caused by 226 transportation to the farm and application of organic amendments. GHG emissions from California 227 agriculture, as with many other regions, have varied responses to organic amendments depending 228 on organic amendment management, soil type and environmental conditions (Zhu et al., 2013b; 229 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 AcostaMartinez, 2017).

235

236 2.5 Cropping system conversion

237 Cropping system conversion practices, e.g. intensification, crop rotation diversification, 238 and conversion of annual to perennial crops, have a significant impact on SOC sequestration (De 239 Gryze et al., 2004; Eagle et al., 2011). The practices improve soil C storage through increases in 240 crop biomass returning to the soils (Kroodsma and Field, 2006; Eagle et al., 2011). Crop rotation 241 diversification often involves changing from a continuously cropped cereal or simple rotation to 242 multiple crops over multiple years of a crop rotation. This change is expected to increase residue 243 biomass, residue quality (e.g. N content), and root exudates, and therefore soil C storage. Eagle et 244 al. (2011) reviewed studies on soil C sequestration for diversifying crop rotations and summarized 245 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 246 247 SOC content from two-year rotations (corn-soybean and sorghum-soybean) was found over 248 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 249 250 diverse crop rotations, respectively (Varvel, 2006). Increasing the number of crops per year can 251 produce higher quantity and quality residues that increase SOC sequestration (Halvorson et al., 252 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⁻¹yr⁻¹ in a 10year 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.

258 In U.S. cropping systems, the conversion of an annual to a perennial crop has been 259 suggested to sequester C in soil, with variable results depending on the crop type (Liebig et al., 260 2005; Grandy and Robertson, 2007). SOC sequestration was estimated to increase by 0.6 Mg C ha⁻¹ vr⁻¹ after the conversion of an annual to a perennial crop, while changing from cropland to 261 262 pasture land yielded a projected average net impact of 1.2 Mg C ha⁻¹ yr⁻¹ (Culman et al., 2014). 263 Using the CASA (Carnegie-Ames-Stanford Approach) model to estimate soil C sequestration 264 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 265 orchards can sequester 0.85 Mg C ha⁻¹ yr⁻¹. 266

267

268 Appendix 3. Calculations for Emission Factors

269

EF(SDI)= EF(furrow)* EF'(SDI)/EF'(furrow) 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) Equation 2

- 276 Where EF' (cover crop) and EF' (no cover crop) are calculated based on the N₂O fluxes reported
- by Kallenbach et al (2010), and EF (no cover crop) is averaged from Kennedy et al. (2013) and
- 278 Burger and Horwath (2012) (for furrow irrigation) or calculated from equation 1(for SDI).

280 Figures:

281



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.

285



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.

290





292 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





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.





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



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



309 Figure 8. Results of the DayCent model input parameterizations for lettuce. Slope of the regression

310 was 1.0, RMSE was 18.4, and the correlation coefficient was 0.73.

- 312 Tables
- 313

314 Table 1. Nitrous Oxide Emission Studies in California Tomato Cropland

315

Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha ⁻¹)	Crop residual N (kg N ha-1)	N ₂ O Emissions		Emission factors (% of applied N emitted as N ₂ O, uncorrected for background flux)		Emission factors (% of applied N emitted as N ₂ O, 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	Ν	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
City Iodin	SDI	Reduced	Ν	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
					0			1.00 ± 0.16^{b}		na	na	na	na	Field	Burger and Horwath, 2012
Yolo silt loam:				15-15-15;	75		na	1.23 ± 0.01^{b}		na	1.63 ± 0.01	na	0.31 ± 0.01	Field	Burger and Horwath, 2012
22% sand, 47%	Furrow	Standard	Ν	Ammonium	162		na	1.81 ± 0.18^{b}		na	1.11 ± 0.11	na	0.50 ± 0.07	Field	Burger and Horwath, 2012
silt, 31% clay				sulfate; UAN-32	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
	Furrow	Standard	Ν	Ammoniun nitrate	120	74	na	4.04 ^d		na	3.37 ^d	na	2.08 ^d	Field	Kallenbach et al., 2010
Reiff loam and	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
Yolo silt loam	SDI	Standard	Ν	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

316 317

*Emission factors were corrected based on considering either crop residue N input or N₂O emissions from zero synthetic N input treatment

318 ^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.

- 319 ^b Tomato growing season.
- 320 ° 120 kg N ha⁻¹ synthetic N plus 107 kg N ha⁻¹ biomass N from cover crop residue.
- 321 ^d Data based on only 13 hourly flux measurements.
- 322 na: No data available; SDI: subsurface drip irrigation.

323

324

326 Table 2. Nitrous Oxide Emission Studies in California Lettuce Cropland

Soil texture	Irrigatio n type	Tillage	Cover crop	N source	N input (kg N ha	Crop residual N (kg N ha		N ₂ O Emissi	ons	Emission fac applied N e N ₂ O, unco	ctors (% of emitted as rrected for	Emission fa applied N N ₂ O, cor	ctors (% of emitted as rected 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	Commerci al 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
					0		0.16±0.12	0.27 °	1.67	na	na	na	na	Greenhouse	Pereira, 2014
	SDI ^a	Standard a			56		$0.35 {\pm} 0.05$	0.58 °	1.67	0.63±0.08	1.04	0.34±0.12	0.55	Greenhouse	Pereira, 2014
Yolo silt clay loam:	SDI ^a	Standard ^a	N	Commerci	112		0.79±0.06	1.32 °	1.67	0.71±0.05	1.18	0.56±0.07	0.94	Greenhouse	Pereira, 2014
silt; 34% clay	SDI ^a	Standard a	IN	fertilizer	168		1.00±0.21	1.67 °	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 °	1.67	0.51±0.06	0.85	$0.44{\pm}0.08$	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
t					84	15	0.34±0.02	$0.64{\pm}0.05$	1.89	0.4	0.76±0.06	0.34	0.65 ± 0.03	Field	Burger and Horwath, 2012
Loam: 54% sand;	SDI	Standard	Ν	UAN-32	168	19.5	$0.50{\pm}0.04$	$0.91{\pm}0.14$	1.82	0.3	0.54 ± 0.08	0.27	$0.49{\pm}0.04$	Field	Burger and Horwath, 2012
29% silt; 17% clay					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

327

328 ^aSimulated practice

329 ^bValue averaged from the scaling factors obtained from Burger and Horwath 2012

330 ^cCalculated based on scaling factor

331 ^d Background corrected using N₂O emission background reported in Pereira, 2014

Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha ⁻¹)		N ₂ O Emission	s	Emission f applied N en	factors (% of hitted 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		
Loam: 48% sand		No till				$0.20{\pm}0.02^{b}$	5.04 ^c	28	0.38±0.04	9.69		
33% silt, 19% clay	Drip	Standard	Y	UAN-32	52 ^a	$0.14{\pm}0.04^{b}$	3.92 ^c	28	0.27±0.08	7.54	Field	Garland et al., 2011
Loom, 480/ cond			Y		52 ^a	0.14	3.92	28	0.27	7.54		
23% silt, 19% clay	Drip	Standard	Ν	UAN-32	5.4	0.3	0.56	1.87	5.56	10.4	Field	Garland et al., 2014
Sandy clay loam:					129 ^d	0.76	2	2.63	0.59	1.55		Verhoeven and Six,
50% sand, 27% silt, 23% clay	Drip	Standard	Ŷ	UAIN-32	275 ^e	0.37	1.6	4.32	0.13	0.58	Field	2014
			Ν			na	0.47			na		Steenworth at al
Loam	Drip	Standard	Y	na	na	na	0.57			na	Field	2008
			Y			na	0.69			na		2000
Loam: 33% sand.		Reduced	Y	Potassium	43.4 ^r	na	0.14±0.03			0.32 ± 0.07		
42% silt, 25% clay	Drip	Standard	Y	nitrate	42 ¹	na	0.17 ± 0.01			0.40 ± 0.02	Field	Wolff, 2015
		Standard	N		27.6 ^r	na	0.13±0.03			0.47±0.11		
Loam: 33% sand.		Reduced	Y	Potassium	38.8 ^g	na	0.15±0.04			0.39±0.10		
42% silt, 25% clay	Drip	Standard	Y	nitrate	44 ^g	na	0.20±0.04			0.45±0.09	Field	Wolff, 2015
-		Standard	N		38.6 ^g	na	0.19±0.04			0.49±0.10		
					0	na	0.03"			na		
unknown	Drip	Standard	unknown	unknown	6	na	0.05 ^h			0.96	Field	Smart et al., 2006
					45	na	0.09 ^h			0.2		

333 Table 3. Nitrous Oxide Emission Studies in California Vineyard

335 ^a5 kg N ha⁻¹ synthetic N plus 47 kg N ha⁻¹ organic N from cover crop residue

336 ^bValue calculated from berm (contributes 26%) and row emissions (contributes 74%)

337 °Value calculated based on scaling factor

338 ^d9.6 kg N ha⁻¹ synthetic N plus 119.6 kg N ha⁻¹ organic N from cover crop residue

339 °66.4 kg N ha⁻¹ synthetic N plus 208.8 kg N ha⁻¹ organic N from cover crop residue

- 340 f8.4 kg N ha-1 synthetic N plus organic N from alley crop residue
- 341 g16.8 kg N ha-1 synthetic N plus organic N from alley crop residue
- 342 ^hValue calculated from the fluxes reported in the literature and were not included in the statewide emission calculation

343 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 ₂ O Emissions		Emission factors (% of applied N emitted as N ₂ O, uncorrected for background flux)		Emission factors (% of applied N emitted as N ₂ O, 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%	Microsprinkler	No till		UAN: 4 times application	224	105		0.80±0.19		0.35±0.08			0.24±0.06		Schellenberg et al
sand, 17% silt, 19% clay	Microsprinkler	No till	Ν	CAN: 4 times application	224	102		0.53±0.11			0.23±0.05		0.16±0.03	Field	2012
	Drip	No till		UAN: 4 times application	336	105 ^a	0.781	1.17	1.5	0.23	0.35	0.18	0.27		
Sandy loam: 64% sand, 17% silt,	Drip	No till	Ν	UAN: 8 times application	336	105 ^a	1.036	1.55	1.5	0.31	0.46	0.23	0.35	Field	Wolff, 2015
1970 Clay	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 loom	Drip	No till	N	unknown	236	80 ^b		1.61±0.68			0.68		0.47	Field	Alaina at al. 2012
Salidy Ioalli	Microsprinkler	No till	IN	unknown	236	80 ^b		0.6±0.25			0.25		0.18	rielu	Alsina et al., 2015
Sandy loam: 67%	Drin	No till	N	114 N22	225	80 ^b		1.3±0.6			0.58		0.43	Eald	Colmonula 2015
sand, 19% silt, 14% clay	Drip	no till	IN	UAIN32	252	80 ^b		0.70 ± 0.02			0.3		0.21	Field	Carrecycle, 2015
Sandy loam: 60% sand 27% silt	Microsprinkler	No till	N	UAN32	258	78		0.65±0.07			0.25±0.03		0.19	Field	Decock et al. 2017
13% clay	merosprinkler	1.0 th	.,	0.11(52	280	82		0.53±0.19			0.19±0.07		0.18	1 Iolu	20000 et ul, 2017

344

^{*}N returned in hull and shall

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

347 ^aValue was adopted from Schellenberg et al., 2012

348 ^bValue was adopted from Decock et al., 2017

349	Table 5	Cable 5. Nitrous Oxide Emission Studies in California Walnut Orchard												
	Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha ⁻¹)	N2 O Emissions	Emission factors (% of applied N emitted as N ₂ C						

	Soil texture	Irrigation type	Tillage	Cover crop	N source	N input (kg N ha ⁻¹)	-	N ₂ O Emissions	ŝ	Emission factors (% of applied N emitted as N ₂ O)		Study scale	References
							Growing season (kg N ha ⁻¹)	Annual (kg N ha ⁻¹ yr ⁻¹)	Scaling factor (annual/sea son)	Seasonal	Annual		
	Silty clay loam: 18.8% sand, 47.6%	Microsprink ler	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
350	silt, 33.6% clay	Microsprink ler	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		al., 2016

351 ^aValue calculated from the emission factors from tree row (contributes 33%) and tractor row (contributes 74%); value

352 was not included in the statewide emission calculation

354 Table 6. California Specialty Crops Planting Acreage, Management Practices, N2O Emission Factors, and Statewide N2O emissions

						Uncorrect	ted for backgrou	ind flux	Correcte	d for backgrour	nd flux		IPCC Tier 1 method
Crop	Planted area (1,000 ha)	N input (kg N ha ¹)	Irrigation type	Tillage	Cover crop	Annual emission factors (% of applied N emitted as N ₂ O-N)	Statewide annual N ₂ O-N emissions, Mg	Statewide annual N ₂ O-N emissions, Gg CQ eq	Annual emission factors (% of applied N emitted as N ₂ O-N)	Statewide annual N ₂ O-N emissions, Mg	Statewide annual N ₂ O-N emissions, Gg CQ eq	Number of observations	derived statewide N ₂ O- N emissions, Gg CQ eq
Tomato	Fresh-market tomato: 12.14; Processing tomato: 121.4	Fresh-market tomato: 108; Processing tomato: 178	Furrow	Standard	Ν	$1.35\pm0.24^{a} \\$	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	Ν	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	Ν	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	Ν							0	6.9
Lettuce	131	166	Sprinkler/Drip	Standard	Ν	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
Cauliflower	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	Ν	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	Ν	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	Ν	0.43±0.15*	71.0±24.8	33.2±11.6	$0.31\pm0.04*$	51.2 ± 6.60	24.0 ± 3.09	0	77.3
Pistachio	126	131	Mircrosprinkler	No till	Ν	$0.25 \pm 0.05*$	41.3±8.25	19.3±3.86	$0.19\pm0.01\ast$	31.4 ± 1.65	14.7 ± 0.77	0	77.3
Walnut	148	122	Drip	No till	Ν	0.43±0.15*	77.6±27.1	36.3±12.7	$0.31 \pm 0.04*$	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	$0.19 \pm 0.01*$	34.3 ± 1.81	16.1 ± 0.85	0	84.6

355

^a Value is averaged from Kennedy et al (2013) and Burger and Horwath (2012)

356 357 ^b Value is calculated from equation 1

358 359 ^c Value is calculated from equation 2

* Value is adopted from almond system

360 Note: the EFs present in this table were calculated based on limited studies.

502		Stupe production classes, stratened by	giowing 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
363			
364			
363 364	<u>R4</u>	1,945	2,222

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

365 Table 8. DayCent model input parameterization statistics for almonds

	1		
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

368 Table 9. Model verification results for lettuce

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