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Compost Effects on Nitrogen and Carbon Cycling Dynamics and Water Use Efficiency in California Agroecosystems

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Compost Effects on Nitrogen and Carbon Cycling Dynamics and Water Use Efficiency in California Agroecosystems

By

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DISSERTATION

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Abstract

Composts are increasingly used as soil amendments to enhance soil health and agricultural sustainability. However, their effects on nitrogen (N) availability, and therefore their fertilizer potential, greenhouse gas (GHG) emissions, carbon (C) sequestration, and crop water use efficiency (WUE) are unclear. We conducted a three-year field experiment in sub-surface drip-irrigated tomatoes to investigate the effects of co-compost (FW; food waste and green waste) and green waste compost (GW) on yield and fertilizer N use, as co-composts are gaining research interest as a means of diverting organic waste from landfills and increasing the nutrient value of GW compost. Three FW and GW compost rates (0, 9 t ha^{-1}, or 18 t ha^{-1}) and four fertilizer N levels (0%, 70%, 85% and 100% of recommended rate) were examined, and treatment combinations were chosen to replace a certain fertilizer N input with compost N. ^{15}N-labeled fertilizer was used to determine fertilizer N crop use efficiency (true NUE), while apparent NUE was determined by comparing crop N uptake between treatments and no compost controls. In years two and three, FW and GW sustained and/or increased yield compared to no compost when low fertilizer rates were applied (0% N and 70% N), with lower apparent and true NUE observed in the compost treatments compared to no compost when 70% N and 85% N was supplied. These results suggest that compost served as an N source or primed the mineralization of soil N, potentially replacing fertilizer N crop uptake. Fertilizer N remaining in topsoil post-harvest was greatest for FW compared to GW and controls, while no difference in nitrate leaching potential was found among treatments, except in year two FW had the lowest nitrate leaching potential. These findings did not consistently produce statistically significant effects but show the potential role of compost in immobilizing fertilizer N and priming soil N mineralization.

Within the same field experiment, we also evaluated the effects of FW and GW compost on GHG fluxes and cumulative emissions, soil C content, and nematode populations and diversity as a potential biological mechanism for C sequestration. FW and GW composts had no significant effects on the fluxes or annual cumulative emissions of N_2O or CH_4, likely because emissions under sub-surface drip irrigation were already low and the considerable variation in low fluxes obscured the detection of significant changes. The application of compost significantly increased soil respiration (carbon dioxide (CO_2) emissions). The level of fertilizer N played a significant role in regulating N_2O and CH_4 emissions, as these fluxes decreased with lower levels of fertilizer N. Soil C content increased with the addition of FW and GW composts compared to the no-compost controls in the 0-15 cm soil layer.

This increase was not observed in deeper soil layers of 15-30, 30-60, or 60-90 cm. FW and GW compost treatments reduced $\delta^{13}C$ values in the topsoil, indicating that the newly added soil C was derived from the compost. Furthermore, compost application increased the presence of certain individual genera of bacterial- and fungal-feeding nematodes compared to the no-compost controls, offering insights into potential biological mechanisms for C decomposition and sequestration. However, no significant treatment effects were observed on individual genera of nematodes. These findings indicate that although compost may not directly lead to reductions of GHG, it holds potential for mitigating climate change by sequestering C in agricultural soils.

We also conducted a one-year field experiment in two, surface drip-irrigated, super-high density olive orchards (in Woodland and Stockton, California, hereafter named the MR and ST sites, respectively) to investigate the effects of compost and fertilizer N management on yield and intrinsic WUE (iWUE; the ratio between net CO_2 fixation and stomatal conductance). The olive industry in California is rapidly growing, and there is a need for irrigation and N management guidelines to be updated, taking climate change impacts on the state's diverse microclimates and increased tree density into consideration. At each field site in our trial, two GW compost rates (0 or 9 t ha^{-1}) and three fertilizer N rates (84, 112, or 140 kg N ha^{-1} at the MR site, and 28, 42, or 56 kg N ha^{-1} at the ST site) representing a low, medium, and high fertilizer rate were applied. During the growing season, monthly leaf sampling and concurrent soil sampling were conducted for analyzing $\delta^{13}C$, iWUE, and $\delta^{18}O$, which are proxies for plant water status, and soil gravimetric water content (GWC), NH_4^+, and NO_3^-. Compost increased yield compared to no-compost controls at the lowest fertilizer N level at the ST site, while no treatment effects on yield were observed at the MR site. Compared to no-compost controls, compost tended to increase olive $\delta^{13}C$ and iWUE at lower fertilizer N levels, indicating increased stomatal closure and plant water stress. In contrast, compost decreased $\delta^{13}C$ and iWUE at the highest fertilizer N level, indicating that additional N decreased compost-induced plant water stress. These treatment effects on $\delta^{13}C$ and iWUE were more significant at the MR site than the ST site, and the overall iWUE was lower at the ST site than the MR site, indicating less plant water stress in the ST site, likely due to higher precipitation, older tree age, and thus larger root zones for water uptake masking treatment effects. The result of compost increasing iWUE, suggesting that the compost increased water stress for olive trees, was unexpected because compost typically improves soil water-holding capacity for crop uptake. However, in our study, soil GWC results contradicted these typical compost effects on soil water, as there was generally a lack of significant treatment effects on GWC at both field sites. There were also no consistent compost or fertilizer N effects on olive $\delta^{18}O$, an

indicator for transpiration, and little to no treatment effects on soil NH_4^+ or NO_3^-. Soil NH_4^+ and NO_3^- did not show significant correlation with plant iWUE or $\delta^{18}O$, whereas soil GWC was positively correlated with iWUE at the MR site. Our results suggest olive iWUE was regulated more by soil water content than plant-available N content.

Chapter 1: Compost and fertilizer effects on yield and fertilizer nitrogen use efficiency in irrigated tomato crops

1.1. Introduction

The growing emphasis on achieving sustainability goals is prompting growers and food producers to increasingly turn to organic amendments. In the United States, nearly 40% of the food supply goes to waste and ends up in landfills, causing significant environmental concerns. In California for example, food waste accounts for 18% of the state's waste stream, highlighting the potential of diverting this material from landfills to expand the compost industry (Cotton 2019). Co-composted food and yard wastes (here after named FW), which combine food waste (5-15% by mass) with green waste (GW), are gaining interest in California and beyond as a means of diverting organic waste from landfills and increasing the nutrient value of GW compost. In addition to the desired outcomes of reducing the negative environmental impacts of landfills and promoting compost production and use, FW compost can improve soil health and deliver essential crop nutrients, possibly reducing N losses.

Composts are valuable amendments that improve soil structure, tilth, organic matter content, and carbon (C) sequestration (Diacono and Montemurro 2010; O'Connor et al. 2021), while enhancing water infiltration rates and water-holding capacity, making them particularly advantageous for arid regions like California with irrigation-intensive agriculture (Diacono and Montemurro 2010; Harrison et al. 2020). The benefits of compost application extend beyond soil health to include increased yields and decreased dependence on herbicides, pesticides, and fertilizers (Brown and Cotton 2011). These benefits are attributed to compost's positive impacts on soil physical properties, macronutrients, micronutrients, and total organic matter (Wong et al. 1999; Hargreaves et al. 2008; Brown and Cotton 2011). Recent literature on FW composts indicates their potential as biofertilizers and nutrient sources (O'Connor et al. 2021), particularly for the high-protein food waste feedstock. Historically, GW such as yard trimmings were the primary feedstock for compost used in California as a soil-health amendment, but they are an inconsistent nutrient source for crops (Hartz et al. 1996). The trend of composting GW together with FW is gaining popularity and is anticipated to yield composts with enhanced soil nutritional benefits, particularly in available nitrogen (N) content because of protein-derived N in FW feedstock (Sullivan et al. 2002; Farrell and Jones 2010; Kovács et al. 2014). Despite this, there is a significant gap in field-scale studies to understand how crop yield and N uptake are influenced by these composts. This information is essential for helping growers reassess fertilizer N inputs after applying compost.

After compost application, N mineralization from the organic fractions varies over time, and it may take years for consistent amounts to become available. This slow release of N allows it to be stored in soil for extended periods (Kumazawa 1984). To minimize losses from fertilizer N inputs, it is essential to synchronize soil N availability with crop demand, which can be affected by various factors, including soil properties, amount of fertilizer, and compost types and rates (Hamid and Ahmad 1995; Flavel and Murphy 2006; Zhu-Barker et al. 2015; Lazicki et al. 2020; Santos et al. 2021). The N mineralization rate and amount of N available for crop uptake depend on factors such as biochemical composition of compost, its C: N ratio, and application rate (Santos et al. 2021; Rothardt et al. 2021). FW compost is anticipated to have lower C: N ratios than GW compost due to higher N content from protein-rich food feedstocks, while GW compost feedstocks have higher percent C due to their high lignocellulose content (Liu et al. 2023). These potential differences in C: N ratios affect the composts' fertilizer potential since composts function as a fertilizer when the C: N ratio is between 20 to 30 (Reynolds et al. 2015). Additionally, composts with C: N ratios above 15 may cause temporary N immobilization in soil after application (Reynolds et al. 2015), as the compost's organic C stimulates microbial activity, leading to the uptake of mineral N and its conversion into organic forms. However, there is a lack of or confounding field-scale research on the effects of compost application on soil N transformation, bioavailability, and losses (O'Connor et al. 2021). Moreover, there is a need to understand the contributions of fertilizer N to soil and crop N pools when compost and fertilizer are applied together, which affects same-season and subsequent crop productivity (Choi et al. 2001). Therefore, it is crucial to comprehend the temporal effects of consecutive annual field applications of FW composts, an evolving scenario in sustainable agriculture.

However, there is a flip side: using FW compost to improve soil health might also heighten concerns about nitrate (NO_3^-) leaching, with FW compost serving as a source of N or promoting soil mineralized N that could contribute to this loss pathway. Generally, the effect of compost on NO_3^- leaching might be influenced by its C:N ratio and C quality, which govern N processes like immobilization and (de)nitrification, subsequently impacting NO_3^- leaching (Xu et al. 2020). For example, the high organic C content in compost could promote N immobilization in soils and therefore reduce NO_3^- leaching (Sullivan et al. 2003; Diacono and Montemurro 2010). In a sandy soil, both FW and GW composts have been found to reduce NO_3^- leaching compared to food-based digestate and pig slurry (Nicholson et al. 2017). Further investigation is warranted to fully explore the impact of FW and GW composts on N cycling dynamics in cropping systems.

In this study, the N cycling dynamics were investigated in a processing tomato site with silty clay loam soil, which had been under long-term conventional management in the Sacramento Valley, California. The agricultural practices in this region include intensive use of fertilizer N, subsurface drip irrigation, and frequent tillage, which could affect the sustainability of N management and soil health outcomes. Thus, this agroecosystem provided an opportune setting to study the impacts of GW and FW compost application. Our field experiments spanned three consecutive years and involved three different application rates of GW and FW composts in combination with a range of fertilizer N levels. To assess fertilizer N used by crops and to estimate crop “true” fertilizer N use efficiency (NUE), we used isotopically labeled ^{15}N-fertilizer. While apparent fertilizer NUE considers all N sources (i.e., compost, soil, fertilizer), true NUE considers only fertilizer N. The amount of compost N in crops and soil cannot be quantified without using isotopically labeled compost, which is cost-prohibitive and was thus not implemented in this study. The primary aim of this study was to evaluate the effects of FW and GW composts on crop N uptake and fate of fertilizer N in the cropping systems. To achieve this goal, we investigated tomato yield, true and apparent NUE, NO_3^- leaching potential, and fertilizer N remaining in soil after harvest. This study provides insight to develop sustainable N management practices that include recommendations for compost use in combination with fertilizer N.

1.2. Materials and methods

1.2.1. Site descriptions and agronomic management

Field experiments were conducted at the University of California-Davis Russell Ranch Sustainable Agriculture research site (38°32’33’’N, 121°52’33’’W). Climate conditions of monthly air temperature and precipitation for the site are shown in Fig. S1.1 (see Supplementary information). The soil is classified as Rincon silty clay loam, a fine smectitic, thermic Mollic Haploxeralf according to the United States Department of Agriculture, National Cooperative Soil Survey. This site has been in a tomato-corn rotation from 2013 to 2018 with corn in even years and tomato in odd years and has been managed under conventional practices (e.g., fertilizer NPK only, conventional tillage, and no organic amendments). Subsurface drip irrigation with fertilizers (i.e., fertigation) was implemented since 2014 and represents industry standard practice. During the project term, tomato crops were mechanically transplanted on May 1, 2019; April 21, 2020; and April 27, 2021. Following harvest each year, crop residues were incorporated into the soil. Baseline soil characteristics are summarized in Table 1.1.

Table 1.1. Characteristics of soils before the setup of field trials.

Depth (cm)	Total N ($g\ kg^{-1}$)	Total C ($g\ kg^{-1}$)	pH (H_2O 1: 1)	DOC ($mg\ kg^{-1}$)	NH_4^+ ($mg\ kg^{-1}$)	NO_3^- ($mg\ kg^{-1}$)	Bulk density ($Mg\ m^{-3}$)
0-15	1.47	15.2	7.13	17.2	1.13	11.0	1.43
15-30	1.26	14.1	7.43	13.8	1.09	4.83	1.68

1.2.2. Compost/Fertility management and experimental design

At the research site, sixteen experimental treatments were set up as a split-plot randomized block design with three blocks (replicates). The treatments included two compost types (GW or FW) × three compost rates (0, 9 t ha^{-1}, or 18 t ha^{-1}) × two fertilizer N levels (0 or 100% of recommended rate at 202 kg N ha^{-1}). In addition, different compost application rates combined with a corresponding reduction in fertilizer N rates were selected to replace N from the fertilizer with compost sources: 85% of recommended N rate × compost (GW or FW at the rate of 0 or 9 t ha^{-1}) and 70% of recommended N rate × compost (GW or FW at 0 or 18 t ha^{-1}). The fertilizer N application rates were chosen to compare a range of rates: 0% N as the control, 100% N as the full locally recommended rate, and 15% and 30% fertilizer N reductions (85% and 70% N) were chosen because N recommendations and budgets often overestimate fertilizer N requirements and lower rates have been reported to be sufficient for tomato crops (Geisseler et al. 2020). The compost rates were chosen to have a no-compost control, and 9 t ha^{-1} (4 ton $acre^{-1}$) is the recommended dry weight compost application rate for annual crops in California (Gravuer and Gunasekara 2016) and was combined with the 15% fertilizer N reduction rate, and double the recommended compost application rate (18 t ha^{-1} (8 ton $acre^{-1}$)) was combined with the 30% fertilizer N reduction rate. The amount of N applied in each compost treatment in 2019 was as follows: 9 t FW ha^{-1} = 120 kg N ha^{-1}, 9 t GW ha^{-1} = 113 kg N ha^{-1}, 18 t FW ha^{-1} = 240, 18 t GW ha^{-1} = 226, and in 2020 was as follows: 9 t FW ha^{-1} = 150 kg N ha^{-1}, 9 t GW ha^{-1} = 153 kg N ha^{-1}, 18 t FW ha^{-1} = 300 kg N ha^{-1}, 18 t GW ha^{-1} = 306 kg N ha^{-1}. Due to field setup limitations, fertilizer levels were the main factors, while compost types and rates served as split factors. The size of each experimental plot was 4.6 m × 6.1 m. The N fertilization occurred as separate fertigation events every two to three weeks during the growing period through subsurface drip irrigation lines (UAN 32 at a rate of 34-56 kg N ha^{-1} each event). Treatment applications were carried out over three consecutive years. Commercially purchased composts were hand spread evenly on the soil surface and disked in with standard equipment to a depth of 10-15 cm in the spring of 2019 and the fall of 2019 and 2020. The FW compost was 5% food waste and 95% urban yard waste. The GW was 100% urban yard waste. See Table 1.2 for compost characteristics.

In 2019, subplots with no fertilizer N inputs were selected and ^{15}N-urea (10 atom% ^{15}N enrichment) was injected (to mimic subsurface irrigation fertigation). Each subplot was 1.5 m × 1.7 m in size and contained 5 plants. In these microplots throughout the growing season at the same times, volumes, and rates, N fertilization events were matched to N application in other plots to determine the contribution of fertilizer N to crop N uptake and to calculate fertilizer true NUE and fertilizer N remaining in soil at the plot level post-harvest. The injections were done by first inserting a hollow metal tube ~5 cm from the irrigation drip tape between plants to a depth of 25 cm to match the depth of subsurface drip lines. A syringe filled with the appropriate volume and concentration of ^{15}N-urea was connected to the tube with plastic tubing and the isotopically labeled fertilizer was injected.

Table 1.2. Compost characteristics used in each compost treatment application.

Season/Year	Compost	pH	Total C (g kg^{-1})	Total N (g kg^{-1})	C: N	NH_4^+ (mg kg^{-1})	NO_3^- (mg kg^{-1})
Spring 2019	FW	8.08	285.3	10.38	27.49	3.32	119.13
	GW	8.21	266.7	13.83	19.28	28.71	103.10
Fall 2019	FW	7.90	282.6	13.32	21.22	607.94	8.59
	GW	7.23	312.3	12.52	24.94	32.66	301.32
Fall 2020	FW	7.20	311.8	16.67	18.70	106.97	0.67
	GW	6.97	298.4	16.97	17.58	227.59	1.11

1.2.3. Soil sampling and analysis

Soil samples from the 0-15 cm soil layer of each treatment plot were collected from 4 composite borings with a 1.83-cm diameter steel corer before and after fertigation events and approximately monthly during the remainder of each year. Additional soil samples were collected from the ^{15}N microplots after each year's harvest to a depth of 30 cm using a PN150 JMC Environmentalist's Sub-Soil Probe and were separated into 0-15 cm and 15-30 cm soil layers.

Nitrate (NO_3^-) and ammonium (NH_4^+) (together named inorganic N) and dissolved organic C (DOC) were measured in all the soil samples by extracting well-mixed soil with 0.5 M K_2SO_4 (4:1 extractant volume to soil mass ratio). The soil extract was analyzed colorimetrically for NH_4^+ and NO_3^- using a Shimadzu spectrophotometer (Model UV-Mini 1240) (Forster 1995; Doane and Horwáth 2003). Gravimetric soil moisture was calculated by comparing the field-moist and oven-dry (105ºC) mass of soil samples. DOC was measured using a total organic carbon (TOC) ultra-violet (UV)-persulfate oxidation analyzer (Model Phoenix 8000, Teledyne Tekmar, Mason, OH). The pH of soil was measured in a soil slurry (1:1 soil to water ratio) with a pH meter (Thermo Scientific Orion 9156BNWP Combination pH Electrode, Taylor Scientific, St. Louis, MO). The data for NO_3^-, NH_4^+, DOC, and pH are shown in Figs. S1.2, S1.3, S1.4 and S1.5 (see Supplementary information). The ^{15}N isotopic analyses of soil samples were performed at the UC Davis Stable Isotope Facility.

1.2.4. Nitrate leaching potential determination

Resin bags were buried 30 cm deep in years 2020 and 2021 to determine NO_3^- leaching potential over the winter rainy season in the 100% N plots with no compost and with the highest application rate (18 t ha^{-1}) of FW and GW composts. The resin bags were made by filling nylon stockings with 50 g NO_3^- specific ion exchange resin (AmberLite™ PWA 5, Dow Chemical Co., Waterfall City, Midrand). After the resin bags were removed from the ground in March both years, the resin was extracted with 150 mL of 1M KCl. The extracts were analyzed colorimetrically for NO_3^- following the same protocol as mentioned above (Doane and Horwath, 2003).

1.2.5. Yield measurements and plant analysis

Each year, tomatoes were harvested in late August. In each plot, three adjacent tomato plants were randomly selected, and the aboveground biomass were separated into fruits and residues. Yields, biomass, and N content of the aboveground plant parts (tomatoes and leaf and stem residues) were measured. Subsamples of residues and tomatoes were collected. Leaf and stem residues were dried in a 60ºC oven and prepared for total N and C analysis. Tomatoes were blended into a slurry, and the dry weight of the fruit was assessed by lyophilizing a portion of the tomato slurry. The freeze-dried samples were subsequently analysed for total N and C content using an elemental analyser

(EAS 4010, Costech Analytical Technologies Inc., Valencia, CA). In the subplots that received ^{15}N-labeled fertilizer, three central tomato plants were harvested and prepared using the same method as described above, and after subsampling the labeled residues were returned to the subplots. The ^{15}N isotopic analyses for plant samples collected from the isotopically N-labeled microplots were performed at the UC Davis Stable Isotope Facility. Total N and ^{15}N enrichment values were used to calculate apparent NUE, true NUE, and fertilizer N remaining in soil after harvest.

1.2.6. Calculations

Crop biomass was used to determine the total crop N uptake in hectares. The contribution of fertilizer N to total crop N uptake and fertilizer NUE were then determined. The fertilizer apparent NUE and true NUE were calculated using data collected from the plots receiving unlabeled fertilizer (without ^{15}N addition) and isotopically labeled fertilizer (^{15}N), respectively. Apparent NUE is the percentage of crop N out of the total amount of applied N (Eq. 1.1). True NUE is the percentage of fertilizer N taken up by crops. True NUE was calculated based on the N content and ^{15}N enrichment of plant material, the applied fertilizer's amount and ^{15}N enrichment (10 atom%), and the background ^{15}N signatures of plant samples in the unlabeled plots (without ^{15}N addition). Calculations for both NUEs are as follows:

$$Fertilizer\ apparent\ NUE\ (\%) = 100\ \times\ [(\text{plant total N from fertilized plots} - \text{plant total N from unfertilized plots})\ /\ (\text{fertilizer N} + \text{compost N applied})]\quad \text{(Eq. 1.1)}$$

$$Fertilizer\ true\ NUE\ (\%) = 100\ \times\ [\mathrm{p}\ (\mathrm{c} - \mathrm{b})\ /\ \mathrm{f}\ (\mathrm{a} - \mathrm{b})]\quad \text{(Eq. 1.2)},$$

where p = total N in plant material, f = fertilizer N applied, c = ^{15}N atom% in plant material, a = ^{15}N atom% in fertilizer, and b = natural abundance of ^{15}N (atom%) in plants (Cabrera and Kissel 1989). The calculation for fertilizer N remaining in soil is as follows:

$$Fertilizer\ N\ in\ soil\ (g\ m^{-2}) = [\mathrm{p_s}\ (\mathrm{c_s} - \mathrm{b_s})\ /\ (\mathrm{a} - \mathrm{b_s})]\qquad \text{(Eq. 1.3)},$$

where p_s = total N in soil, c_s = ^{15}N atom% in soil, a = ^{15}N atom% in fertilizer, and b_s = natural abundance of ^{15}N (atom%) in soil.

Nitrate leaching potential was calculated based on the NO_3^- in a resin bag extract, volume of KCl used in the extract, and area of the circumference of the resin bag.

1.2.7. Statistical analyses

Statistical analyses were conducted using linear mixed models in R program to investigate relationships between compost and fertilizer N treatments. For each metric, compost and N fertilizer treatments were treated as fixed effects and replicates were treated as random effects. Analysis of variance were performed to compare treatment effects and to determine if effects were significant ($p<0.05$). Data was transformed as needed to normalize distributions and homogenize variances prior to statistical analyses. ANOVA was performed for the metrics of yield, apparent NUE, true NUE, and fertilizer N remaining in soil with fixed factors of compost, N level, and compost x N level interaction (Table 1.3). The groups in the compost factor are no compost (control), FW 9 t ha^{-1}, FW 18 t ha^{-1}, GW 9 t ha^{-1}, and GW 18 t ha^{-1} and the groups in the N level factor are 70%, 85%, 100%, and 0% of the local recommended N rate when applicable.

1.3. Results

1.3.1. Tomato yield

The influence of compost types and rates on crop yield varied considerably (Fig. 1.1). Compared to no N addition, adding N fertilizer generally increased tomato yield, as expected. However, the yields in the no compost treatments with 100% N were not higher than those with 85% N in any of the three years, indicating that yield was not negatively affected by this reduction in N. In 2020, the compost treatments began to exhibit a higher yield tendency than the no compost treatments when reduced fertilizer N was applied, although differences are not greatly statistically significant (Fig. 1.1b). Moreover, the yields in the compost treatments of 70% N in 2020 were similar to those in 2019, while yields in plots without compost declined from 2019 (~130 t ha^{-1}) to 2020 (~92 t ha^{-1}). In 2021, the impact of compost was only evident in the 0% N treatments, where a significant decrease in yield was observed in the absence of compost (Fig. 1.1c).

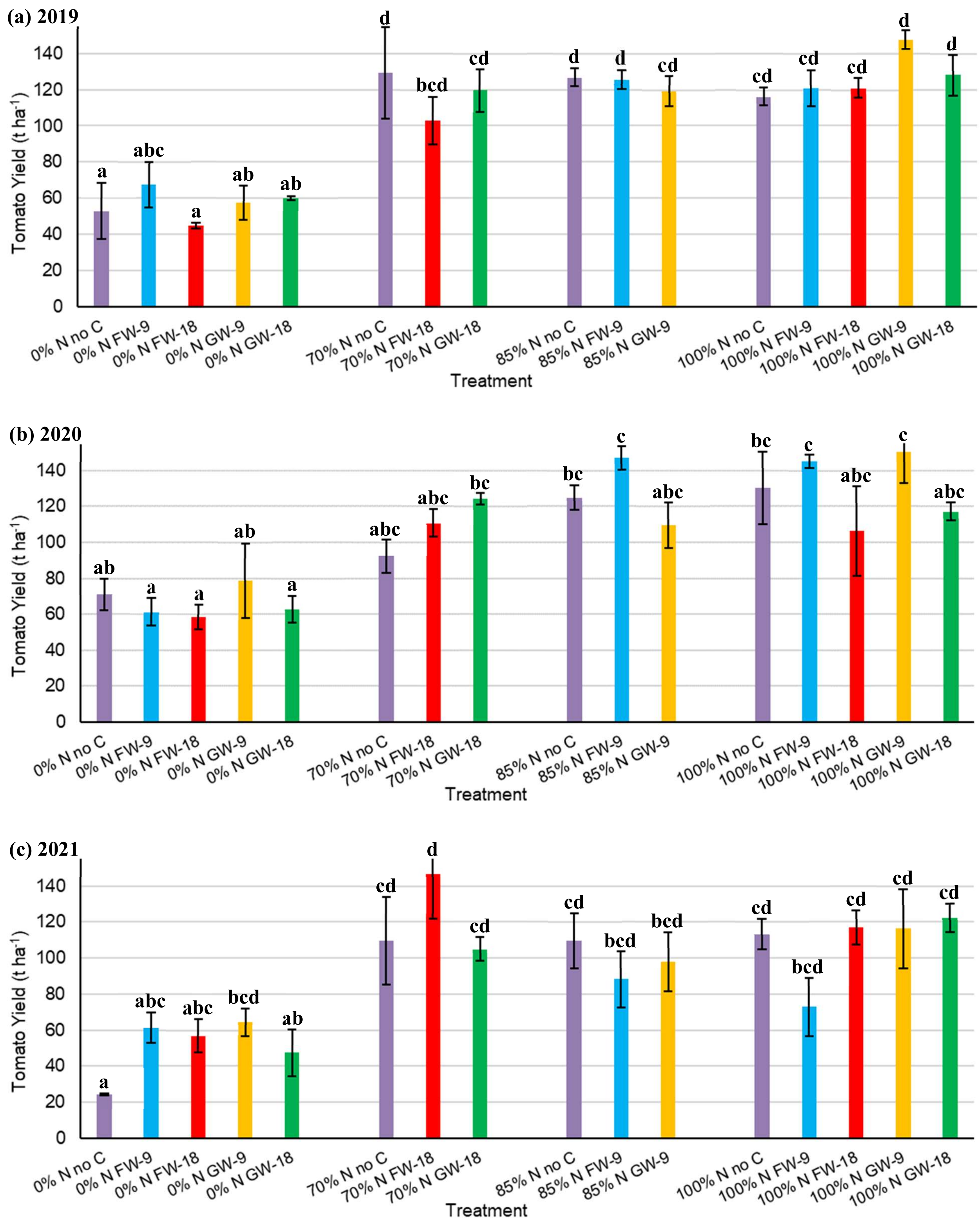


Figure 1.1. Tomato yield for (a) 2019, (b) 2020, and (c) 2021 for the two compost types (FW and GW) at three application rates (no C = no compost, 9 = 9 t ha^{-1}, and 18 = 18 t ha^{-1}), and four N levels (0%, 70%, 85%, and 100%)

of the recommended amount. The line bars represent standard error (n = 3). Letters indicate significant differences in yield among treatments at a statistical significance level of alpha = 0.05.

1.3.2. Apparent NUE

In the present study, apparent NUE (also known as total NUE or plant NUE in the literature) represents the plant N derived collectively from all N sources, including fertilizer, compost, and soil. Overall, compost application resulted in a reduction of apparent NUE in all three years, as illustrated in Fig. 1.2. Notably, FW compost treatments tended to exhibit the lowest apparent NUE across all fertilizer N levels in each year, while the apparent NUE of GW compost treatments was also generally lower than that of the control treatments, except for the treatment involving 9 t ha^{-1} of GW compost with 100% N in 2019. The apparent NUE of compost treatments tended to increase from 2019 to 2020, except for GW compost with 100% N. This trend was then followed by a decrease in apparent NUE from 2020 to 2021, although the 2021 apparent NUE values were typically higher than those in 2019.

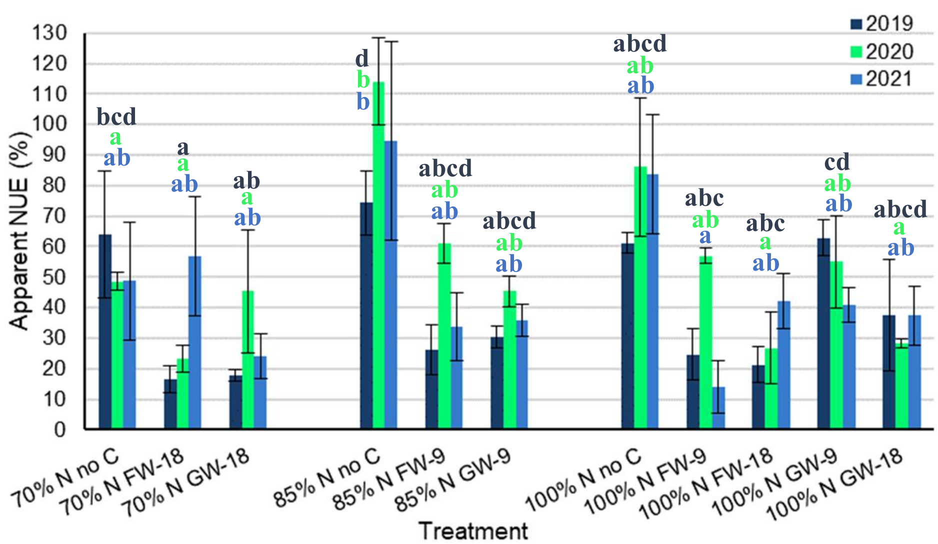


Figure 1.2. Apparent NUE of tomato crops for three consecutive years (2019-2021) for the two compost types (FW and GW) at three application rates (no C = no compost, 9 = 9 t ha^{-1}, and 18 = 18 t ha^{-1}), and three N levels (70%, 85%, and 100%) of the recommended amount. The line bars represent standard error (n = 3). Letters indicate significant differences in apparent NUE among treatments at a statistical significance level of alpha = 0.05.

1.3.3. True NUE

True NUE, which is defined as the percentage of fertilizer N recovered by plants, is calculated based on data from ^{15}N microplots (Fig. 1.3). Unlike apparent NUE, this metric only considers N derived from fertilizer and excludes compost or soil sources. The yearly data for the same treatments were stacked to illustrate the true total fertilizer N recovery over the three-year study period, given that ^{15}N labeled fertilizer was only applied in the first year (2019) (Fig. 1.3). Results indicate that compost application generally reduced true NUE compared to controls over the three years, although statistically significant differences were not observed. After year one, FW compost tended to have the lowest true NUE. Compared to controls, compost application with reduced fertilizer rates (70% N and 85% N) resulted in lower true total fertilizer N recovery after three years. At the 100% N fertilizer level, FW compost exhibited the lowest total values, while GW compost application had similar values to the control.

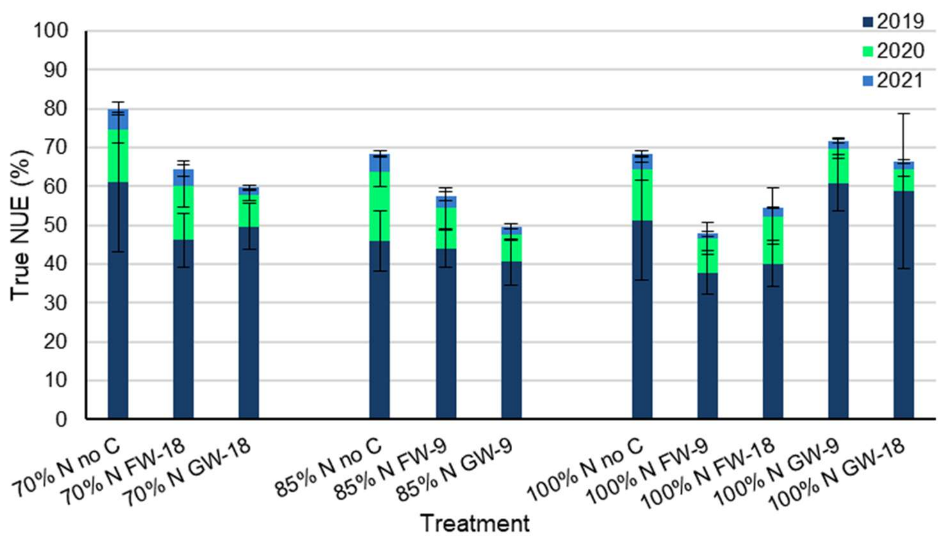
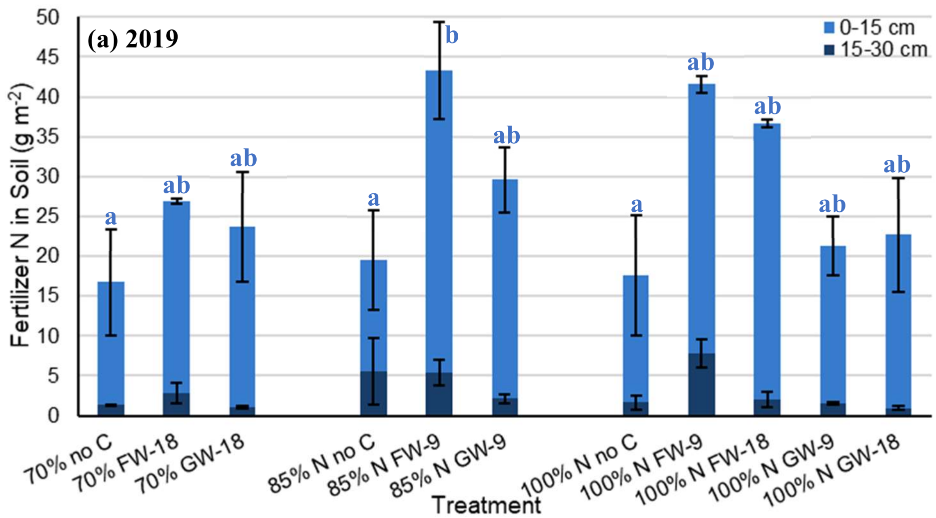


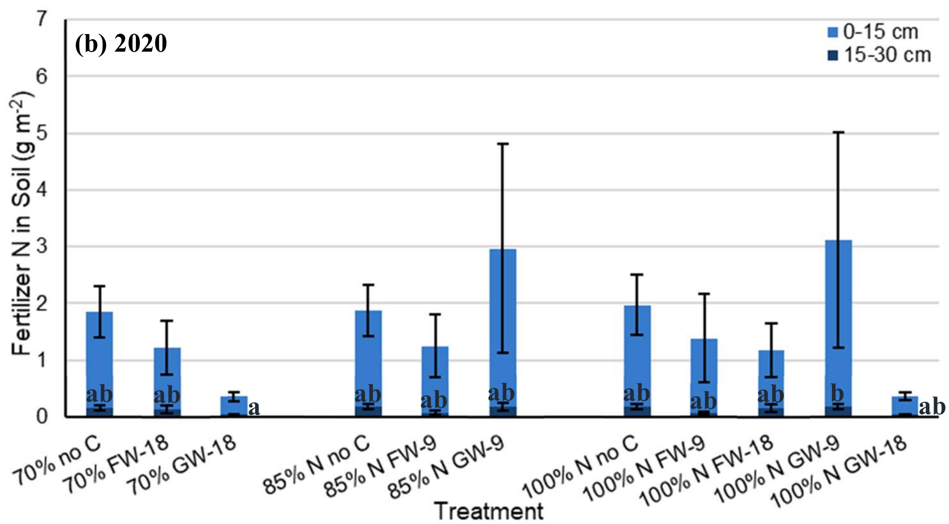
Figure 1.3. True NUE (percent of fertilizer N recovered (NdF) by tomato crops) for three consecutive years (2019-2021) for the two compost types (FW and GW) at three application rates (no C = no compost, 9 = 9 t ha^{-1}, and 18 = 18 t ha^{-1}), and three N levels (70%, 85%, and 100%) of the recommended amount. The line bars represent standard error (n = 3). No statistically significant differences among treatments were observed.

1.3.4. Fertilizer N remaining in soil after harvest

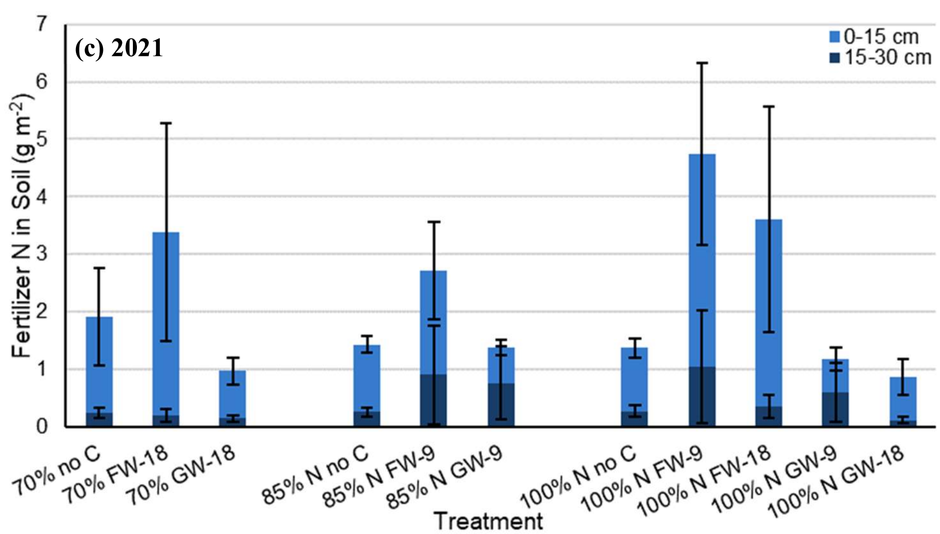
The compost types and rates had an impact on the fertilizer N remaining in the 0-15 cm and 15-30 cm soil layers after each year's harvest, as shown in Fig. 1.4. However, due to the high variability, the observed differences were not statistically significant among these treatments, except in the 0-15cm soil layer in 2019 and the 15-30 cm soil layer in 2020. The overall decrease in fertilizer N soil retention observed across all treatments from 2019 to subsequent years can be attributed to the fact that ^{15}N-labeled fertilizer was only applied during the first year's growing season. Throughout the three years, the majority of the fertilizer N was retained in the 0-15 cm soil layer, while the 15-30 cm soil layer contained much less fertilizer N. In 2019 and 2021, FW compost application generally resulted in higher levels of fertilizer N retained in the soil, whereas the application of GW compost led to similar levels of fertilizer N retained in the soil compared to the control where no compost was applied (Fig. 1.4a and c).



(a) 2019
0-15 cm
15-30 cm
Fertilizer N in Soil (g m-2)
Treatment
70% no C
70% FW-18
70% GW-18
85% N no C
85% N FW-9
85% N GW-9
100% N no C
100% N FW-9
100% N FW-18
100% N GW-9
100% N GW-18



(b) 2020
0-15 cm
15-30 cm
Fertilizer N in Soil (g m-2)
Treatment
70% no C
70% FW-18
70% GW-18
85% N no C
85% N FW-9
85% N GW-9
100% N no C
100% N FW-9
100% N FW-18
100% N GW-9
100% N GW-18



(c) 2021
0-15 cm
15-30 cm
Fertilizer N in Soil (g m-2)
Treatment
70% no C
70% FW-18
70% GW-18
85% N no C
85% N FW-9
85% N GW-9
100% N no C
100% N FW-9
100% N FW-18
100% N GW-9
100% N GW-18

Figure 1.4. Fertilizer N remaining in the 0-15 cm and 15-30 cm soil layers in the microplots after tomato harvest in (a) 2019, (b) 2020, and (c) 2021 for the two compost types (FW and GW) at three application rates (no C = no compost, 9 = 9 t ha^{-1}, and 18 = 18 t ha^{-1}), and three N levels (70%, 85%, and 100%) of the recommended amount. Note that the y-axis values are greater for 2019 than 2020 and 2021. The line bars represent standard error (n = 3). No statistically significant differences between treatments were observed at a level of alpha = 0.05, except for the 0-15 cm soil layer in 2019 and the 15-30 cm soil layer in 2020.

1.3.5. Nitrate leaching potential

The NO_3^- leaching potential of the highest application rate (18 t ha^{-1}) of FW and GW composts was compared to control plots with 100% N fertilizer, based on NO_3^- concentrations extracted from ion exchange resin bags buried in two consecutive winter rainy seasons. The results, presented in Fig. 1.5, show that in 2020 after two years of compost application, the plots that received FW compost application had the lowest leaching potential of NO_3^-, while the plots that received GW compost had similar NO_3^- leaching potential to the controls. However, in 2021, no significant difference was observed between compost and the control. It should be noted that this metric was only analyzed for years 2020 and 2021, and further experimentation is necessary to confirm the short- and longer-term observations.

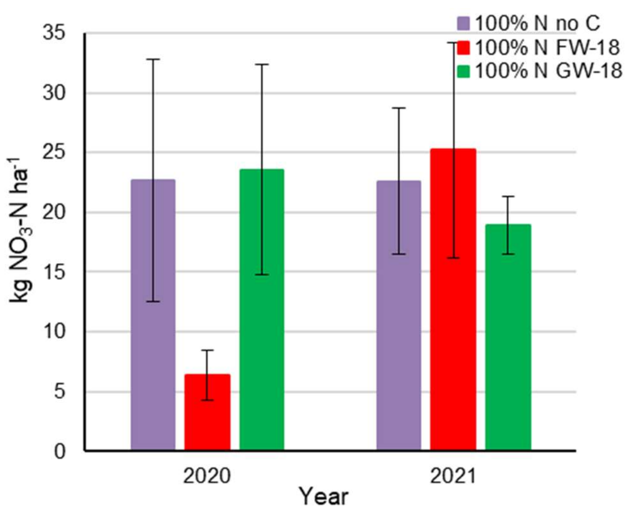


Figure 1.5. Nitrate (NO_3^-) leached during the rainy season of 2020 and 2021 from the top 30 cm of soil in the control and highest compost application rate (18 = 18 t ha^{-1}) plots. The line bars represent standard error (n = 3). No statistically significant differences between treatments were observed at a level of alpha = 0.05.

1.4. Discussion

Our results shed light on the role of compost in crop N uptake: providing a source of N and/or promoting soil to mineralize N for crops while immobilizing fertilizer N in soil, a phenomenon called "priming" effect that has been first reported by Jenkinson *et al.* (1985). The findings of our study reveal that the effects of FW and GW composts application on tomato yield are not consistent across years, and the greatest positive impacts were observed in 2020 and 2021 when either compost was applied in combination with reduced fertilizer N inputs compared to the controls. Total crop N uptake was similar between controls and all compost treatments each year (Fig. S1.6). These yield and N uptake data indicate that FW and GW composts can sustain or even improve crop yield in the absence or reduction of fertilizer N, likely because they can serve as a source of N for plant uptake and/or prime soil N mineralization. Nevertheless, the yields obtained without fertilizer N inputs were significantly lower compared to those obtained with sufficient N inputs (Fig. 1.1). These results are consistent with previous studies that demonstrated increased yields in various crops after applying FW compost (Sullivan et al. 2002, 2003; Yang et al. 2014; Kovács et al. 2014; Drury et al. 2014; Reynolds et al. 2015; Chew et al. 2018) and GW compost (Hartz et al. 1996; Drury et al. 2014; Ben-Laouane et al. 2021). However, the present study highlights that multiple years of compost application are likely necessary to achieve consistent yield benefits, as we observed positive compost effects only in the second (2020) and third (2021) years. This delayed yield effect may be due to temporary N immobilization, followed by N mineralization, which is in line with previous studies that demonstrated a lack of significant yield increase after a single high-rate FW application in the first year, but increased yields in following years (Sullivan et al. 2002, 2003; Reynolds et al. 2015). Similar results were also found in a field trial in Iran where the marketable tomato yield was increased from year one to year two following GW compost application, but with no change in total tomato yield between years (Ghorbani et al. 2008).

Irrespective of compost addition, a reduction of up to 15% of the recommended N rate can be achieved while maintaining comparable tomato yields, as evidenced by the similar yield tonnage observed in the no-compost treatments at 100% and 85% N fertilizer rates in all three years (Fig. 1.1). This finding suggests there is over-fertilization beyond the maximum yield efficiency point and economic optimum fertilizer rate where the addition of extra fertilizer does not increase the yield at the locally recommended seasonal application rate of 202 kg N ha^{-1}. California tomato growers apply a wide range of N fertilizer with seasonal rates ranging from 140 to 280 kg N ha^{-1}, and it has been found that under normal growing conditions, maximum yields can be obtained with 112 to 168 kg N ha^{-1} (Hartz et al. 2008), and the 85% N treatment in our study equates to 172 kg N ha^{-1} which falls into the higher end of the optimal fertilization range. Despite our yield results without and with compost application, it is important to note that the increase in yield alone does not establish a correlation with improved N supply from either compost or fertilizer following compost addition. This is examined further by investigating the NUE among treatments.

The concept of temporary N immobilization followed by enhanced mineralization in later years is supported by both fertilizer apparent and true (based on ^{15}N labeled fertilizer) NUE results, which quantify the proportion of plant N derived from either soil + compost + fertilizer or fertilizer alone, respectively (Figs. 1.2 and 1.3). The distinction between these two NUE metrics is important to consider because only 40% or less of fertilizer N has been reported to contribute directly to plant uptake (Jenkinson et al. 1985; Yan et al. 2020). The contribution of compost N to crop uptake cannot be quantified without using isotopically labeled compost, which was not implemented in this study due to prohibitive costs. However, by comparing the compost and no compost treatments, we can at least speculate the influence of compost on crop N uptake. For all fertilizer levels and years, we found that apparent NUE and true NUE were highest when no compost was applied. This finding is supported by the A-value concept where adding another N source, as in our case is compost, dilutes the fertilizer N taken up by crops (Broadbent 1970). ANOVA results further support that compost had a significant effect on both apparent and true NUE, and this effect gained significance each consecutive year for true NUE (Table 1.3). The lower true NUE observed in compost-treated crops provides evidence that compost addition reduced the uptake of fertilizer N by tomato crops, indicating that the assimilated plant N originated partly from compost or compost-primed soil N. Previous studies have shown that composts with high C:N ratios (>15), such as those used in our study (Table 1.2), can induce temporary immobilization of fertilizer N in soils (Hadas et al. 1996; Reynolds et al. 2015). This process likely further contributes to the low true NUE observed in compost-treated crops. Nevertheless, the increased apparent NUE in compost-treated crops from 2019 to 2020, may be attributed to reduced immobilization of fertilizer N, as a result of microbial turnover.

Table 1.3. ANOVA summary of statistical significance of the treatment factors: compost, N level, and compost x N level interaction, for the metrics of yield, apparent NUE, true NUE, and fertilizer N remaining in soil for 2019-2021.

Treatment factor	Year	Yield	Apparent NUE	True NUE	Fertilizer N remaining in soil (0-15 cm)	Fertilizer N remaining in soil (15-30 cm)	Total fertilizer N remaining in soil (0-30 cm)
Compost	2019	NS	***	NS	**	*	**
	2020	NS	**	.	***	***	***
	2021	NS	**	*	**	NS	*
N level	2019	***	NS	NS	NS	NS	NS
	2020	***	NS	NS	NS	NS	NS
	2021	***	NS	NS	NS	NS	NS
Compost × N level	2019	NS	NS	NS	NS	NS	NS
	2020	NS	.	NS	NS	NS	NS
	2021	**	NS	NS	NS	NS	NS

NS Not significant.

. Significant at the 0.1 probability level.

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

The lower true NUE trend observed in FW and GW composts treatments could also be attributed to the priming effect of compost on indigenous soil N mineralization, which in turn enhances the immobilization of fertilizer N and leads to lower fertilizer N uptake (Jenkinson et al. 1985). Studies reported that compost application causes high immobilization of urea-N, resulting in lower fertilizer N uptake efficiency in a corn system (Choi et al. 2001). Other studies have also demonstrated that the retention of applied N in soil can be enhanced by the addition of organic straw amendments, as a result of the stimulation of fertilizer N immobilization by the straw amendment and the accumulation of straw N in N pools (Bai et al. 2020). With each successive year of compost application, more compost N may accumulate in the soil, providing future available N. It has been demonstrated that FW compost provides a consistent, slow-release N source for grass growth in the long-term (Sullivan et al. 2003) and other composts including GW compost similarly exhibit slow N release (Diacono and Montemurro 2010; Cassity-Duffey et al. 2020). The release of inorganic N from FW compost is slow through the action of competitive and diversified microbial processes, thereby increasing the longevity of nutrient provisions and reducing soil N losses in the forms of nitrous oxide (N_2O) emissions or NO_3^- leaching (Lee et al. 2004; Palaniveloo et al. 2020). Furthermore, fertilizer N immobilization may drive a pool substitution process, where microbes take up fertilizer N and in turn “repay” indigenous soil N pool to the soil available N pool (Xu et al. 2023). This effect increases in significance in soils with high N immobilization rates (Mary et al. 1996; Xu et al. 2023). Therefore, compost may serve as a reliable source of N for current and future crops after a few years of consecutive applications.

Food waste compost leading to fertilizer N immobilization was further evidenced by the higher rates of fertilizer N remaining in soil post-harvest in plots that received FW compost compared to GW compost and controls, though treatment effect did not always show statistical significance (Fig. 1.4). More fertilizer N remaining in the 0-15 cm soil layer than the 15-30 cm soil layer post-harvest also demonstrates that N leaching is likely not a major source of fertilizer N loss during the growing seasons (Fig. 1.4). FW compost leading to temporary N immobilization was further supported by its lower NO_3^- leaching potential during the winter rainy seasons compared to GW compost and controls in 2020 after two years of application, although no statistically significant differences were found among treatments due to relatively high standard error (Fig. 1.5). This trend disappeared in 2021 after three years of consecutive compost applications, suggesting that the N immobilization was temporary. GW compost had similar leaching potential as the controls each year. These findings are consistent with previous studies that reported no significant differences in NO_3-N leachate between non-amended and GW compost-amended soils (Hartz et al. 1996), and very low leaching from both FW and GW composts (Nicholson et al. 2017). Other studies have also shown that substituting fertilizer N with organic amendments such as composted pig and cattle manure can significantly reduce N leaching loss (Xu et al. 2020; Rothardt et al. 2021). Moreover, Colombani et al. (2020a, 2020b) demonstrated that urban organic waste compost reduces NO_3^- leaching by increasing denitrification rates, since NO_3^- serves as a substrate for this process, as evidenced by their modeling and validation (2020b) and soil column leaching experiments (2020a). The statistically insignificant effects between FW and GW compost treatments throughout this study is likely partially due to the similarity of the compost feedstocks, with FW compost containing only 5% food waste and 95% green waste, compared to 100% green waste in the GW compost. Another reason for the lack of difference is the field site had baseline inorganic N content of ~40.9 kg N ha^{-1} in the top 30 cm of soil (Table 1.1), indicating that there was high residual N from previous crop management that likely overshadowed the differences between FW and GW compost treatments.

The use of compost as an additional source of N for agricultural soils raises concerns about potential NO_3^- leaching into groundwater. However, the results of this study indicate that these concerns may be mitigated, and the use of FW compost could even temporarily improve local groundwater NO_3^- levels, given that it has been previously suggested as an alternative to fertilizer N for reducing groundwater pollution by retaining nutrients in soil (Palaniveloo et al. 2020). Although the results suggest that three years of FW or GW compost application may not increase water NO_3^- levels, longer experimentation would be necessary to confirm this trend in subsequent years. It should also be noted that leaching potential was only investigated in plots with 100% N fertilizer, and the use of compost in combination with reduced fertilizer application would likely result in further reductions in NO_3^- leaching.

In addition to immobilization, other N cycling processes may have contributed to the low NO_3^- leaching potential observed in the FW compost treatment in this study. The added organic C in compost may have stimulated denitrification by providing a substrate for denitrifiers and creating anaerobic conditions through increased microbial respiration and depletion of soil oxygen (Saha et al. 2021), resulting in the loss of N as N_2O or N_2 (Santos et al. 2021). However, the low temperatures during the data collection period in this study (Fig. S1.1) suggest that denitrification may have been reduced due to low microbial activity (Renault and Sierra 1994). This indicates that immobilization may have played a more dominant role in reducing leaching, particularly during the winter months. Additionally, compost may have decreased N availability by binding N onto the phenolic or humified sites in the compost through chemical or physical reactions (Choi et al. 2001). While it is possible that a portion of the unrecovered fertilizer N was assimilated into the root biomass, it should be noted that we only measured aboveground biomass-N. Nonetheless, this contribution is expected to be minimal (Choi et al. 2001).

1.5. Conclusion

The use of organic amendments, such as food waste and green waste composts, along with reduced fertilizer inputs, can maintain or even improve crop yields while promoting sustainability through organic waste landfill reduction. Our findings suggest that compost application may lead to temporary fertilizer N immobilization, with potentially greater impacts from FW compost than GW compost, which can have several effects on soil N cycling, including a potential priming effect on soil N mineralization and accumulation, an effect on compost N mineralization, an immobilization effect on fertilizer N, and an overall substitution effect in the N pool. Importantly, FW and GW composts tended to reduce fertilizer true NUE, suggesting that these composts may serve as a source of N and supplement fertilizer N requirements. These observations have important implications for fertilizer N guidelines and waste management policy and infrastructure design. Furthermore, the study highlights the need for future research to investigate longer study periods, N_2O emissions, other compost and fertilizer application rates, and co-composts with higher food waste content to further refine our understanding of the effects of organic amendments on soil N cycling and crop productivity.

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1.7. Supplementary information

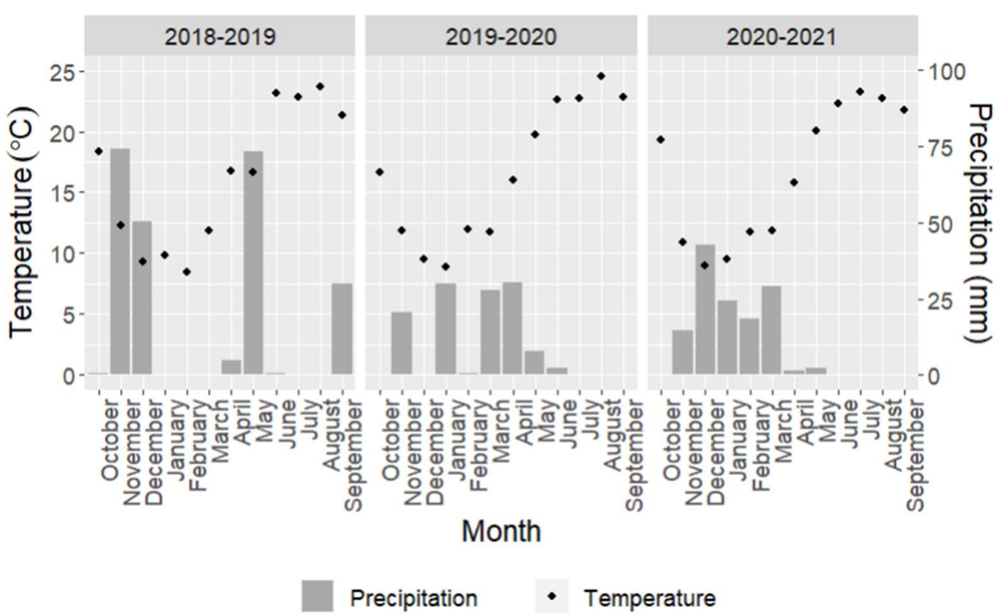


Figure S1.1. Climate conditions of monthly temperature and precipitation during the three years of experimentation. Data is from the Davis, California automated weather station and was acquired through the California Irrigation Management Information System (CIMIS).

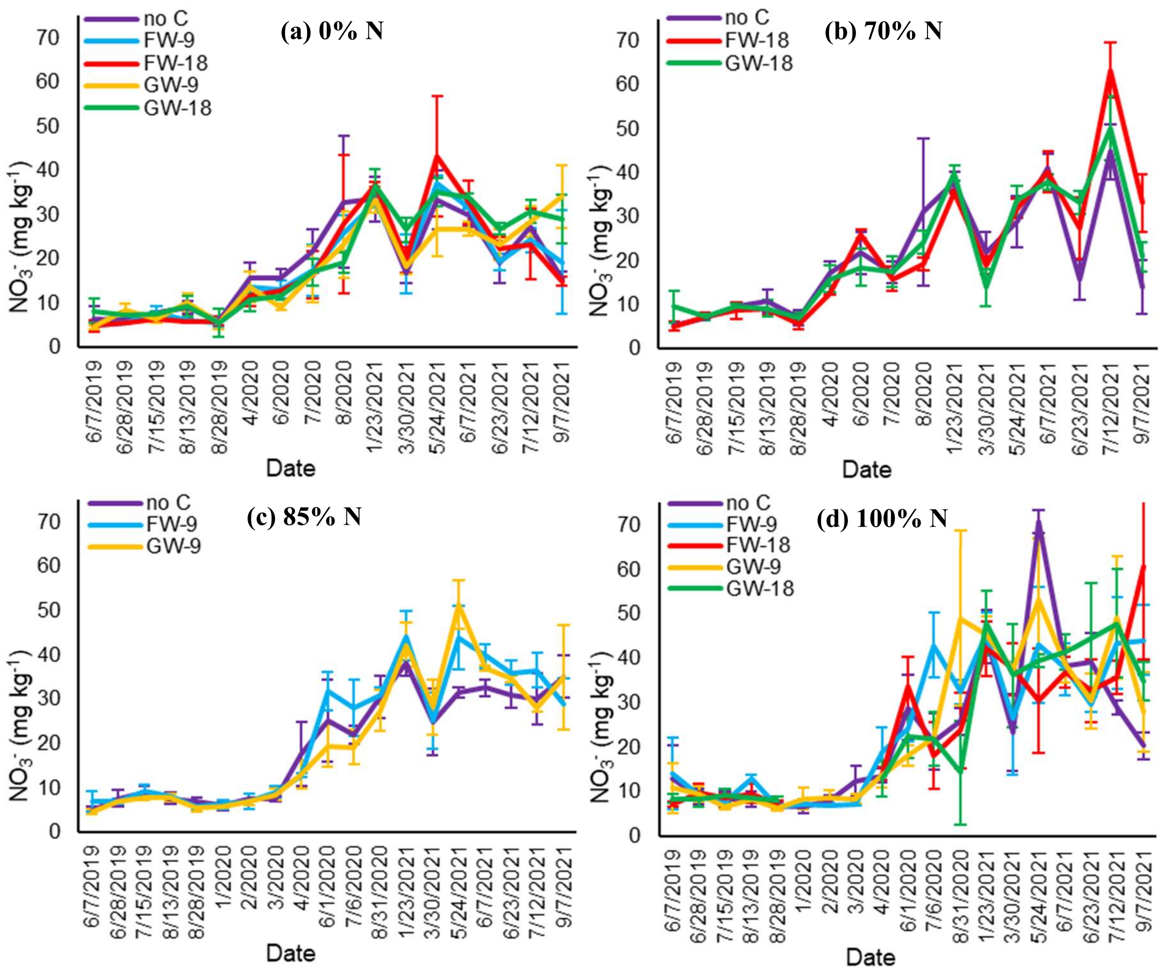


Figure S1.2. NO_3^- in the 0-15 cm soil layer in treatment plots that received (a) 0% N, (b) 70% N, (c) 85% N, and (d) 100% N. The line bars represent standard error (n = 3).

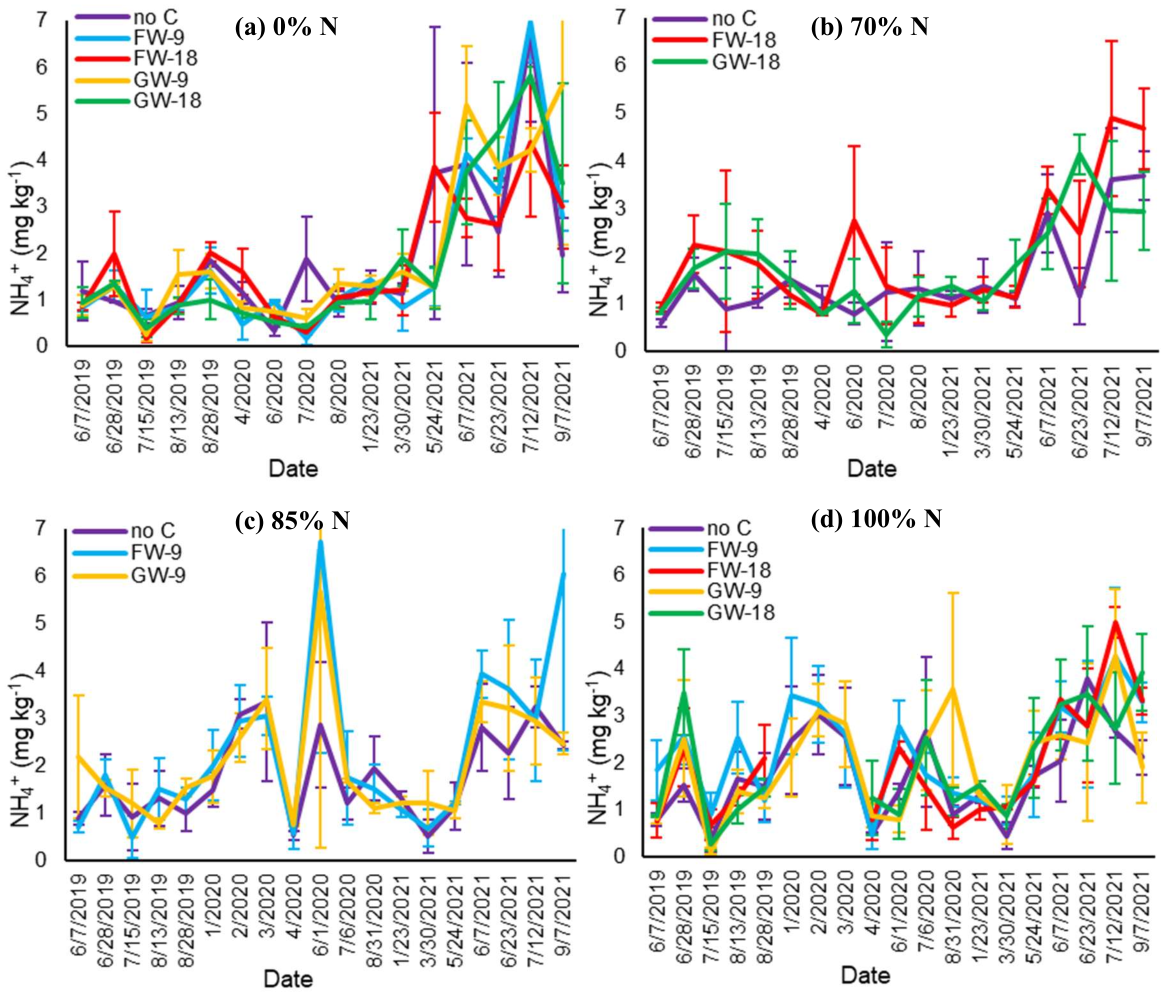


Figure S1.3. NH_4^+ in the 0-15 cm soil layer in treatment plots that received (a) 0% N, (b) 70% N, (c) 85% N, and (d) 100% N. The line bars represent standard error (n = 3).

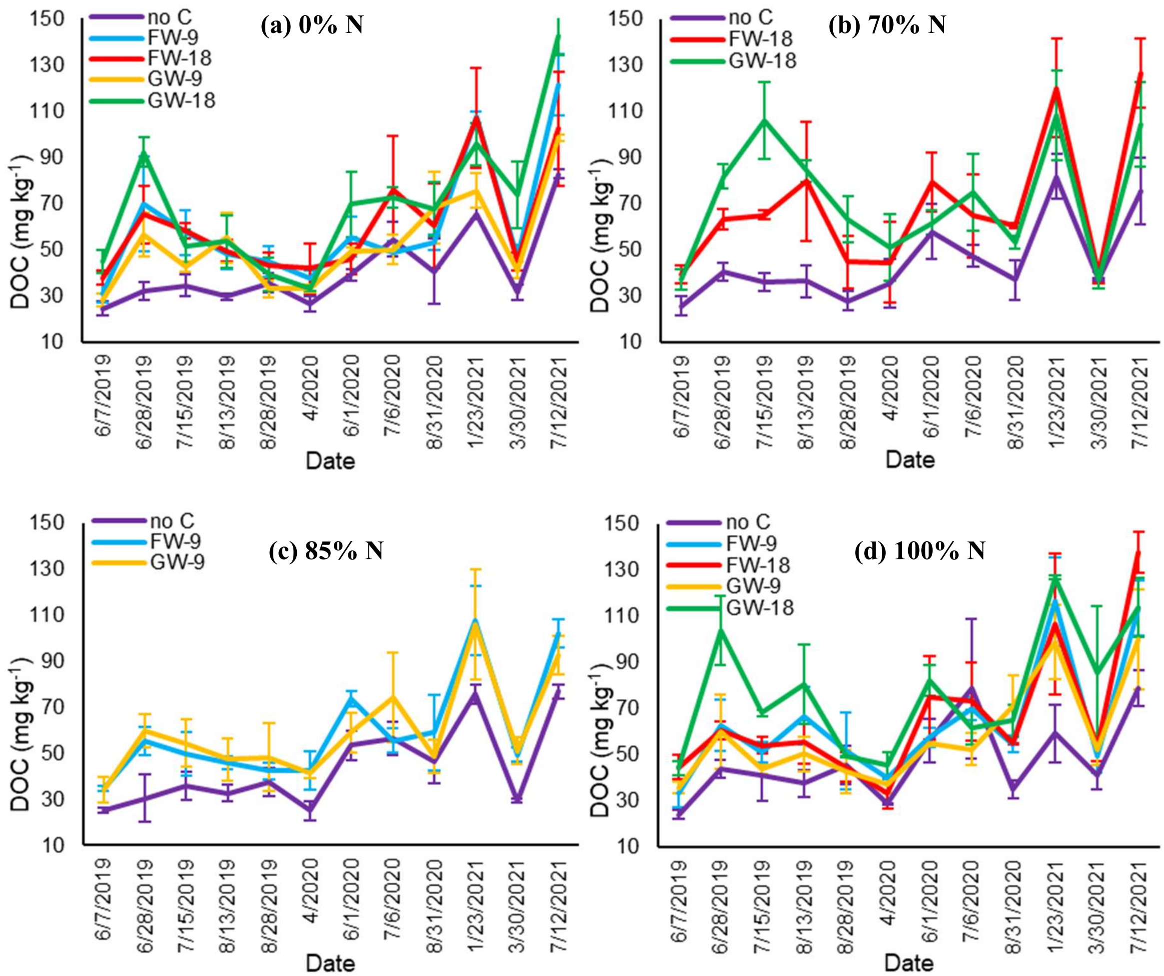


Figure S1.4. Dissolved organic carbon (DOC) in the 0-15 cm soil layer in treatment plots that received (a) 0% N, (b) 70% N, (c) 85% N, and (d) 100% N. The line bars represent standard error (n = 3).

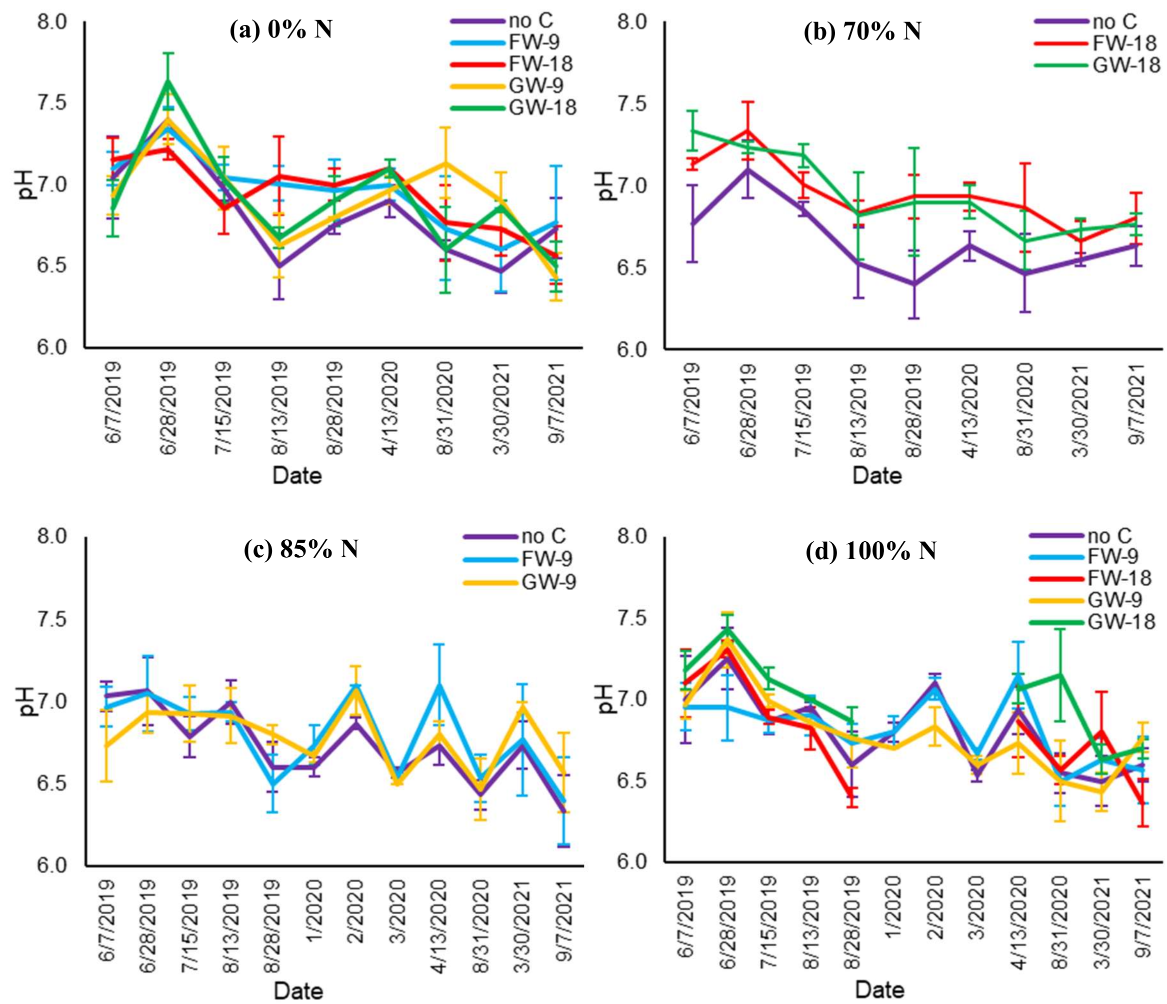


Figure S1.5. Soil pH in the 0-15 cm soil layer in treatment plots that received (a) 0% N, (b) 70% N, (c) 85% N, and (d) 100% N. The line bars represent standard error (n = 3).

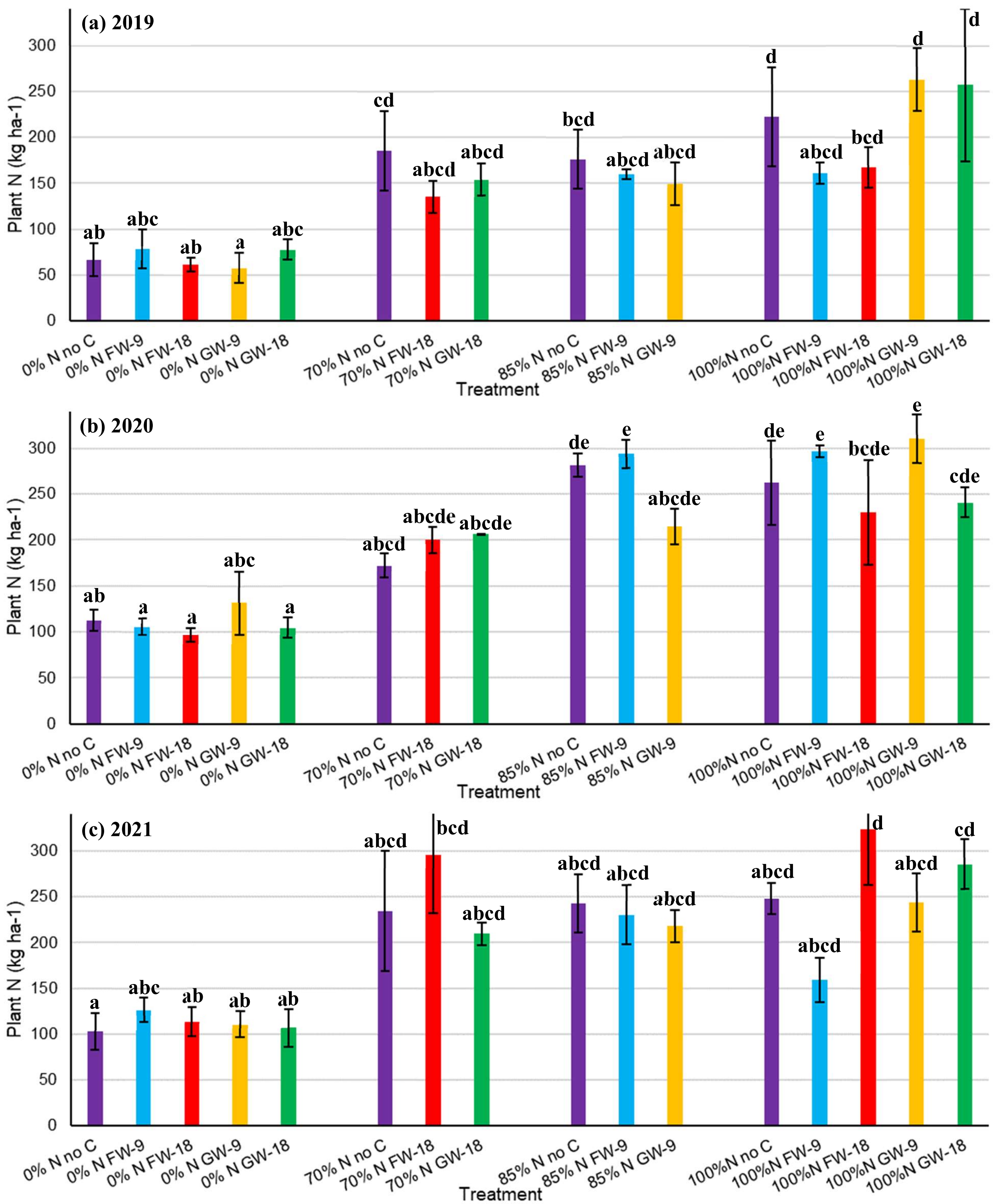


Figure S1.6. Total crop N uptake (a) 2019, (b) 2020, and (c) 2021 for the two compost types (FW and GW) at three application rates (no compost, 9 t ha^{-1}, and 18 t ha^{-1}), and four N levels (0%, 70%, 85%, and 100%) of the recommended amount. The line bars represent standard error (n = 3). Letters indicate significant differences in plant N among treatments at a statistical significance level of alpha = 0.05.

Chapter 2: Compost as a climate-smart agricultural practice for greenhouse gas emissions and soil carbon sequestration

2.1. Introduction

Agricultural soils are of interest for climate change mitigation through management practices that could reduce greenhouse gas (GHG) emissions and increase carbon (C) storage. Soils are the largest reservoir of C in the terrestrial biosphere, containing more C than vegetation and the atmosphere (IPCC 2000), and small changes of even a few percent in global soil organic C (SOC) stocks could lead to proportionally large contributions to the global soil C sink (Paustian et al. 2016). However, agricultural land use leads to SOC loss on average (Sanderman et al., 2017), but this loss has potential for reversal through soil C sequestration practices, such as the use of organic soil amendments like compost (Fabrizio et al. 2009; Zhang et al. 2012; Tautges et al. 2019). Globally, agriculture contributes to 50.63% and 41.8% of anthropogenic emitted methane (CH_4) and nitrous oxide (N_2O), respectively (Denman et al. 2007; Karakurt et al. 2012). Emissions are important to quantify because N_2O is 298 and CH_4 is 25 times the global warming potential (GWP) of carbon dioxide (CO_2) when considered on a 100-year time scale (IPCC 2007). However, more research is needed to understand whether cultivated soils amended with compost are sinks for N and C or sources of N_2O, CH_4, and CO_2.

The application of food waste (FW) compost as a climate-smart agricultural strategy is increasingly recognized alongside the established green waste (GW) compost industry (Levis et al. 2010; Oviedo-Ocaña et al. 2019). FW compost production not only diverts waste from landfills, potentially reducing GHG emissions there, but also holds promise for lowering soil GHG emissions in agricultural use (Schott et al. 2016; Nascimento et al. 2017). However, studies measuring the long-term effects of agricultural management on non-CO_2 GHG emissions, especially in regions like California, remain limited (Suddick et al. 2010). Composts are generally known to influence N_2O emissions, potentially by promoting soil N immobilization and increasing SOC (Wright et al. 2008). The former can reduce N_2O emissions by limiting substrates available for ammonia oxidation and heterotrophic denitrification, while the latter might increase emissions by stimulating microbial activity. Compared to GW compost, FW compost is anticipated to enhance soil nutrient status, but there is a lack of understanding of transformations and bioavailability of C and N in soils following FW and GW compost application (Sullivan et al. 2002; O'Connor et al.

2021). Compost N and fertilizer N are reported to be additive components to the plant-available N supply, with different compost application rates in combination with different fertilizer N rates leading to various soil N contents (Sullivan et al. 2002; Kovács et al. 2014). Given the varying outcomes reported in literature, further research is crucial for understanding how different compost types and N fertilizer rates affect N_2O emissions, aiming to develop best management practices that mitigate climate change without compromising the agronomic performance of cropping systems.

Reducing N_2O emissions fundamentally involves minimizing excess mineral N in soil pore water (McSwiney and Robertson 2005). Composts may help achieve this GHG reduction, as their slow N release can more closely align nutrient supply with crop demand compared to the rapid availability of inorganic N from synthetic fertilizers. N_2O emissions from soils receiving organic amendments are primarily driven by inorganic N released from the mineralization of organic materials. These emissions tend to increase with higher initial soluble N content and lower C:N ratios in the amendment (Huang et al. 2004; Santos et al. 2021).

Methane production in soils occurs as organic matter decomposes under oxygen (O_2)-depleted conditions, facilitating denitrification and methanogenesis (Hossain et al. 2017). Adding FW and GW compost might increase SOC and act as CH_4 sinks (Santos et al. 2021). Nevertheless, the potential C offsets from increased SOC can be temporary and reversible, while those generated by decreased N_2O emissions are permanent (De Gryze et al. 2009). Both N_2O and CH_4 production are influenced by soil moisture and O_2 levels (Signor and Cerri 2013), and composts like FW and GW can improve soil water infiltration and aeration (Diacono and Montemurro 2010), possibly reducing emissions. These varied outcomes highlight the need for further research to determine the effectiveness of FW and GW compost-amended soils in reducing GHG emissions and abating climate change.

The application of compost to soils not only holds the potential for enhanced C sequestration through improved soil aggregation and OC accumulation (Suddick et al. 2010), but may also influence nematode populations and diversity as an indirect effect of compost increasing soil bacterial abundance (Shi et al. 2023). Nematodes, as the most abundant soil invertebrates, play a crucial role in soil food webs and significantly contribute to C decomposition and sequestration through their predation on bacterial communities (Neher 2001; Grandy et al. 2016; Shi et al. 2023). Therefore, investigating nematode populations and diversity, such as bacterial- and fungal-feeding genera, may offer insight into how compost use affects the soil food web, and therefore also a potential biological mechanism for C sequestration. Subsoils exhibit greater and more enduring C storage capacity than surface soils (Rumpel et al. 2012), however most studies have focused on surface soils due to ease of sampling and quicker response to the changes of management. However, this approach overlooks significant C stocks in deeper soil layers, potentially leading to inaccuracies in global C budgets. For example, studies on C flux measurements and modelling often consider only the top 20 or 30 cm of soil. Yet, this layer contains only 30-50% of the C stocks within the top meter of soil (Jobbágy and Jackson 2000; Rumpel et al. 2012). Consequently, global C budgets could be either overestimated or underestimated if the full vertical distribution of SOC is not taken into account. Furthermore, the impact of composts on SOC content must also be weighed against their alternative fates; for FW and GW, this often means ending up in landfills where their decomposition releases C emitted as GHGs (Powlson et al. 2011). Compost C sequestration in soil following field application can be evaluated based on $\delta^{13}C$. Mature composts have lower $\delta^{13}C$ values than fresh feedstock, as the microbial activities depleted ^{12}C and enriched ^{13}C in the compost after the decomposing process. Since the decomposing process of organic material during the composting period is much shorter than the formation of SOC, the $\delta^{13}C$ value of the finished compost is expected to be significantly lower than SOC. Thus the $\delta^{13}C$ can be used as a tracer technique to characterize the dynamics of 'native' and 'new' SOC (Gerzabek et al. 2001; Lynch et al. 2006; Nguyen-Sy et al. 2020; You et al. 2021).

In this study, we investigated N_2O, CH_4, and CO_2 emissions, soil C up to a depth of 90 cm, and soil nematode populations and diversity at a conventional processing tomato site with silty clay loam soil in the Sacramento Valley, California. The field experiments, conducted over three years, included compost application and high frequency GHG measurements during the final two years, with annual soil C assessments. Treatments involved three different application rates of FW and GW composts in combination with a range of fertilizer N levels. The objective of this study was to evaluate the potential of compost amendments in agricultural soils for GHG emission mitigation and soil C storage. We analyzed N_2O, CH_4, and CO_2 fluxes, annual cumulative emissions, N_2O emission factors (EFs), soil C and $\delta^{13}C$ across four depth ranges down to 90 cm, and nematode population and diversity. This study provides insight into the use of compost, potentially in tandem with reduced fertilizer, as a climate-smart agricultural practice for adapting to and mitigating climate change.

2.2. Materials and methods

2.2.1. Site description and agronomic management

Field experiments were conducted between 2019 and 2021 at the University of California-Davis Russell Ranch Sustainable Agriculture research site in fields under tomato cultivation. This site has been in a tomato-corn rotation from 2013 to 2018 with corn in even years and tomato in odd years. Previous management involved conventional industry practices (e.g., fertilizer NPK only and conventional tillage) with subsurface drip irrigation with fertilizers (i.e., fertigation). The soil is classified as Rincon silty clay loam, a fine smectitic, thermic Mollic Haploxeralf, according to the United States Department of Agriculture, National Cooperative Soil Survey. Baseline soil characteristics are described in Table 2.1. Tomato crops were transplanted on May 1, 2019; April 21, 2020; and April 27, 2021. Following each harvest, crop residues were incorporated into the soil. This region has a Mediterranean climate with a short winter rainy season from November to March, and a longer hot, dry period of little to no rain from April to October during which the growing season occurs.

Table 2.1. Baseline soil characteristics before the setup of field trials.

Depth (cm)	Total N ($g\ kg^{-1}$)	Total C ($g\ kg^{-1}$)	pH (H_2O 1: 1)	DOC ($mg\ kg^{-1}$)	NH_4^+ ($mg\ kg^{-1}$)	NO_3^- ($mg\ kg^{-1}$)	Bulk density ($Mg\ m^{-3}$)
0-15	1.47	15.2	7.13	17.2	1.13	11.0	1.43
15-30	1.26	14.1	7.43	13.8	1.09	4.83	1.68

2.2.2. Field experimental design and fertility management

A split-plot randomized design with sixteen treatments and three blocks (replicates) was implemented for three consecutive years. Treatments included two compost types (FW or GW) × three compost rates (0, 9 t ha^{-1}, or 18 t ha^{-1}) × two fertilizer N levels (0% or 100% of recommended N rate). Additional treatments consisted of different compost application rates combined with a corresponding reduction in N rates, which were selected by replacing fertilizer N with compost N: 85% of recommended N rate × compost (FW or GW at the rate of 0 or 9 t ha^{-1}) and 70% of recommended N rate × compost (FW or GW at 0 or 18 t ha^{-1}). The fertilizer N rates were chosen to compare a range of rates from a no-fertilizer control to the full recommended rate, including fertilizer N reductions to 85%

and 70% because N recommendations and budgets often overestimate fertilizer N requirements and lower rates have been reported to be sufficient for tomato crops (Geisseler et al. 2020). The compost treatments were chosen to determine if compost N can replace some synthetic fertilizer N, and 9 t ha^{-1} (4 ton $acre^{-1}$) is the recommended rate of dry weight compost for annual crops like tomatoes in California (Gravuer and Gunasekara 2016), which was the compost rate combined with the 15% fertilizer N reduction rate, while double the recommended compost application rate (18 t ha^{-1} (8 ton $acre^{-1}$)) was combined with the 30% fertilizer N reduction rate. The amount of N applied in each compost treatment in 2019 was as follows: 9 t FW ha^{-1} = 120 kg N ha^{-1}, 9 t GW ha^{-1} = 113 kg N ha^{-1}, 18 t FW ha^{-1} = 240, 18 t GW ha^{-1} = 226, and in 2020 was as follows: 9 t FW ha^{-1} = 150 kg N ha^{-1}, 9 t GW ha^{-1} = 153 kg N ha^{-1}, 18 t FW ha^{-1} = 300 kg N ha^{-1}, 18 t GW ha^{-1} = 306 kg N ha^{-1}.The size of each experimental plot was 4.6 m × 6.1 m, with fertilizer levels as main plots and compost types and rates as split factors. Every plot received a preplant application of NPK 8-24-6 starter fertilizer at 28 kg N ha^{-1}. The local recommended in-season N fertilization rate was 202 kg N ha^{-1} (100% N), with application every 2 to 3 weeks during the growing period through subsurface irrigation drip lines (UAN 32 at a rate of 34-56 kg N ha^{-1} each event). Composts were commercially purchased, and hand spread evenly on the soil surface and disked in with standard equipment to a depth of 10-15 cm in the spring of 2019 and the fall of 2019, 2020, and 2021. The FW compost was 5% food waste and 95% urban yard waste. The GW compost was 100% urban yard waste. These two types of compost were those available on the market at the time. See Table 2.2 for compost characteristics.

Table 2.2. Characteristics of compost used each year.

Year	Compost	pH	Total C (g kg^{-1})	Total N (g kg^{-1})	C: N	NH_4^+ (mg kg^{-1})	NO_3^- (mg kg^{-1})	$\delta^{13}C$ (‰)
2019	FW	8.08	285.3	10.38	27.49	3.32	119.13	-27.54
	GW	8.21	266.7	13.83	19.28	28.71	103.10	-27.46
2020	FW	7.90	282.6	13.32	21.22	607.94	8.59	-27.65
	GW	7.23	312.3	12.52	24.94	32.66	301.32	-27.73
2021	FW	7.20	311.8	16.67	18.70	106.97	0.67	-27.58
	GW	6.97	298.4	16.97	17.58	227.59	1.11	-27.81

2.2.3. Gas sampling and analysis

GHG emissions were evaluated in seven out of the sixteen field treatments due to resource limitations, including 0% N × no compost, 85% N × no compost, 85% N × FW 9 t ha^{-1}, 85% N × GW 9 t ha^{-1}, 100% N × no compost, 100% N × FW 9 t ha^{-1}, and 100% N × GW 9 t ha^{-1}, starting in November 2019 which was after the second compost treatment application in the field. These specific treatments were selected for GHG measurements because the compost application rate of 9 t ha^{-1} represents the more common fertility management scenario for growers (Gravuer and Gunasekara 2016), and this compost rate was combined with the 85% N fertilizer rate but not the 70% N fertilizer rate in the full sixteen-treatment experimental design. GHG measurements were conducted using a static closed chamber method, following the USDA-ARS GRACEnet protocol (Parkin and Venterea 2010). A sample of the chamber headspace air was manually pulled through a septum in the chamber into a 20 mL syringe at 0, 20, 40, and 60 minutes after chamber closure, and injected into a pre-evacuated exetainer with a septum. N_2O, CH_4, and CO_2 concentrations were determined using gas chromatography (Shimadzu Model 2014). Fluxes were calculated based on changes in headspace gas concentrations using a linear least-squares fit to the four time points of concentration for each plot. Fluxes were adjusted for chamber volume and internal air temperature. Chamber air temperature was measured at each gas sampling event and time point using a thermocouple in the chamber headspace connected to a digital thermometer (Fluke Corp., Everett, WA). Annual cumulative emissions were calculated by linear interpolation of fluxes between two sampling events and integrated over November 2019 to October 2020 (2019-2020) and over November 2020 to October 2021 (2020-2021) (Rothardt et al. 2021). N_2O EFs were calculated for 2019-2020 and 2020-2021 by first background correcting cumulative emissions as the difference between treatments and no N input (0% N × no compost), then dividing background corrected cumulative emissions by the total compost and fertilizer N applied.

2.2.4. Soil sampling and analysis

Soil samples were collected from four composite borings for each treatment plot from the 0-15 cm soil layer before and after fertigation events and approximately monthly during the remainder of each year to cover the gas sampling timeframe. Additional soil samples were collected from each treatment plot after each year's harvest, before tomato transplant, and at the end of the experimental period to a depth of 90 cm using a PN150 JMC Environmentalist's Sub-Soil Probe. These deep soil cores were separated into 0-15 cm, 15-30 cm, 30-60 cm, and 60-90 cm soil layers.

Bulk density (BD) was measured by collecting 5 cm diameter × 6.7 cm long soil cores in the 0-15, 15-30, 30-60, and 60-90 cm soil layers at the same times as deep soil core sampling, followed by drying of the cores at 105ºC. Bulk density was calculated as the mass of dry soil collected in the core divided by the cylinder volume.

Soil samples were extracted with 0.5 M K_2SO_4 (4:1 extractant volume to soil mass ratio). The soil extracts were analyzed colorimetrically for NH_4^+ and NO_3^- using a Shimadzu spectrophotometer (Model UV-Mini 1240) (Forster 1995; Doane and Horwáth 2003). DOC was measured in extracts using a total organic carbon (TOC) ultra-violet (UV)-persulfate oxidation analyzer (Model Phoenix 8000, Teledyne Tekmar, Mason, OH). Soil gravimetric water content (GWC) was calculated by comparing the field-moist and oven-dry (105ºC) mass of soil samples and these values were used for calculating soil NH_4^+, NO_3^-, and DOC content. Total C in soils were measured by an elemental C and N analyzer (Costech Analytical Technologies Inc., Valencia, CA) using the dry combustion method (Dumas, 1848) after grinding air-dried representative soil samples to a fine powder. The soil pH was measured in a soil slurry (1:1 soil to water ratio) with a pH meter (Thermo Scientific Orion 9156BNWP Combination pH Electrode, Taylor Scientific, St. Louis, MO).

Soil samples from 90 cm cores collected in the 0% N level main plots after the final tomato harvest in September 2021 were submitted to the University of California-Davis Stable Isotope Facility for $\delta^{13}C$ analysis. Soils were analyzed for ^{13}C isotopes using an Elementar Vario EL Cube or Micro Cube elemental analyzer (Elementar Analysensysteme GmbH, Hanau, Germany) interfaced to an Isoprime VisION IRMS (Elementar UK Ltd, Cheadle, UK). The final $\delta^{13}C$ values were expressed relative to international standards VPDB (Vienna Pee Dee Belemnite).

At the end of the experiment in January 2022, soil samples were collected from the 0-15 cm soil layer of the same seven treatment plots that gas samples were collected from and sent to the University of California-Davis Soil Ecology and Pest Management Lab to analyze nematode population and diversity. Nematodes were counted and identified to genus.

2.2.5. Statistical analysis

Statistical analyses were conducted using linear mixed models in R program. For the investigated metrics of N_2O, CH_4, and CO_2 fluxes; cumulative N_2O, CH_4, and CO_2 emissions; soil C; and nematode genus, compost and N fertilizer level and their interactions were treated as fixed effects and replicates (blocks) were treated as random effects. For $\delta^{13}C$, compost and soil depth were treated as fixed effects, and blocks and the interaction of blocks × compost were treated as random effects. Soil depth was also a fixed effect for soil C content analysis. Likelihood ratio tests (LRT) were performed for lmer models using the "anova" function to ensure no additional random effect interactions were needed for the best fit and appropriate complexity of models. Analysis of variance (ANOVA) were performed for cumulative GHG emissions, $\delta^{13}C$, and nematode metrics to compare treatment effects and to determine if effects were significant ($p<0.05$). Data was transformed as needed to normalize distributions and homogenize variances prior to statistical analyses. N_2O, CH_4, and CO_2 flux data did not meet the assumption of normality, including after data transformations, therefore permutational multivariate analysis of variance (PERMANOVA) was performed using the "adonis2" function and "Euclidean" method, which is more appropriate for datasets containing both positive and negative values. Correlation analysis was performed using the "cor.test" function and Pearson's correlation coefficient for continuous variables (GWC, NH_4^+, NO_3^-, DOC, and pH) that were measured along with fluxes to test for correlation among soil variables and N_2O, CH_4, and CO_2 fluxes.

2.3. Results

2.3.1. Environmental conditions

Climatic data, including monthly temperature and precipitation over the two years of GHG measurements, are shown in Fig. 2.1. This data was obtained from the Davis, California automated weather station and was acquired through the California Irrigation Management Information System (CIMIS). The temperature and precipitation trends were similar across both years. The observed patterns align with typical Mediterranean climatic conditions, characterized by a rainy and cool winter from November to March, and a dry, hot summer from April to October.

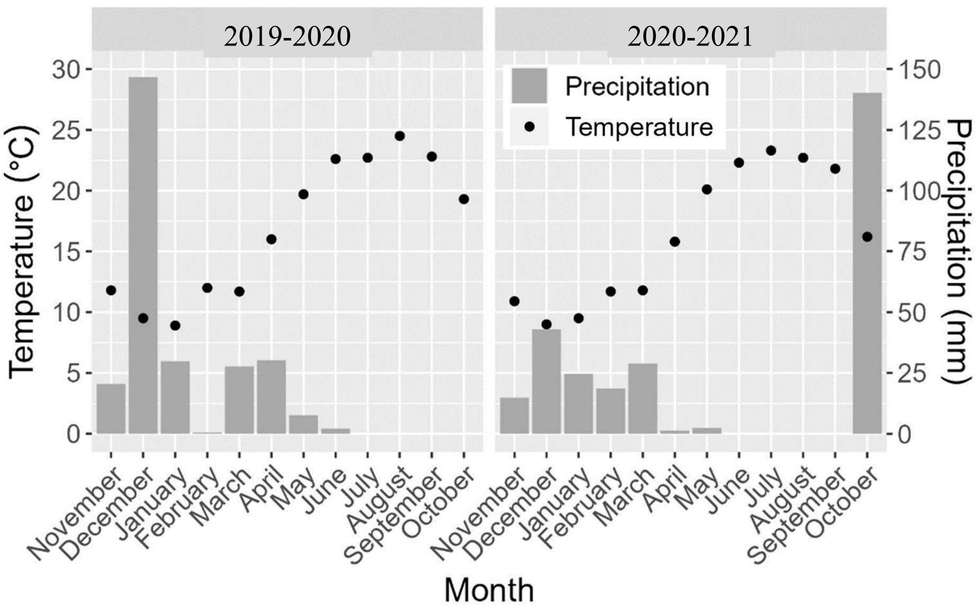


Figure 2.1. Monthly temperature and precipitation during the two years (2019-2020 and 2020-2021) of GHG measurements. Data is from the Davis, California automated weather station and was acquired through the California Irrigation Management Information System (CIMIS).

2.3.2. Gas emissions

Nitrous oxide (N_2O) fluxes observed over the two-year field experiment are shown in Fig. 2.2. As expected, N_2O fluxes showed a slight increase following precipitation events and a more notable increase after fertigation events. When comparing no-compost and compost treatments across the three fertilizer N levels, the positive fluxes increased in the order of 0% N, 85% N, and 100% N, noting that no composts were applied at the 0% N level. Among the compost treatments, their effects on N_2O flux were variable, without a clear trend. However, during the 2020 growing season from May to August at the 100% N level, the no-compost treatment typically exhibited the highest flux, followed by GW compost, with FW compost generally showing the lowest flux (Fig. 2.2d). It is important to note that these data points displayed a high standard error relative to the rest of the dataset.

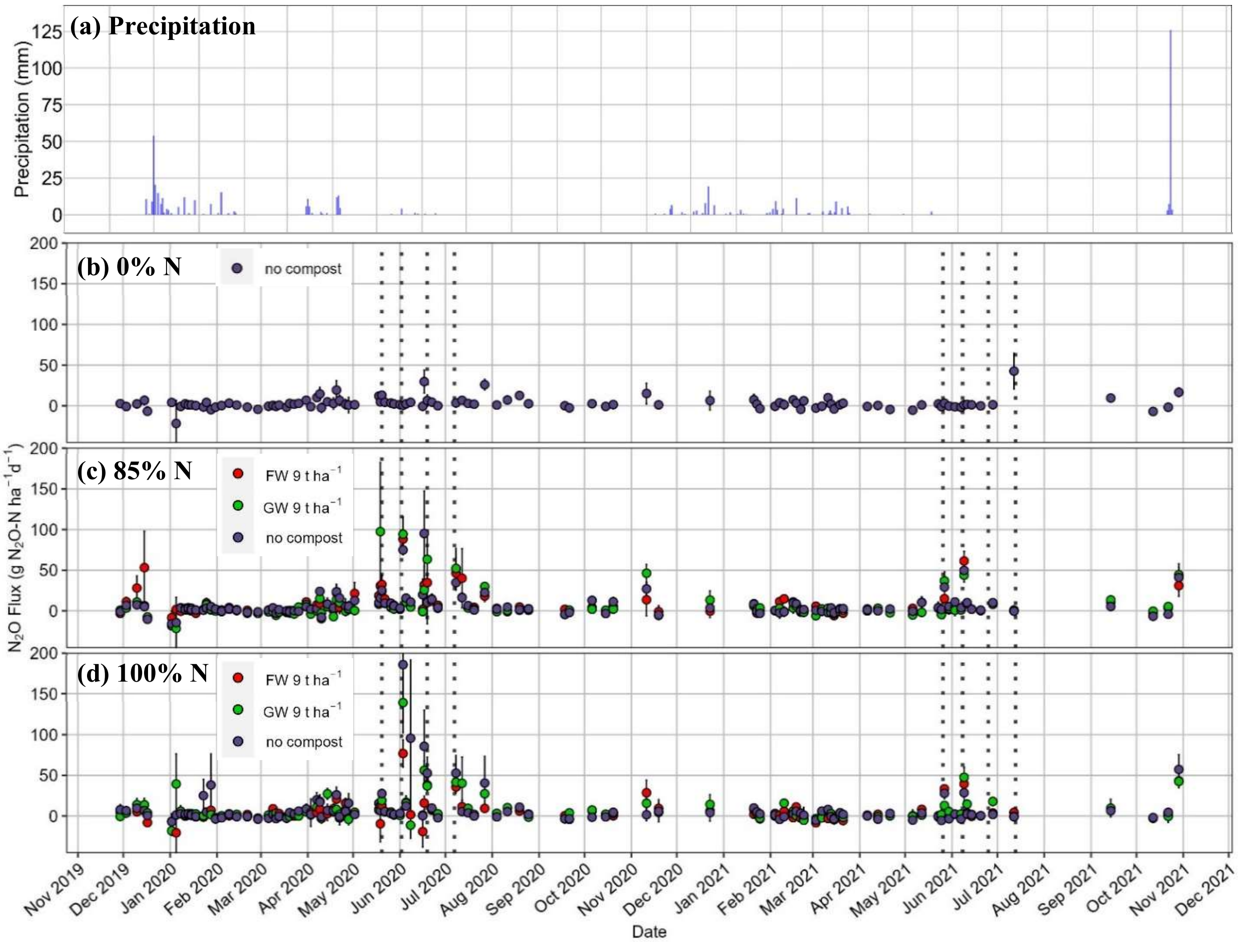


Figure 2.2. (a) Precipitation and nitrous oxide (N_2O) fluxes over the two-year experimental period for the two compost types (FW and GW) at two application rates (no compost or 9 t ha^{-1}), and at the fertilizer levels of (b) 0% N, (c) 85% N, and (d) 100% N. Blue bars show daily precipitation. Grey dotted lines represent fertigation events during the growing seasons. The black line bars represent flux standard error (n = 3).

Annual cumulative N_2O emissions for the 2019-2020 and 2020-2021 periods of the study were calculated using daily N_2O fluxes (Fig. 2.3). Across both years, no statistically significant effects of compost treatment were observed on cumulative N_2O emissions. In 2019-2020, the treatment of 0% N level without compost exhibited lower emissions compared to the 85% and 100% N levels when no compost was applied. However, this trend was not observed in 2020-2021.

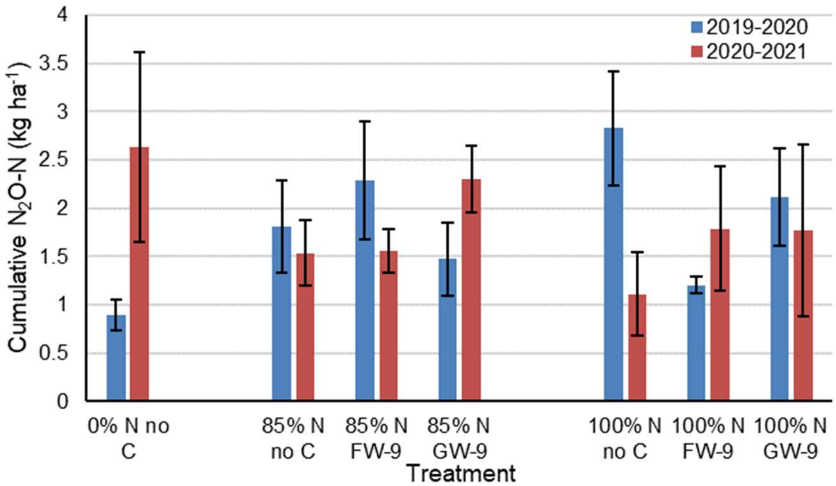


Figure 2.3. Annual cumulative N_2O emissions for years 2019-2020 and 2020-2021 of GHG measurements for the two compost types (FW and GW) at two application rates (no C = no compost and 9 = 9 t ha^{-1}), and three N levels (0%, 85%, and 100%) of the recommended amount. The line bars represent standard error (n = 3). No significant differences among treatments were observed either year at a statistical significance level of alpha = 0.05.

Annual cumulative N_2O emissions were used to calculate EFs as a percent of fertilizer N and compost N applied in each treatment and were corrected for background soil emissions by taking the difference between treatments and controls with no N input (Table 2.3). Because the cumulative N_2O emissions in the no-compost at the 0% N level control were greater than the other treatments in 2020-2021, background correcting emissions would lead to negative values so data for this year was omitted. In 2019-2020, EFs for all treatments were less than 1%. The no-compost controls at both the 85% and 100% N levels had higher EFs than compost treatments, and the 100% N level no-compost control had the highest EF (0.96%) out of all six treatments.

Table 2.3. Nitrous oxide (N_2O) EF values (means ± 1 SE) (%) for the 2019-2020 measurement period for the two compost types (FW and GW) at two application rates (no compost and 9 = 9 t ha^{-1}), and two N levels (85% and 100%) of the recommended amount.

Year	85% N no compost	85% N FW-9	85% N GW-9	100% N no compost	100% N FW-9	100% N GW-9
2019-2020	0.53 (0.31)	0.48 (0.18)	0.20 (0.09)	0.96 (0.32)	0.10 (0.03)	0.39 (0.12)

Methane (CH_4) fluxes measured throughout the experimental period are shown in Fig. 2.4. Due to the high variations among replicates, it is challenging to discern significant differences among compost treatments. Generally, CH_4 fluxes were low, with about half of the flux measurements being negative, suggesting soils might serve as a sink. However, the data did not show a consistent trend of positive or negative fluxes correlating with fertigation or precipitation events.

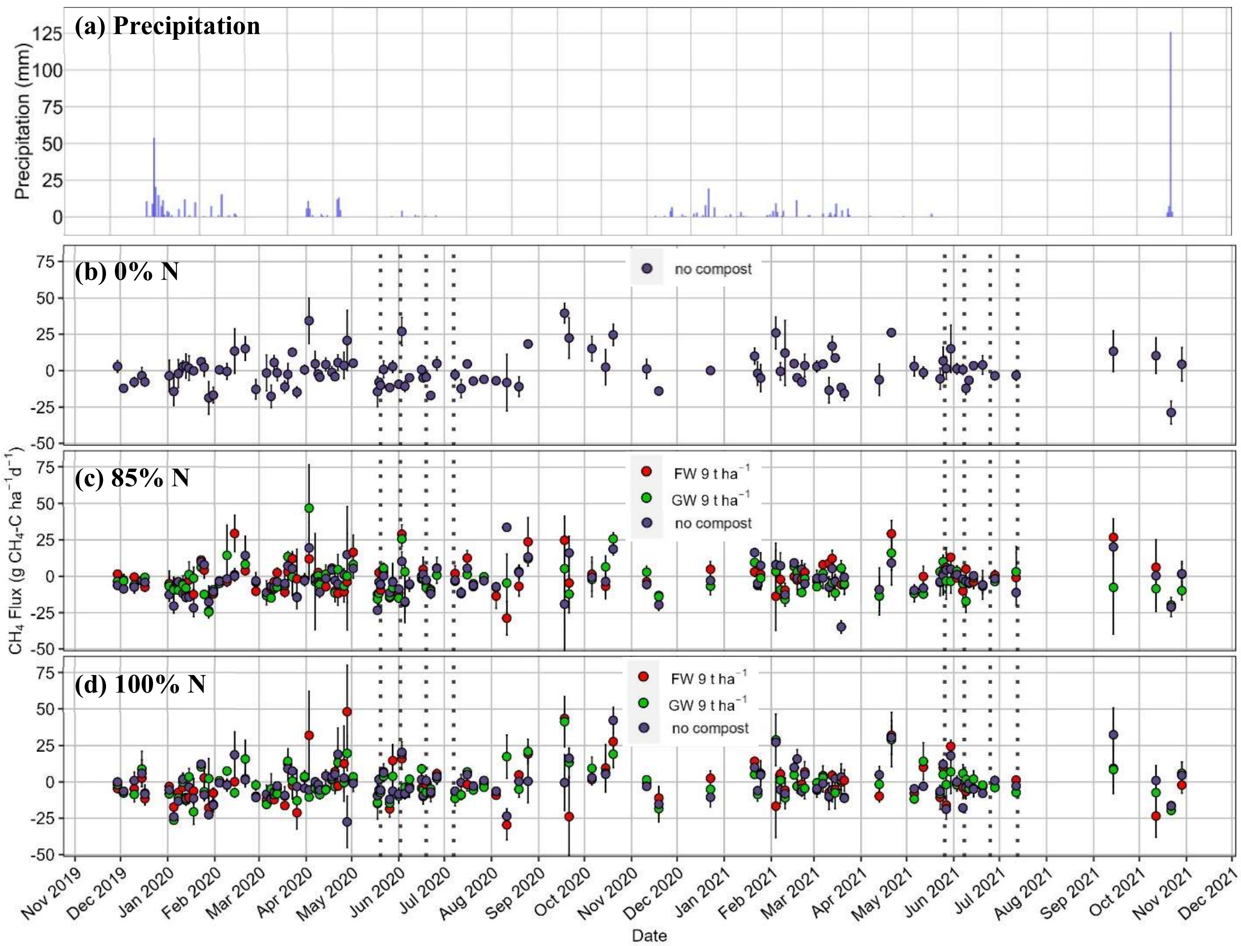


Figure 2.4. (a) Precipitation and methane (CH_4) fluxes over the two-year experimental period for the two compost types (FW and GW) at two application rates (no compost or 9 t ha^{-1}), and at the fertilizer levels of (b) 0% N, (c) 85% N, and (d) 100% N. Blue bars show daily precipitation. Grey dotted lines represent fertigation events during the growing seasons. The black line bars represent flux standard error (n = 3).

The influence of compost type and fertilizer N level on annual cumulative CH_4 emissions varied considerably (Fig. 2.5). However, due to the high variability in CH_4 flux measurements, the observed differences among treatments for cumulative emissions did not reach statistical significance in either the 2019-2020 or 2020-2021 periods.

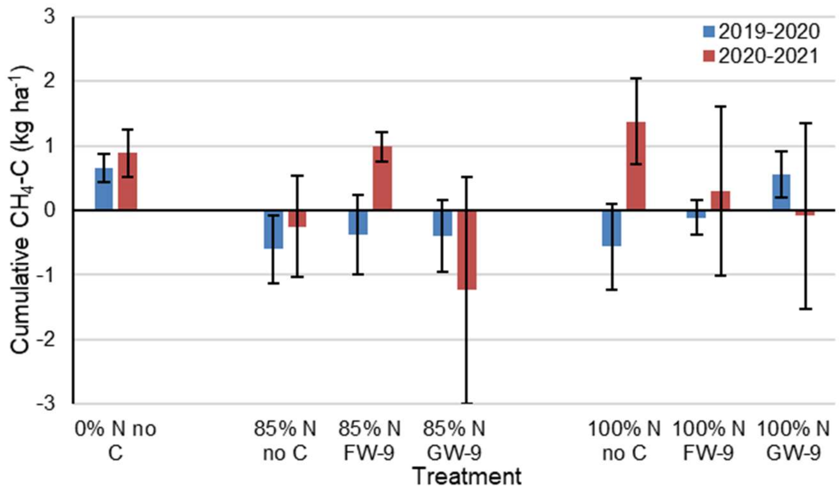


Figure 2.5. Annual cumulative CH_4 emissions for years 2019-2020 and 2020-2021 of GHG measurements for the two compost types (FW and GW) at two application rates (no C = no compost and 9 = 9 t ha^{-1}), and three N levels (0%, 85%, and 100%) of the recommended amount. The line bars represent standard error (n = 3). No significant differences among treatments were observed either year at a statistical significance level of alpha = 0.05.

Carbon dioxide (CO_2) fluxes throughout the study period are shown in Fig. 2.6. As anticipated, CO_2 fluxes generally increased following fertigation and precipitation events, with more notable rises during higher rainfall. When comparing the three fertilizer levels, CO_2 fluxes increased in the order of 0% N, 85% N, and 100% N. At the 85% N level, CO_2 fluxes were typically lowest in the no-compost treatments, followed by GW compost, while FW compost treatments exhibited the highest fluxes (Fig. 2.6c). In contrast, at the 100% N level, the lowest CO_2 fluxes were observed in FW compost treatments, followed by no compost, and GW compost treatments showed the highest fluxes (Fig. 2.6d).

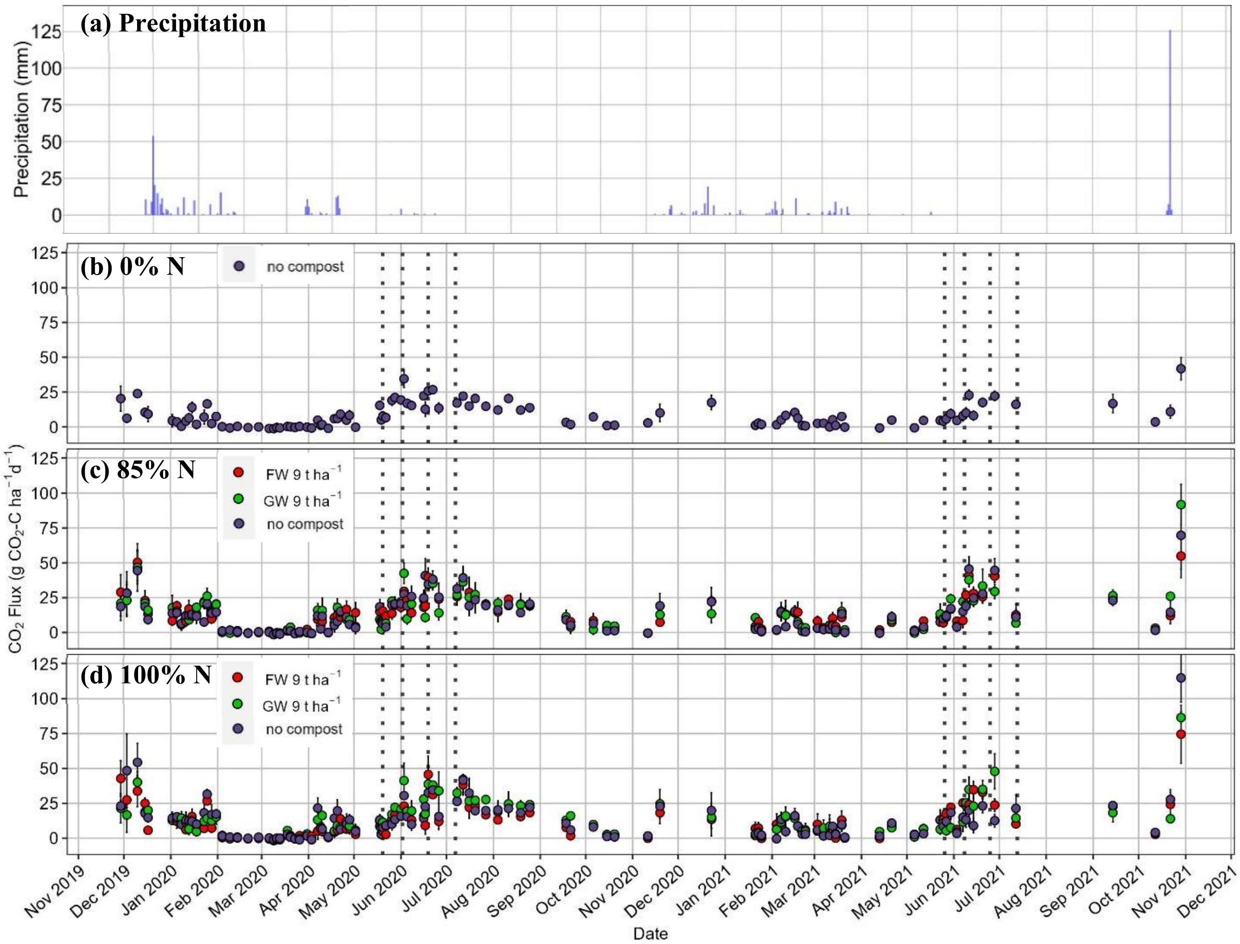


Figure 2.6. (a) Precipitation and carbon dioxide (CO_2) fluxes over the two-year experimental period for the two compost types (FW and GW) at two application rates (no compost or 9 t ha^{-1}), and at the fertilizer levels of (b) 0% N, (c) 85% N, and (d) 100% N. Blue bars show daily precipitation. Grey dotted lines represent fertigation events during the growing seasons. The black line bars represent flux standard error (n = 3).

Annual cumulative CO_2 emissions did not vary greatly between 2019-2020 and 2020-2021 periods, nor between compost treatments and no-compost controls (Fig. 2.7). In 2019-2020, FW compost at the 85% N level increased CO_2 emissions, whereas at the 100% N level, it decreased CO_2 emissions. CO_2 emissions from the GW compost treatments were similar to the no-compost controls at both fertilizer N levels. During the 2019-2020 period, the 0% N level without compost had lower CO_2 emissions than the 85% and 100% N levels with or without compost. However, this trend was not as pronounced in 2020-2021, and no statistically significant treatment effects were observed during this period.

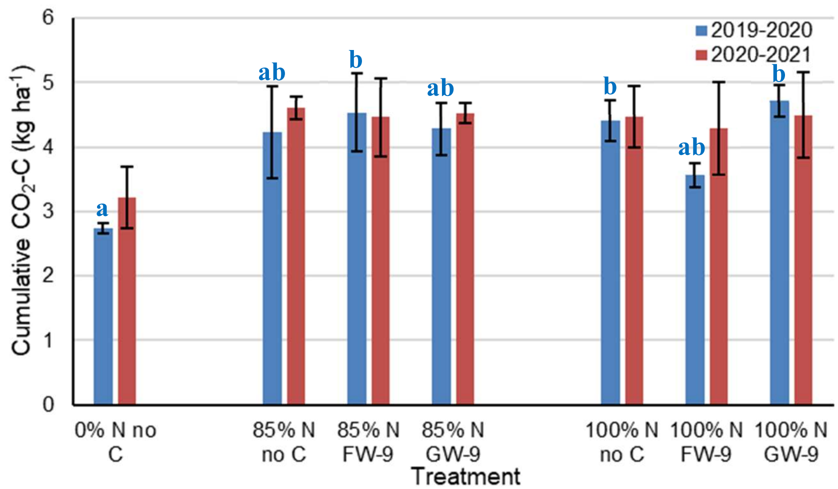


Figure 2.7. Annual cumulative CO_2 emissions for the years 2019-2020 and 2020-2021 of GHG measurements for the two compost types (FW and GW) at two application rates (no C = no compost and 9 = 9 t ha^{-1}), and three N levels (0%, 85%, and 100%) of the recommended amount. The line bars represent standard error (n = 3). Letters indicate significant differences in cumulative CO_2 among treatments in 2019-2020 at a statistical significance level of alpha = 0.05, and no statistically significant differences among treatments were observed in 2020-2021.

The effects of compost and N level treatments and their interactions on gas fluxes and cumulative emissions were assessed using PEMANOVA and ANOVA, respectively (Table 2.4). Compost significantly influenced CO_2 flux, but not N_2O or CH_4 fluxes. Fertilizer N level significantly affected the fluxes of all three gases (N_2O, CH_4, and CO_2). However, the interaction between compost and N level did not significantly change any gas fluxes or cumulative emissions. The only significant treatment effect on cumulative emissions was observed for the treatment factor of N level on CO_2.

Table 2.4. Effects of compost and fertilizer N level on soil gas fluxes and cumulative emissions. Pr (>F) values from PERMANOVA for gas fluxes and ANOVA for cumulative emissions.

	Flux			Cumulative emissions		
Source	N_2O	CH_4	CO_2	N_2O	CH_4	CO_2
Compost	0.618	0.733	0.010**	0.8795	0.6952	0.4288
N level	0.001***	0.020*	0.001***	0.9903	0.2777	<0.0001***
Compost × N level	0.072	0.589	0.427	0.6608	0.5045	0.2330

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

The correlations between N_2O, CH_4, and CO_2 fluxes and various soil variables measured concurrently with gas sampling throughout the experiment are summarized in Table 2.5. Soil GWC, NH_4^+, NO_3^-, DOC, and pH did not show significant correlations with N_2O fluxes. CH_4 fluxes were not significantly correlated with NH_4^+, NO_3^-, DOC, or pH, but exhibited a negative correlation with GWC. CO_2 fluxes were positively correlated with DOC and negatively with GWC, as expected. Additionally, there were inter-correlations among the gases: a positive correlation between N_2O and CO_2, a negative correlation between CO_2 and CH_4, and no significant correlation between N_2O and CH_4 (Table 2.5).

Table 2.5. Correlation matrix (r values) and statistical significance of correlations among the measured soil variables N_2O flux, CH_4 flux, CO_2 flux, GWC, NH_4^+, NO_3^-, DOC, and pH.

Variable	N_2O Flux	CH_4 Flux	CO_2 Flux	GWC	NH_4^+	NO_3^-	DOC	pH
N_2O Flux	1.000							
CH_4 Flux	0.036	1.000						
CO_2 Flux	0.371***	-0.052*	1.000					
GWC	-0.107	-0.122*	-0.388***	1.000				
NH_4^+	0.017	0.069	0.093	-0.213***	1.000			
NO_3^-	-0.061	0.042	0.298***	-0.498***	0.083	1.000		
DOC	-0.002	-0.055	0.276***	-0.458***	0.212**	0.618***	1.000	
pH	0.143	0.052	-0.344***	0.305***	-0.131	-0.400***	-0.165	1.000

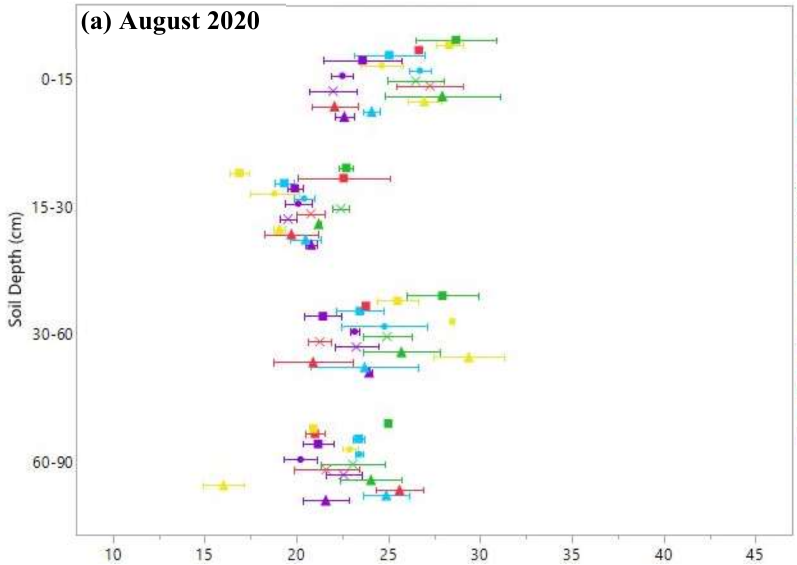
* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

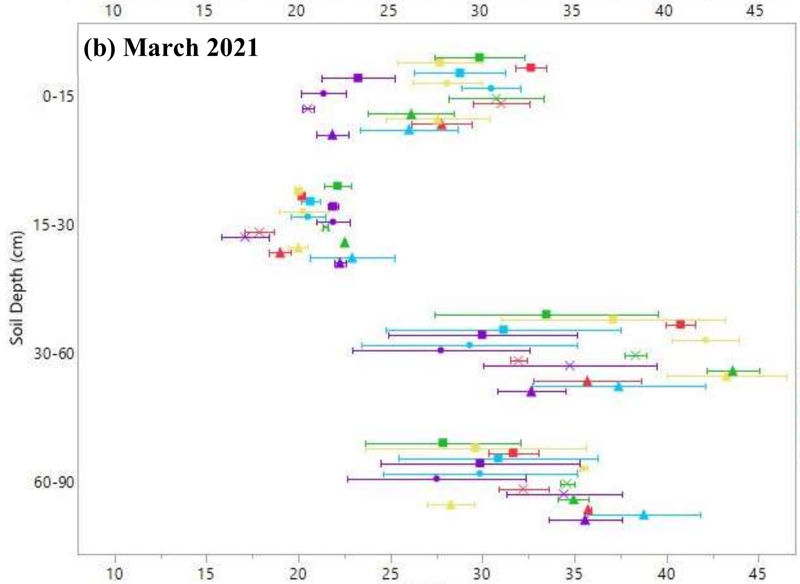
*** Significant at the 0.001 probability level.

2.3.3. Soil C content and $\delta^{13}C$ value at various soil depths

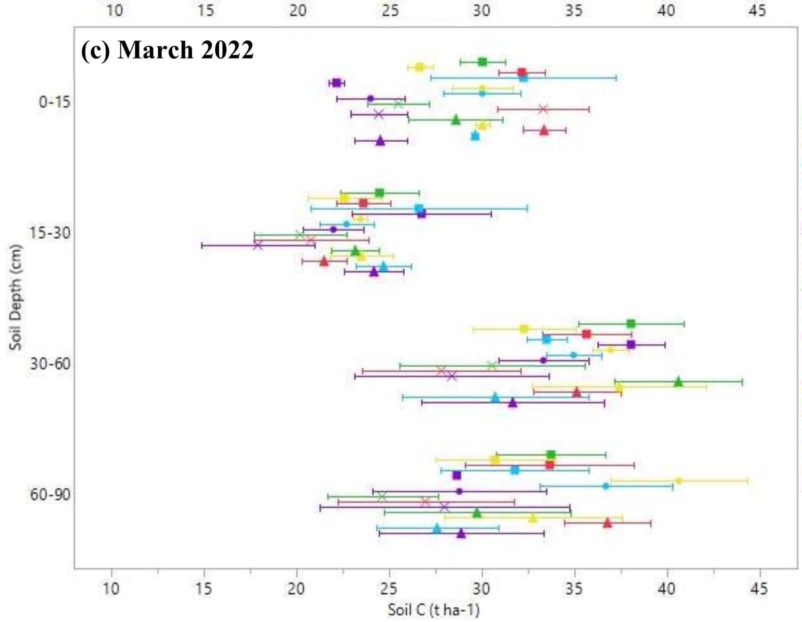
The influence of compost type and application rate on soil C content across different depths (0-15 cm, 15-30 cm, 30-60 cm, and 60-90 cm) varied considerably (Fig. 2.8). Data reflects soil samples collected post-second-year tomato harvest (August 2020), pre-third-year tomato transplant (March 2021), and at the end of the experiment (March 2022). In August 2020, significant effects of compost on soil C were observed in the 0-15 cm soil layers at 0% and 70% N levels, and in the 30-60 cm soil layer at the 100% N level, with the lowest soil C in no-compost controls (Fig. 2.8a). By March 2021, no-compost controls showed significantly lower soil C than compost treatments in the 70%, 85%, and 100% N levels of the 0-15 cm soil layer and in the 70% N level of the 15-30 cm soil layer, but compost had no significant effect on soil C in the 30-60 or 60-90 cm soil layers (Fig. 2.8b). In March 2022, compost significantly increased soil C across all fertilizer N levels in the 0-15cm soil layer and at the 85% N level in the 60-90 cm soil layer (Fig. 2.8c). Our ANOVA results show that compost and soil depth had a significant effect on soil C content, but fertilizer N level did not (Table 2.6).



Treatment	Soil Depth (cm) 0-15	15-30	30-60	60-90
0% N no C	a	a	ab	b
0% N FW-9	ab	a	ab	b
0% N FW-18	a	a	a	b
0% N GW-9	ab	a	b	a
0% N GW-18	b	a	ab	b
70% no C	a	a	a	a
70% FW-18	b	a	a	a
70% GW-18	ab	a	a	a
85% N no C	a	a	a	a
85% N FW-9	a	a	a	a
85% N GW-9	a	a	a	a
100% N no C	a	ab	a	a
100% N FW-9	a	ab	ab	a
100% N FW-18	a	b	ab	a
100% N GW-9	a	a	ab	a
100% N GW-18	a	b	b	a



Treatment	Soil Depth (cm) 0-15	15-30	30-60	60-90
0% N no C	a	ab	a	a
0% N FW-9	a	b	a	a
0% N FW-18	a	a	a	a
0% N GW-9	a	ab	a	a
0% N GW-18	a	b	a	a
70% no C	a	a	a	a
70% FW-18	b	a	a	a
70% GW-18	b	b	a	a
85% N no C	a	a	a	a
85% N FW-9	b	a	a	a
85% N GW-9	ab	a	a	a
100% N no C	a	a	a	a
100% N FW-9	ab	a	a	a
100% N FW-18	b	a	a	a
100% N GW-9	ab	a	a	a
100% N GW-18	ab	a	a	a



Treatment	Soil Depth (cm) 0-15	15-30	30-60	60-90
0% N no C	a	a	a	a
0% N FW-9	ab	a	a	a
0% N FW-18	b	a	a	a
0% N GW-9	ab	a	a	a
0% N GW-18	ab	a	a	a
70% no C	a	a	a	a
70% FW-18	b	a	a	a
70% GW-18	ab	a	a	a
85% N no C	a	a	a	a
85% N FW-9	a	a	a	ab
85% N GW-9	a	a	a	b
100% N no C	a	a	a	a
100% N FW-9	b	a	a	a
100% N FW-18	b	a	a	a
100% N GW-9	ab	a	a	a
100% N GW-18	b	a	a	a

Figure 2.8: Soil C content of the four soil layers 0-15 cm, 15-30 cm, 30-60 cm, and 60-90 cm for the two compost types (FW and GW) at three application rates (no C = no compost, 9 = 9 t ha^{-1}, and 18 = 18 t ha^{-1}), and four N levels (0%, 70%, 85%, and 100%) of the recommended amount, following crop harvest in (a) August 2020, before tomato transplant in (b) March 2021, and at the end of the experiment in (c) March 2022. The line bars represent standard error (n = 3). Lowercase letters next to the legend indicate significant differences among compost treatments within a fertilizer level and soil layer at a statistical significance level of alpha = 0.05.

Table 2.6. Effects of compost, fertilizer N level, and soil depth on soil C stock (t ha^{-1}). Pr (>F) values from ANOVA.

Source	Pr (>F)
Compost	0.000116***
N level	0.0829
Depth	<0.0001***
Compost × N level	0.842
Compost × Depth	0.00517**
N level × Depth	0.624
Compost × N level × Depth	0.999

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

After the final harvest in September 2021, we analyzed the effect of compost application on soil $\delta^{13}C$ at four soil depths (0-15 cm, 15-30 cm, 30-60 cm, and 60-90 cm) in the 0% N main plots (Table 2.7). The effects varied by depth. In the 0-15 cm soil layer, compost application resulted in more negative $\delta^{13}C$ values than the no-compost control, with higher compost application rates showing the most pronounced differences from controls. There were no significant differences in $\delta^{13}C$ between FW and GW compost treatments in the 0-15, 15-30, or 30-60 cm soil layers. A general trend was observed where $\delta^{13}C$ became more negative with increasing depth, except in the treatment of 18 t ha^{-1} GW compost. ANOVA results indicated that soil depth significantly influenced soil $\delta^{13}C$, but compost did not if all depths were considered (Table 2.8).

Table 2.7. $\delta^{13}C$ values (means ± 1 SE) (‰) across five compost treatments in the 0% N main plots for soil layers 0-15 cm, 15-30 cm, 30-60 cm, and 60-90 cm following the final harvest in September 2021. Lowercase letters indicate significant differences among treatments within each soil layer and uppercase letters indicate significant differences between soil depths within each treatment, both at a statistical significance level of alpha = 0.05.

	Depth (cm)			
Treatment	0-15	15-30	30-60	60-90
no C	-24.55 (0.31)b C	-24.99 (0.07)a BC	-25.39 (0.05)a AB	-25.77 (0.23)ab A
FW-9	-25.21 (0.28)ab B	-25.11 (0.39)a B	-25.50 (0.20)a B	-26.31 (0.38)a A
FW-18	-25.43 (0.34)a AB	-24.94 (0.23)a B	-25.28 (0.14)a AB	-25.53 (0.20)ab A
GW-9	-25.08 (0.06)ab BC	-24.73 (0.25)a C	-25.38 (0.10)a AB	-25.74 (0.15)ab A
GW-18	-25.42 (0.11)a A	-24.84 (0.23)a B	-25.59 (0.13)a A	-25.33 (0.07)b AB

Table 2.8. Effects of compost and soil depth on soil $\delta^{13}C$ (‰). Pr (>F) values from ANOVA.

Source	Pr (>F)
Compost	0.610
Depth	<0.0001***
Compost × Depth	0.000411***

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

2.3.4. Nematode population and diversity

Nematode population and diversity, important for understanding C mineralization and sequestration, were analyzed to assess treatment effects in this study. Nematode individual genera were evaluated in soils from the same seven treatments as the gas sampling: 0% N × no compost, 85% N × no compost, 85% N × FW 9 t ha^{-1}, 85% N × GW 9 t ha^{-1}, 100% N × no compost, 100% N × FW 9 t ha^{-1}, and 100% N × GW 9 t ha^{-1} (Fig. 2.9). Differences were found between treatments, though not significant. In the 85% N fertilizer level group, the application of GW compost led to higher populations of bacterial-feeding Panagrolaimus compared to no-compost treatments (Fig. 2.9a). Additionally, the populations of fungal-feeding Aphelenchoides were increased by both compost types in the 85% N fertilizer level (Fig. 2.9c). At the 100% N fertilizer level, fewer Acrobeloides were found in compost treatments compared to no-compost treatments (Fig. 2.9b). ANOVA analysis of each nematode genus across the seven treatments showed no significant treatment effects.

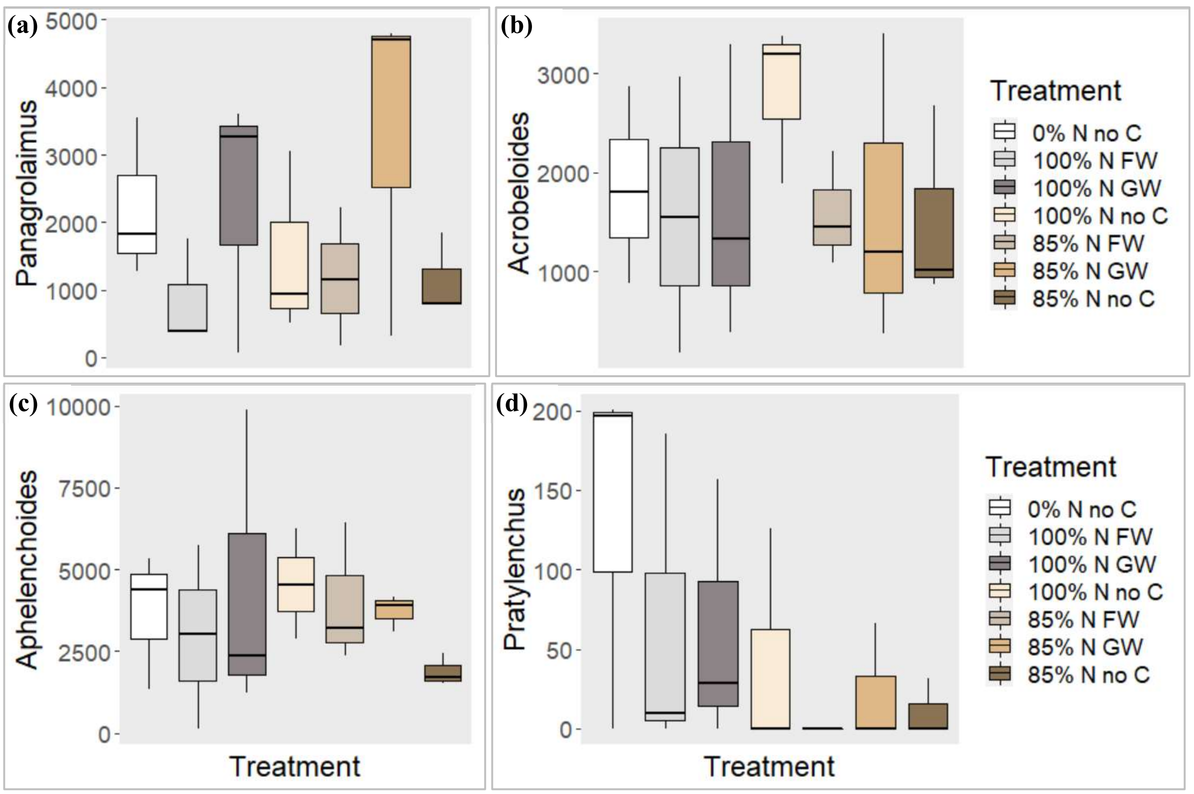


Figure 2.9. Individual genera of nematodes present in soil at the end of the experiment in January 2022, including (a) Panagrolaimus, (b) Acrobeloides, (c) Aphelenchoides, and (d) Pratylenchus, for the two compost types (FW and GW) at two application rates (no C = no compost or 9 t ha^{-1}), and three N levels (0%, 85%, and 100%) of the recommended amount. The line bars represent standard error (n = 3). Data and statistical analyses were provided by the University of California-Davis Soil Ecology and Pest Management Lab, and no significant treatment effects were observed.

2.4. Discussion

2.4.1. Compost and N fertilizer effects on gas emissions

Our results demonstrated that fertilizer N level had a significant effect on N_2O and CH_4 fluxes, but the application of compost did not affect cumulative N_2O or CH_4 emissions, although there was a compost effect on CO_2 fluxes. For each treatment used in the gas component of this study, the N_2O EFs were less than 1% of the applied N and the compost treatments at both fertilizer N levels had lower EFs than no-compost controls. These treatment EFs are lower than the IPCC documented EF, which states that 1% of N applied to soil is typically emitted as N_2O (IPCC 2019). Overall, the observed N_2O emissions in this study were relatively low, likely due to the use of subsurface drip irrigation, an irrigation method known to reduce soil N_2O emissions compared to other conventional practices such as furrow irrigation (Kallenbach et al. 2010). The effects of compost type and rate on N_2O emissions in this study might have been obscured by complete denitrification, a process that is favored when sufficient C substrate, as provided by compost application, is present in the environment. This process leads to the reduction of N_2O to N_2, a gas that was not measured in our study. The main factors contributing to an increased ratio of N_2O/N_2 include a rise in oxidant (NO_3^- or NO_2^-), an increase in O_2 levels, and a decrease in C availability (Firestone and Davidson 1989). When the availability of oxidant greatly exceeds the availability of reductant (OC), the oxidant may be incompletely utilized and N_2O is produced (Firestone and Davidson 1989). In the presence of O_2, nitrification is a process in which N_2O is mainly produced by nitrifying bacteria, but higher N_2O emissions generally occur under anoxic conditions through denitrification (Khalil et al. 2004). Reducing N_2O emissions primarily focuses on lowering the availability of mineral N. Using organic amendments with high C:N ratios have been shown to lower mineral N through immobilization, potentially reducing N_2O emissions by up to 45% (Rothardt et al. 2021). A laboratory incubation study on different agro-industrial wastes reported that composts with high initial soluble N contents and low C:N ratios led to increased N_2O emissions (Santos et al. 2021). In our study, both FW and GW composts had C:N ratios consistently >17 each year (Table 2.2). In two out of three years, the C:N ratios of FW compost were slightly higher than those of GW compost, which contained more inorganic N than FW compost. This might explain the results that we obtained from a crop N use efficiency study using the same composts where FW compost applied with 100% N fertilizer immobilized fertilizer N (Haas et al. 2023, unpublished), possibly leading to lower N_2O

fluxes in FW compost treatments compared to the no-compost control during the 2020 growing season from May to August in our current study (Fig. 2.2d).

The fluxes of CH_4 remained low over the two-year measurement, with approximately half of the measurements being negative, indicating alternate periods of minor positive emissions and a sink effect. The annual cumulative CH_4 emissions ranged from -1 to 1 kg CH_4-C ha^{-1}, marked by relatively high standard errors, leading to no statistically significant differences among treatments in either year. A recent study comparing clay soil amended with eight different composted agro-industrial wastes, those more representative of FW than GW, found that six out of eight compost treatments exhibited an overall sink effect for CH_4 (Santos et al. 2021). However, in our study, due to the high variability in CH_4 flux measurements, compost application did not show a statistically significant effect on CH_4 fluxes or cumulative emissions (Table 2.4).

Annual cumulative CO_2 emissions did not vary significantly between treatments during the investigation years. In 2019-2020, FW compost increased CO_2 emissions when fertilizer N was applied at the rate of 85% N, whereas it decreased CO_2 emissions at the 100% N level. Conversely, CO_2 emissions from the GW compost treatments were similar to the no-compost controls at both 85% and 100% fertilizer N levels. However, in 2020-2021, no statistically significant treatment effects were observed. Previous studies have reported increases in CO_2 emissions following organic amendment incorporation in both laboratory and field conditions (Ray et al. 2020; Santos et al. 2021). An incubation study conducted by Ma et al. (2021) also found higher CO_2 released in FW and GW compost treatments than controls, with CO_2 emissions differing between these two composts. They attributed this difference to the C substrate type, such as soluble C and lignin content (Ma et al. 2021; Santos et al. 2021). Typically, FW compost is expected to have lower lignin, but higher soluble C content compared to GW compost. Consequently, significantly higher CO_2 emissions were observed from FW compost treatments at the 85% N level, whereas GW compost had similar emissions as no-compost controls at both fertilizer N levels.

The fluxes of N_2O and CO_2 increased following fertigation and precipitation events (Fig. 2.2 and 2.6), as expected. Fertigation contributes to these increases by raising inorganic N availability and creating anoxic conditions in the soil, thereby stimulating denitrification and N_2O production. Similarly, precipitation increases water-filled pore space and reduces soil O_2 levels, further driving the anaerobic process of denitrification. Additionally, wetting of dry soil due to precipitation boosts microbial activity and respiration, including that of denitrifying bacteria, leading to increased production of CO_2 and N_2O (Smith and Parsons 1985; Almagro et al. 2009; Ray et al. 2020). In contrast, CH_4 flux did not follow the same increasing trend following fertigation or precipitation events (Fig. 2.4), likely because the conditions that these events created did not sufficiently lower redox potential to the point that methanogenesis could be enhanced to produce CH_4.

2.4.2. Correlation matrices between gas fluxes and soil variables

Correlations between gas fluxes and soil variables varied in direction and strength of significance (Table 2.5). These analyses were conducted to determine the soil factors that impacted gas fluxes in this study. Surprisingly, N_2O fluxes did not show significant correlations with soil GWC, NH_4^+, NO_3^-, DOC, or pH. Typically, GWC is positively correlated with N_2O emissions (Smith and Parsons 1985; Mosier et al. 1998), and NH_4^+ and NO_3^-, as substrates for N_2O production, are expected to be positively correlated with N_2O emissions (Firestone and Davidson 1989; De Rosa et al. 2016). The lack of these correlations might be due to the placement of subsurface drip lines at a depth of 30 cm, causing water saturation primarily around the drip zone, while soil samples were taken from the top 15 cm. Additionally, the high variability in measuring low N_2O concentrations under subsurface drip irrigation (Kallenbach et al. 2010) might have further obscured any potential correlations between these fluxes and soil variables. CH_4 fluxes did not show significant correlation with NH_4^+, NO_3^-, DOC, or pH, yet it was unexpectedly negatively correlated with GWC. Typically, soil CH_4 emissions are expected to increase with higher soil saturation and DOC content (Le Mer and Roger 2001; Nazaries et al. 2013). CO_2 flux, positively correlated with DOC, was unexpectedly negatively correlated with GWC, despite the usual increase in microbial respiration of CO_2 with greater soil moisture (Almagro et al. 2009). A significant positive correlation was observed between N_2O and CO_2 fluxes, consistent with both increasing after fertigation and precipitation events, due to enhanced denitrification and microbial respiration under increased soil moisture (Smith and Parsons 1985; Almagro et al. 2009; Ray et al. 2020). The negative correlation between CO_2 and CH_4 fluxes algins with previous findings (Ishizuka et al. 2002), although positive correlations have been observed in other contexts like compost windrows (Williams et al. 2019).

2.4.3. Compost effects on soil C content, $\delta^{13}C$, and nematode population and diversity

Our results demonstrated that FW and GW composts tended to increase soil C content compared to no-compost controls, but the level of significance varied by fertilizer N levels and soil depth (Fig. 2.8). Generally, compost had the greatest impact on soil C content in the 0-15 cm soil layer, and occasionally a significant compost effect was observed in deeper soils. Our ANOVA results suggested that compost and soil depth had a significant effect on soil C content, but fertilizer N level did not (Table 2.6). Decreasing soil C content with depth is the normal C distribution found in soil horizons, and microbial biomass and activity generally decline with soil depth as well. The main C sources in subsoils are DOC, root biomass, and physically or biologically transported particulate organic matter (Jobbágy and Jackson 2000; Rumpel et al. 2012); however, roots in the tomato cropping system used in this study were concentrated within the top 40 cm of soil and tillage only occurred in the topsoil, limiting these mechanisms of downward C movement.

The $\delta^{13}C$ results presented in this study also support that FW and GW composts increased topsoil C content and that the composts were the main source of newly added C, as $\delta^{13}C$ can be used as a tracer technique to characterize the dynamics of 'native' and 'new' SOC (Gerzabek et al. 2001; Lynch et al. 2006; Nguyen-Sy et al. 2020; You et al. 2021). The FW and GW composts used in this study had $\delta^{13}C$ values ranging from -27.81 to -27.46‰ (Table 2.2). Our $\delta^{13}C$ results demonstrated that compost treatments had lower $\delta^{13}C$ values than the no-compost controls and the $\delta^{13}C$ values decreased with increasing compost application rates in the 0-15 cm soil layer (Table 2.7). There were no significant differences between FW and GW compost in the 0-15, 15-30, or 30-60 cm soil layers. These results are in line with our finding that compost had the greatest impact on soil C stocks in the top 15 cm of soil and indicates that compost-derived C was present in the topsoil because the depleted $\delta^{13}C$ organic matter from compost was incorporated into the soil and thus lowered the soil $\delta^{13}C$ signature. Similar results were reported in a previous study where, for various soil types, SOC was reported to be 3 times greater in compost-amended soils than in controls in the 0-15 cm layer soil, while no significant difference in SOC was found in the 15-30 cm layer (Brown and Cotton 2011).

These findings suggest that compost-C was sequestered into the soil, and therefore annual compost application may contribute to long-term soil C storage. It should be noted that this study spanned a period of only three years, and significant compost effects may take longer to develop, which may partially explain why compost only increased soil C in the topsoil. Given a longer experimental period, compost can eventually increase SOC at deeper soil depth. For example, a long-term field experiment lasting 18 years found that compost application increased SOC content (Ding et al. 2013). Another study reported that a single, high-rate application (300 t ha^{-1}) of FW compost increased

SOC and maintained a significantly increased level for the duration of the 11-year study in a corn cropping system (Reynolds et al. 2015).

Nematode populations and diversity play an important role in soil C sequestration (Shi et al. 2023), and our investigation into compost effects on nematodes provides further support to compost's role in improving C sequestration. Despite a lack of significant treatment effects on individual genera of nematodes, notable differences were found in specific genera (Fig. 2.9). At the 85% N fertilizer level, greater numbers of bacterial feeding Panagrolaimus were present with GW compost application, and both FW and GW composts increased fungal feeding Aphelenchoides compared to no-compost controls. These observations provide insight into compost's impacts on the soil food web, which is crucial given the role of nematode predation on bacterial communities in C decomposition and sequestration (Neher 2001; Grandy et al. 2016; Shi et al. 2023). The increase in nematode diversity and biomass following compost application may accelerate organic matter decomposition, leading to effective nutrient transformations and enhanced C and N content in soils.

2.5. Conclusion

The application of food waste and green waste composts in combination with synthetic fertilizer N does not change GHG emissions compared to no-compost controls but has potential to sequester C into agricultural soils. Both compost types had no significant effect on N_2O or CH_4 fluxes or annual cumulative emissions compared to no-compost controls, because N_2O mitigation effects of subsurface drip irrigation and high flux variations made treatment differences insignificant. Fertilizer N level played a more significant role in controlling GHG emissions, with lower fertilizer N rates reducing GHG fluxes. Compost increased C content in the top 15 cm of soil, and the new C was derived from compost. Compost also increased some individual genera of bacterial and fungal feeding nematodes, shedding insight into how compost influences the soil food web and supporting nematode predation on bacterial communities as a possible biological mechanism of C decomposition and sequestration. It should be noted that potential C offsets from increases in SOC are temporary and reversible, while those generated by decreases in N_2O emissions are permanent. The information presented in this work may be used to refine soil C and GHG emissions models, understanding of consequences of compost application and reduced fertilizer N inputs, and strategies for crop management to mitigate rising atmospheric GHG levels and climate change. Furthermore, this study emphasizes the need for GHG flux measurement methodology with improved precision to reduce variability and for future research involving longer field trials to improve our understanding of long-term compost effects on GHG emissions and climate change mitigation in agricultural soils.

2.6. References

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Chapter 3: Compost and nitrogen management effects on intrinsic water use efficiency in intensive California olive cultivation

3.1. Introduction

Agriculture accounts for 70% of global freshwater use (Calzadilla et al. 2010), which is of growing concern as global freshwater reserves are dwindling (Sophocleous 2004). Therefore, many water-security solutions are being proposed that incorporate increases in water use efficiency (WUE; biomass production per water consumed) in agricultural systems. California (CA) features a Mediterranean climate, with its arid summers increasingly challenging the state's irrigation-dependent agricultural sector due to growing water insecurity and more frequent droughts exacerbated by climate change. Despite these challenges, CA's Mediterranean climate is ideal for water-efficient olive production, leading to rapid growth in the olive industry, even as it faces unpredictability in other Mediterranean regions (Vossen 2007; IOC 2019; Pehlivanoglu et al. 2021). Olive trees are known for their drought tolerance and high water-use efficiency, requiring only 312 g of water to produce 1 g of dry matter (Gertsis et al. 2017), making them less water demanding than other tree crops. Research indicates that olive trees under water stress, with evapotranspiration (ETc) levels maintained between 33 and 40%, yield higher quality olive oils than those from trees without water stress (Berenguer et al. 2006). These findings make olive crops ideal for deficit irrigation practices, offering a resilient solution to future concerns over water insecurity.

Irrigation guidelines in CA need to be reassessed to ensure crop security as climate change impacts the state's diverse microclimatic regions. Additionally, high and super-high density (SHD) orchards are increasingly being adopted globally, specifically in CA, due to their suitability for mechanical harvesting, an approach that offers cost savings from reduced manual labor for olive harvest and increased yield per land area (Grattan et al. 2006). Olive trees were traditionally cultivated on non-irrigated drylands and spaced widely apart in orchards, and higher density orchards are expected to increase the crop's demand for nutrients and water. The existing irrigation guidelines for olives in CA's Sacramento Valley and San Joaquin Valley were established by Beede and Goldhamer (2005); however, developments in olive varieties and the impacts of climate change over the past decade have necessitated an update to these guidelines. Similarly, nutrient guidelines for CA's olive production also require re-evaluation, as the predominant nutrient management data used for these guidelines in CA are based on studies from other global regions such as Israel (Haberman et al. 2019), Italy (Regni and Proietti 2019), Spain (Morales-Sillero et al. 2009; Centeno et al. 2017), Portugal (Ferreira et al. 2020), and Greece (Chatzistathis et al. 2016). Updates to these guidelines should consider impacts on WUE since increasing the WUE of olive trees is beneficial in terms of water conservation, but it needs to be balanced with agronomic factors, such as yield (Gertsis et al. 2017). However, there is a lack of understanding regarding how nutrient management, including compost application, affects olive WUE.

Nitrogen (N) is an essential nutrient for olives, as it is for other plants. However, there is debate over whether annual applications of N are necessary to maintain productivity (Fernández-Escobar 2011; Haberman et al. 2019). Not only do most N management guidelines originate from other countries, but they are also often based on studies conducted in olive systems with lower tree densities, rain-fed orchards, or in container-grown trees (Ferreira et al. 1986; Erel et al. 2013; Leskovar and Othman 2019; Silva et al. 2023). The effects of N fertilization on N use efficiency have been investigated in olives (Fernández-Escobar et al. 2014; Ferreira et al. 2020), as have the effects of irrigation on WUE (Bacelar et al. 2007; Fernandes-Silva et al. 2010). However, there is a dearth of information on the effect of N fertilizer management on WUE in olives. Studies in other Mediterranean cropping systems have reported positive correlations between N fertilization rates and WUE in barley and bread and durum wheat (Cossani et al. 2012) and in peach trees (Pascual et al. 2016). Compost use has also been considered as a source of N for olives (De Sosa et al. 2022), and it is known to influence WUE in other cropping systems (Adamtey et al. 2010; Abd El-Mageed et al. 2018; Demir and Gülser 2021). The effects of compost on WUE may be attributed to improved soil structure, water holding capacity, and infiltration rates brought about by compost (Diacono and Montemurro 2010; Adugna 2016). However, to what extent compost changes olive WUE has not yet been explored.

WUE is defined as the ratio of carbon (C) assimilated to the water transpired by plants (Gertsis et al. 2017). Intrinsic WUE (iWUE), on the other hand, is the ratio between net carbon dioxide (CO_2) fixation and stomatal conductance, representing water stress when plants have high iWUE. Stomatal conductance refers to the opening of leaf stomata for the exchanges of gases such as CO_2 and water vapor. It is an important indicator of plant water status because stomata close under water stress to conserve water. Isotopic discrimination ($\Delta^{13}C$) is a plant's ability to selectively assimilate $^{12}CO_2$ over $^{13}CO_2$ (Berenguer et al. 2004). Isotopic fractionation of C during the photosynthesis process occurs for two reasons: 1) $^{12}CO_2$ is lighter than $^{13}CO_2$ and thus diffuses more rapidly, and 2) ribulose-1,5-bisphosphate carboxylase (an enzyme in the photosynthesis process) fixes $^{12}CO_2$ faster than $^{13}CO_2$ (Boyer 1996).

The inward diffusion and utilization of $^{12}CO_2$ into the stomata correlates with photosynthesis and plant dry mass. Under limiting water, stomata begin to close and trap $^{13}CO_2$ in the mesophyll where some is converted to biomass production through photosynthesis. Consequently, the relative assimilation of ^{12}C and ^{13}C isotopes by plants correlates with their iWUE (Boyer 1996). Studies have indicated that variations in leaf $\delta^{13}C$ values can reflect the effects of water stress and N enhancement on biomass production (Syvertsen et al. 1997; Brueck 2008). $\delta^{13}C$ is referred to as integrated WUE, and the technique of measuring the natural abundance of this stable isotope is effective because it biochemically records any seasonal cumulative tree stress in the $^{12}C/^{13}C$ fractionation of new plant growth (Grattan et al. 2006). $\delta^{13}C$ enrichment represents higher iWUE and is sensitive enough to capture seasonal changes (Du et al. 2021). Another stable isotope, ^{18}O, can also be used to indicate WUE as it correlates with transpiration, which is increased by higher stomatal conductance. ^{18}O enrichment of plant water occurs during transpiration and through biochemical fractionation during biomass synthesis (Barbour 2007). Thus, under water stress, olive trees are expected to increase the duration of stomatal closure, thereby increasing $\delta^{13}C$ and iWUE while decreasing $\delta^{18}O$.

The goal of this study was to understand the effects of compost application and N fertilization on olive iWUE. To achieve this goal, we conducted field trials in two SHD, 'Arbequina' olive orchards in the Sacramento and San Joaquin Valleys of CA. Different rates of compost combined with various fertilizer N rates were applied at each field site. Monthly leaf sampling during the growing season and concurrent soil sampling were conducted for analyzing $\delta^{13}C$, iWUE, $\delta^{18}O$, and soil gravimetric water content (GWC), NH_4^+, and NO_3^-. Olive yields from each treatment were also recorded. This study provides insight to develop N management guidelines that consider iWUE and yield for CA olives.

3.2. Materials and methods

3.2.1. Site descriptions and agronomic management

Field trials were conducted at two different commercial olive grower's super-high density (SHD) orchards, one in Woodland, CA and one in Stockton, CA, hereafter named the MR and ST sites, respectively. At both sites, the olive variety investigated was Arbequina. This variety was interplanted in large blocks with other varieties for pollination purposes. The MR site was in the Sacramento Valley and the soil is classified as Sehorn cobbly clay, a fine, montmorillonitic, thermic Entic Chromoxererts according to the United States Department of Agriculture, National Cooperative Soil Survey. The MR site was previously a grape vineyard, and the olive trees were planted in 2020, with tree row spacing of 4.3 m and tree spacing of 1.8 m to have ~1282 tree ha^{-1}. The ST site was in the San Joaquin Valley and the soil is classified as Stockton clay, a fine, montmorillonitic, thermic Typic Pelloxererts. Olive trees were planted at the ST site in 2006, therefore the trees were older and more mature than the MR trees, with tree row spacing of 4 m and tree spacing of 1.5 m to have ~1655 tree ha^{-1}. The tree rows at ST were hedged according to industry standard practice for SHD orchards, while the MR trees were not yet hedged because they were still in the young stage. Both MR and ST have been managed under conventional practices, including surface drip irrigation with mineral fertilizer (fertigation), and weed management with herbicides and mowing. Irrigation during the project term at MR represented industry standard practice, while irrigation at ST was applied at a slight deficit compared to standard practice due to older tree age and it being an 'off' year in the alternate bearing pattern for yield. Climatic data, including monthly precipitation and air temperature, were obtained from the automated weather stations in Woodland, CA and Linden, CA for the MR and ST sites, respectively, and were acquired through the CA Irrigation Management Information System (CIMIS). Baseline soil characteristics for both sites are summarized in Table 3.1.

Table 3.1. Baseline soil characteristics. Drainage class retrieved from the United States Department of Agriculture, National Cooperative Soil Survey.

Site	Total N (g kg^{-1})	Total C (g kg^{-1})	pH (H_2O 1: 1)	Drainage
MR	0.74	4.30	7.0	Well drained
ST	0.88	8.14	6.8	Somewhat poorly drained

3.2.2. Compost/Fertility management and experimental design

At both field sites, six experimental treatments were implemented using a randomized complete block design with four replicates (blocks) across ten hectares. Each experimental plot spanned the entire length of the existing tree rows (~300 m) and encompassed three tree rows. The treatments included the application of 9 t ha^{-1} of compost (equivalent to 159 kg compost-N ha^{-1}, see Table 3.2 for compost characteristics) and a control treatment with no compost, both of which were combined with three different levels of fertilizer N. At the MR site, the fertilizer N levels were set at 75%, 100%, or 125% of the locally recommended N fertilization rates of 112 kg N ha^{-1}, resulting in applied fertilizer N rates of 84, 112, or 140 kg N ha^{-1}. Meanwhile, at the ST site, the grower's preferred maximum fertilization rate, based on the orchard's fertility condition, was 56 kg N ha^{-1}. This led to the fertilizer N levels being set at 50%, 75%, or 100% of the locally recommended N fertilization rates, resulting in applied fertilizer N rates of 28, 42, and 56 kg N ha^{-1} at the ST site. At the MR site, N fertilization was conducted through seven separate fertigation events every two to three weeks during the growing period (UAN 32 at a rate of 16 kg N ha^{-1} each event). At the ST site, N fertilization was carried out through three separate fertigation events approximately once a month during the growing period (UAN 32 at a rate of 19 kg N ha^{-1} each event). Commercially purchased compost was applied in a band by tractor along both sides of the tree rows in April 2022 at both the MR and ST sites.

Table 3.2. Compost characteristics.

Feedstock	$\delta^{13}C$ (‰)	Total C (g kg^{-1})	Total N (g kg^{-1})	C:N
Green waste	-27.37	243.8	17.68	14:1

3.2.3. Plant sampling and analysis

At both the MR and ST sites, approximately once a month during the growing season and once in the following spring, four trees were randomly selected from the middle tree row of each treatment plot. From each tree, a west-facing branch was cut, and 10-15 mature leaves were collected from each branch and composited into one sample per plot. The leaf sampling dates were 7/30/2022, 9/1/2022, 10/5/2022, 11/16/2022, and 4/27/2023 at the MR site and 8/1/2022, 8/31/2022, 10/7/2022, 11/15/2022, and 5/3/2023 at the ST site. The leaf samples were dried in a 60° oven and then ground into powder using a ball mill. The ground samples were weighed into tin capsules and sent to the University of California-Davis Stable Isotope Facility (UC Davis SIF) for ^{13}C and ^{18}O analysis. The analysis of ^{13}C was performed using an Elementar Vario EL Cube or Micro Cube elemental analyzer (Elementar Analysensysteme GmbH, Hanau, Germany) interfaced to an Isoprime VisION IRMS (Elementar UK Ltd, Cheadle, UK). The final $\delta^{13}C$ values were expressed relative to international standards Vienna Pee Dee Belemnite (VPDB) (Eq. 3.1). The analysis of ^{18}O was performed using an Elementar vario PYRO cube elemental analyzer interfaced to an Elementar VisION IRMS (Elementar Analysensysteme GmbH, Langenselbold, Germany). The final $\delta^{18}O$ values were expressed relative to international standards Vienna Standard Mean Ocean Water (VSMOW) (Eq. 3.1).

$$\delta^{13}C_p \ \ or \ \delta^{18}O_p = \left(\left(R_{sample}/R_{standard}\right) - 1\right) \times 1000 \qquad \text{(Eq. 3.1).}$$

The leaf $\delta^{13}C_p$ provided by the UC Davis SIF was used to calculate $\Delta^{13}C$ as follows:

$$\Delta^{13}C = \left(\delta^{13}C_a - \delta^{13}C_p\right)/\left(1 + \left(\delta^{13}C_p/1000\right)\right) \qquad \text{(Eq. 3.2),}$$

where $\Delta^{13}C$ is the plant isotopic discrimination relative to the source, the atmosphere, and is derived from plant $\delta^{13}C$ (C_p) and atmospheric $\delta^{13}C$ (C_a) ($\delta^{13}C_a$ = -8‰) (Farquhar et al. 1989). The discrimination $\Delta^{13}C$ was then used to directly calculate iWUE:

$$iWUE = \frac{A}{g_{H_2O}} = C_a(b - \Delta^{13}C)/1.6(b - a) \qquad \text{(Eq. 3.3),}$$

where *a* is the discrimination against $^{13}CO_2$ during diffusion through leaf stomata (a = 4.4%), *b* is the net discrimination due to carboxylation (b = 27‰), and C_a is the ambient CO_2 concentration (C_a = 400 ppm) (Farquhar et al. 1989; Du et al. 2021). Ambient gas samples were collected periodically over the experimental period into pre-evacuated exetainers with septum, and CO_2 concentrations were determined using gas chromatography (Shimadzu Model 2014). The direct iWUE calculation (Eq. 3.3) is derived from leaf net photosynthesis (*A*), measured as total CO_2 uptake, and stomatal conductance to water vapor (g_{H_2O}), which is related to stomatal conductance to CO_2 (g_{CO_2}) by a factor of 1.6, as 1.6 is the ratio of water vapor to CO_2 diffusivity (Silva et al. 2015; Du et al. 2021).

Olives were harvested at the ST site on 11/15/2022 and at the MR site on 11/16/2022, using grape harvesters that had been modified to harvest olives, a standard mechanical harvesting practice in CA. The olives were collected from the middle tree row of each treatment plot along the entire length of the tree row, and field weights were converted to yield measured as weight per hectare.

3.2.4. Soil sampling and analysis

Soil samples from the 0-15 cm soil layer of each plot were collected using a 1.83-cm diameter steel soil auger, concurrently with leaf sampling at the MR site on 7/30/2022, 10/5/2022, 11/16/2022, and 4/27/2023 and at the ST site only on 5/3/2023. Four soil borings were collected from the middle tree row of each plot and approximately 15 cm from the irrigation lines and composited into one soil sample per plot. Soil gravimetric water content (GWC) was determined by comparing the field-moist to the oven-dry (105ºC) mass of the soil samples. Nitrate (NO_3^-) and ammonium (NH_4^+) concentrations in all soil samples were measured by extracting well-mixed soil with 0.5 M K_2SO_4 (4:1 extractant volume to soil mass ratio). The soil extract was then analyzed colorimetrically for NH_4^+ and NO_3^- using a Shimadzu spectrophotometer (Model UV-Mini 1240) (Forster 1995; Doane and Horwáth 2003).

3.2.5. Statistical analyses

Statistical analyses were conducted using linear mixed models in R program. For the investigated metrics of olive yield, $\delta^{13}C$, iWUE, $\delta^{18}O$, soil GWC, soil NH_4^+, and soil NO_3^-, treatment factors such as compost and N fertilizer level, along with their interactions, were treated as fixed effects, while replicates (blocks) were treated as random effects. Likelihood ratio tests (LRT) were performed for lmer models using the "anova" function to ensure no additional random effect interactions were needed for the best fit and appropriate complexity of models. Prior to statistical analyses, the assumptions of normal distributions and homogenous variances were verified. Analysis of variance (ANOVA) were performed to compare treatment effects and to determine if these effects were significant ($p<0.05$). Correlation analysis was performed individually for each field site using the "cor.test" function and Pearson's correlation coefficient for plant variables (iWUE and $\delta^{18}O$) and soil variables (GWC, NH_4^+, and NO_3^-).

3.3. Results

3.3.1. Environmental conditions

Climatic data, including monthly precipitation and temperature over the experimental period, are shown in Fig. 3.1. The observed weather patterns align with a typical Mediterranean climate, characterized by a rainy and cool winter from November to March, and a dry, hot summer from April to October. The temperature and precipitation trends were similar at both sites, though total precipitation was higher by 154 mm at the ST site compared to the MR site, and annual and growing season average temperatures were generally slightly higher at MR compared to ST.

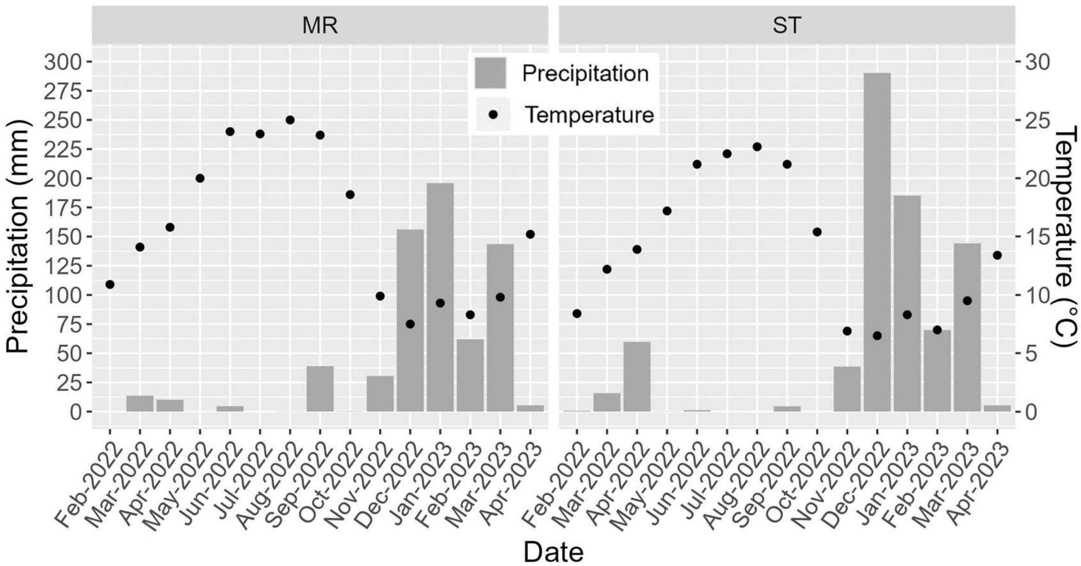

Figure 3.1. Monthly precipitation and temperature for the MR and ST field sites. Data is from the Woodland and Linden, California automated weather stations for the MR and ST sites, respectively, and was acquired through the California Irrigation Management Information System (CIMIS).

The amount of irrigation water applied at the MR and ST sites is shown in Figs. 3.2 and 3.3, respectively. At both sites, the amount of irrigation water applied peaked in July, during the middle of the growing season and the hot, dry summer. This tapered off with the onset of the rainy season, which began early in October 2022. The irrigation water applied at the MR site followed industry standard practice and was greater than the amounts applied at the ST site. At the ST site, irrigation was less crucial to crop productivity during the experimental period, due to the older tree age and the fact that it was on an ‘off’ year in the alternate bearing pattern for yield. Consequently, irrigation at this site was applied at a slightly reduced rate compared to standard practice.

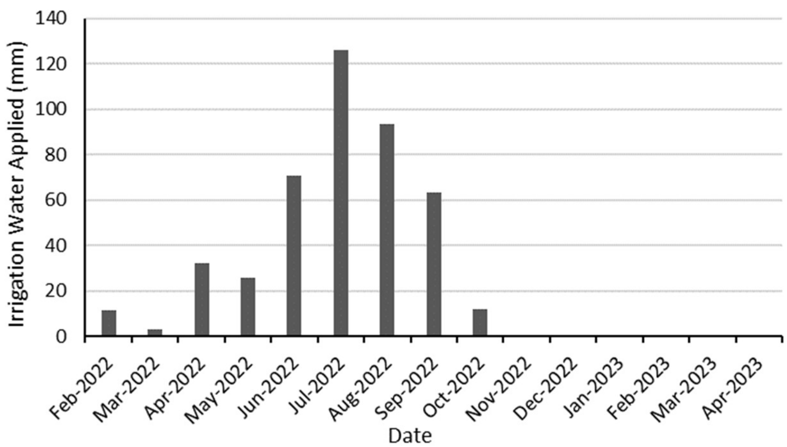

Figure 3.2. Irrigation water applied at the MR site each month starting in February 2022 before the growing season through April 2023.

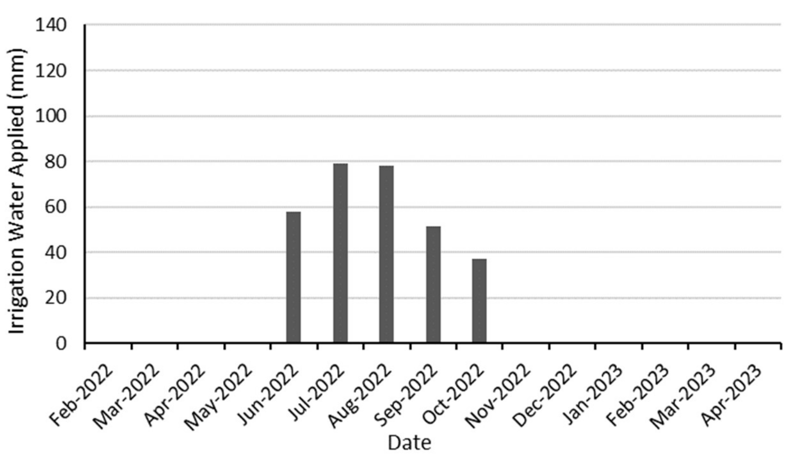

Figure 3.3. Irrigation water applied at the ST site each month starting in February 2022 before the growing season through April 2023.

3.3.2. Olive yield, $\delta^{13}C$, iWUE, and ^{18}O

The influence of compost and fertilizer N on olive yield varied between field sites (Fig. 3.4). At the MR site, compost and fertilizer N levels had no significant effect on yield. At the ST site, the yield of no-compost treatments increased slightly but not significantly with increasing fertilizer N levels, while compost treatments decreased yield with increasing fertilizer N level. Compost increased yield at the lowest fertilizer N level and decreased yield at the highest fertilizer N level compared to no-compost controls at the ST site. Overall, the yields at the ST site were lower than the MR site, due to the trees being in an off year for yield and this site was more impacted by frost during the prior winter season, which decreased bud production.

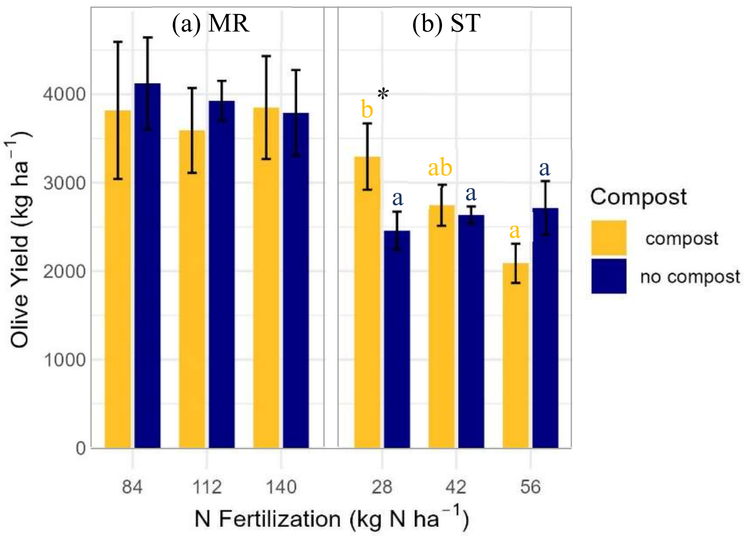

Figure 3.4. Olive yield for no compost controls and compost treatments at three fertilizer N rates at (a) MR (84, 112, and 140 kg N ha^{-1}) and (b) ST (28, 42, 56 kg N ha^{-1}). The line bars represent standard error (n = 4). Asterisk (*) indicates significant difference in yield between no compost and compost treatments at a given N fertilization level, and lowercase letters indicate significant differences in yield between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.

Leaf $\delta^{13}C$ values at the MR site for five sampling dates are shown in Fig. 3.5. In general, but not always significantly, compost application tended to increase $\delta^{13}C$ values at the two lower fertilizer N levels and decrease $\delta^{13}C$ at the highest fertilizer N level, compared to plots that received no compost. Certain statistically significant treatment effects were observed only on 10/5/2022, the end of the hot and dry summer season, and on 4/27/2023, the end of the rainy season and the beginning of warmer weather. On these dates, no significant differences in $\delta^{13}C$ were observed among fertilizer N levels when no compost was applied, while compost significantly decreased $\delta^{13}C$ with increasing fertilizer N levels. On 10/5/2022, compost significantly decreased $\delta^{13}C$ compared to no-compost controls only at the highest fertilzer N rate. On 4/27/2023, compost significantly increased $\delta^{13}C$ compared to no-compost controls only at the lowest fertilzer N rate.

Olive iWUE results had the same trends and significant treatment effects on the same sampling dates as $\delta^{13}C$, due to $\delta^{13}C$ having been used to directly calculate iWUE (Eq. 3.3) (Fig. 3.6). In summary, compost generally increased iWUE at the two lower fertilizer N levels and decreased iWUE at the highest fertilizer N level.

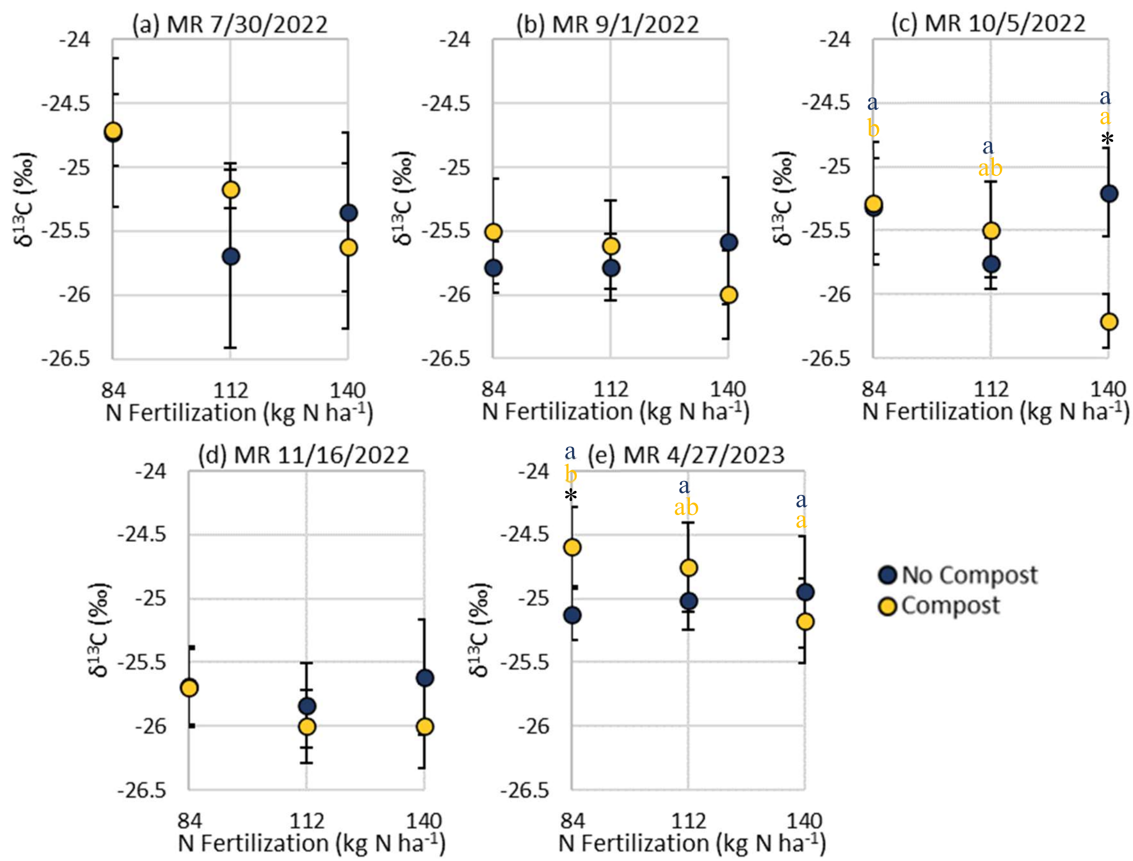

Figure 3.5. Leaf $\delta^{13}C$ on (a) 7/30/2022, (b) 9/1/2022, (c) 10/5/2022, (d) 11/16/2022, and (e) 4/27/2023 for the no compost controls and compost treatments at three fertilizer N rates (84, 112, and 140 kg N ha^{-1}) at MR. The line bars represent standard error (n = 4). Asterisks (*) indicate significant difference in $\delta^{13}C$ between no compost and compost treatments at a given N fertilization level, and lowercase letters indicate significant differences in $\delta^{13}C$ between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.

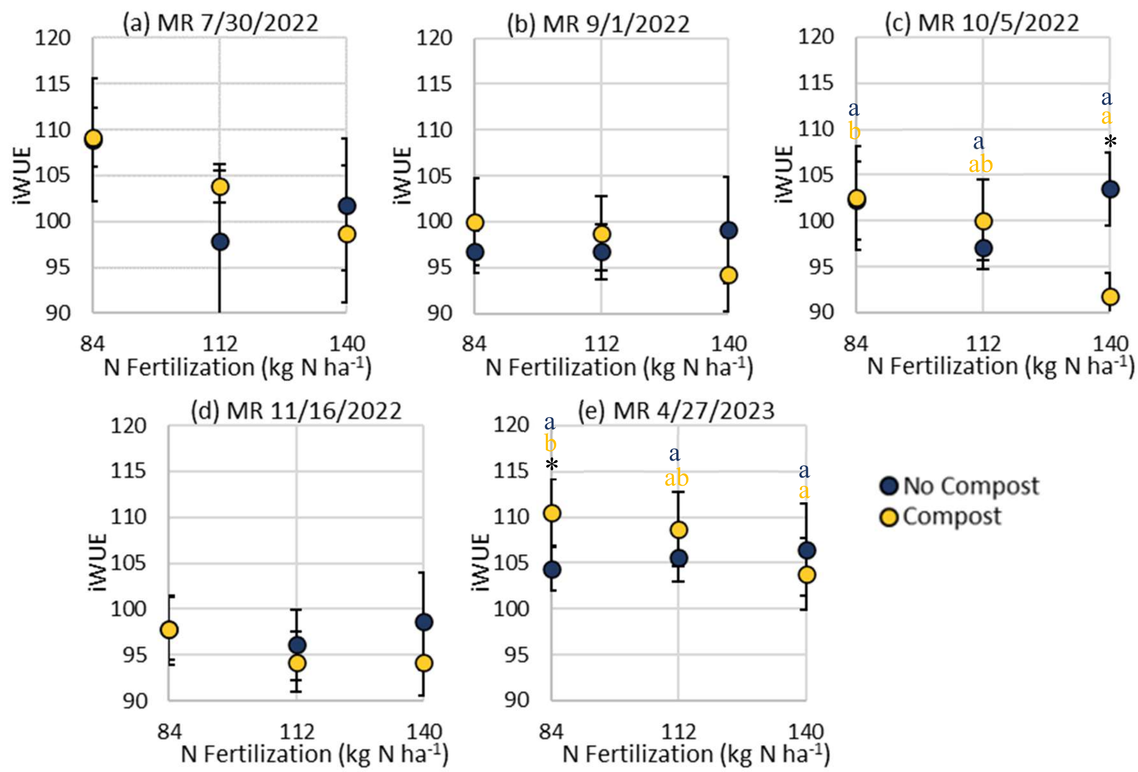

Figure 3.6. Olive iWUE on (a) 7/30/2022, (b) 9/1/2022, (c) 10/5/2022, (d) 11/16/2022, and (e) 4/27/2023 for the no compost controls and compost treatments at three fertilizer N rates (84, 112, and 140 kg N ha^{-1}) at MR. The line bars represent standard error (n = 4). Asterisks (*) indicate significant difference in iWUE between no compost and compost treatments at a given N fertilization level, and lowercase letters indicate significant differences in iWUE between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.

Values of leaf $\delta^{13}C$ and iWUE for the five sampling dates at the ST site are shown in Figs. 3.7 and 3.8, respectively. Olive iWUE was calcualted directly using $\delta^{13}C$, leading to a direct correlation between the two metrics. Generally, compost and fertilizer N levels had no significant effect on either $\delta^{13}C$ or iWUE, except on 8/3/2022, when compost significantly increased $\delta^{13}C$ and iWUE compared to the no-compost treatments in the plots that received the lowest fertilizer N rate.

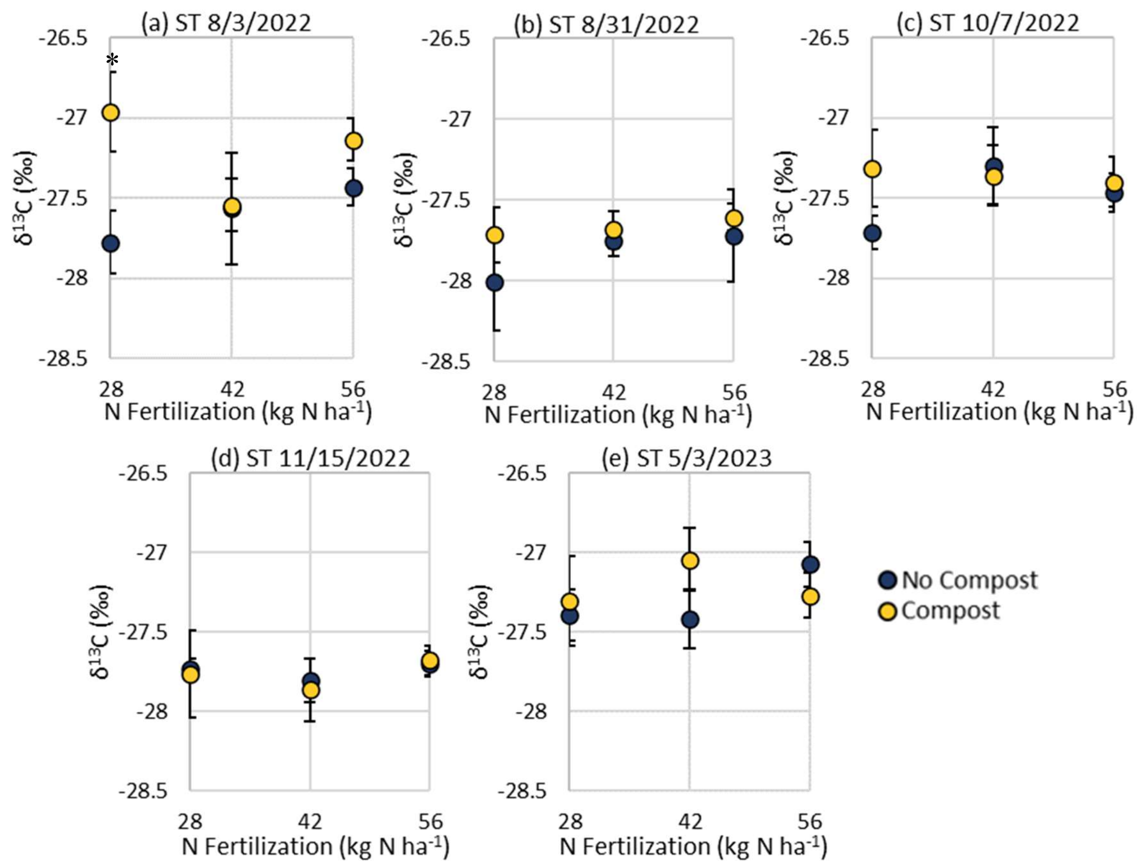

Figure 3.7. Leaf $\delta^{13}C$ on (a) 8/3/2022, (b) 8/31/2022, (c) 10/7/2022, (d) 11/15/2022, and (e) 5/3/2023 for the no compost controls and compost treatments at three fertilizer N rates (28, 42, and 56 kg N ha^{-1}) at ST. The line bars represent standard error (n = 4). Asterisk (*) (8/3/2022 only) indicates significant difference in $\delta^{13}C$ between no compost and compost treatments at a given N fertilization level at a statistical significance level of alpha = 0.05, and no significant differences in $\delta^{13}C$ between N fertilization levels across either compost treatment were observed.

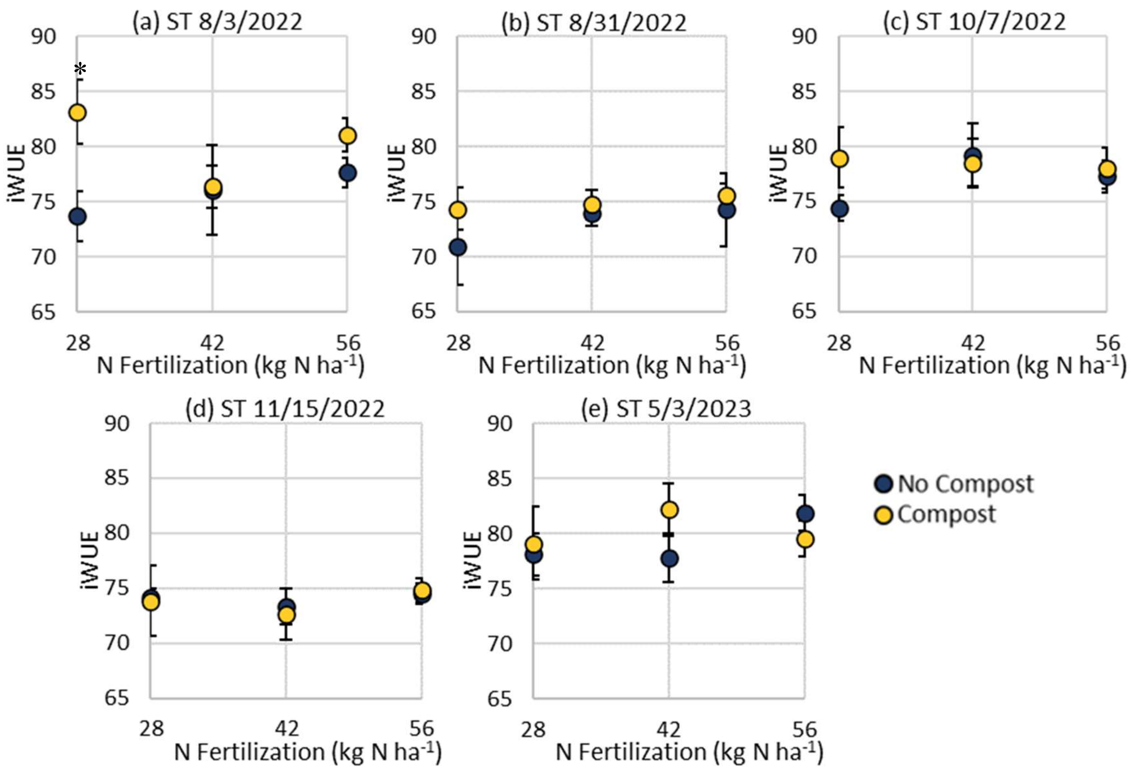

Figure 3.8. Olive iWUE on (a) 8/3/2022, (b) 8/31/2022, (c) 10/7/2022, (d) 11/15/2022, and (e) 5/3/2023 for the no compost controls and compost treatments at three fertilizer N rates (28, 42, and 56 kg N ha^{-1}) at ST. The line bars represent standard error (n = 4). Asterisk (*) (8/3/2022 only) indicates significant difference in iWUE between no compost and compost treatments at a given N fertilization level at a statistical significance level of alpha = 0.05, and no significant differences in iWUE between N fertilization levels across either compost treatment were observed for any sampling date.

Leaf $\delta^{18}O$ values at the MR site for the five sampling dates are shown in Fig. 3.9. No effects from compost or fertilizer N level treatment were observed on 7/30/2022, 11/16/2022, or 4/27/2023. However, on 9/1/2022 and 10/5/2022, significantly lower $\delta^{18}O$ was observed at the highest fetilizer N level compared to the two lower fertilizer N levels when compost was applied. On these same dates, significantly lower $\delta^{18}O$ was also observed in the compost treatments compared to the no-compost controls when the highest fertilizer N rate was applied. However, on 10/5/2022, compost increased $\delta^{18}O$ compared to the no-compost controls at the medium fertilizer level.

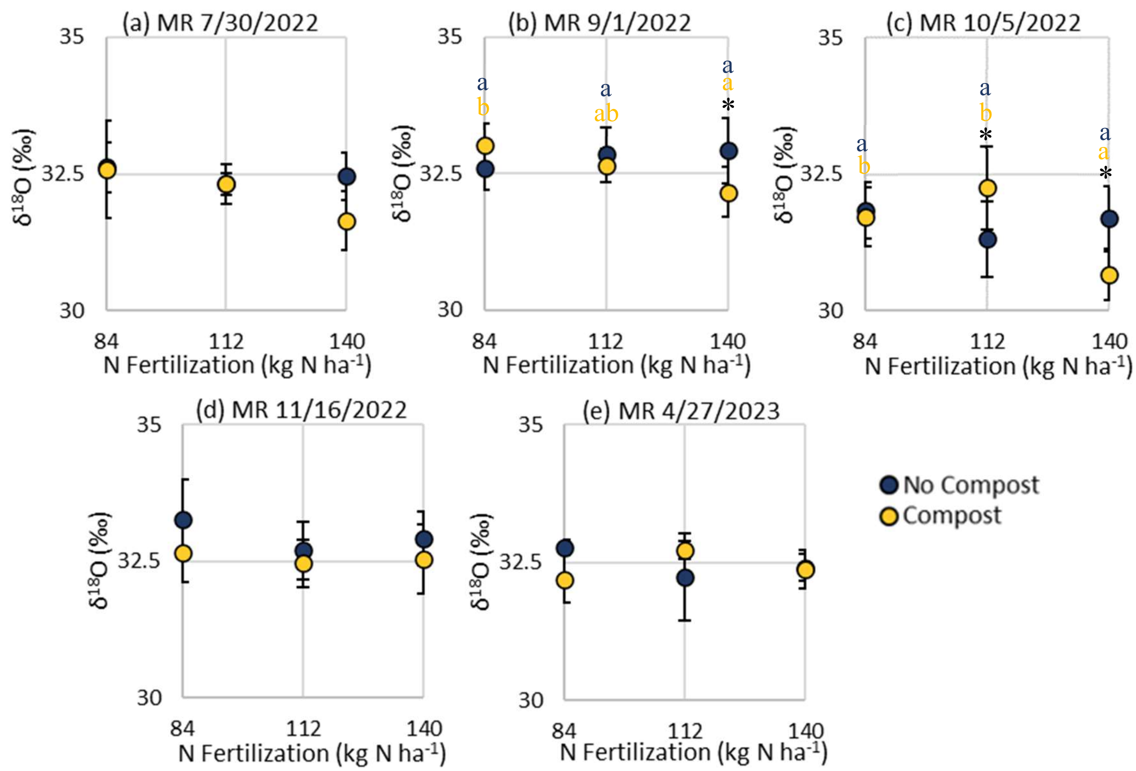

Figure 3.9. Leaf $\delta^{18}O$ on (a) 7/30/2022, (b) 9/1/2022, (c) 10/5/2022, (d) 11/16/2022, and (e) 4/27/2023 for the no compost controls and compost treatments at three fertilizer N rates (84, 112, and 140 kg N ha^{-1}) at MR. The line bars represent standard error (n = 4). Asterisks (*) indicate significant difference in $\delta^{18}O$ between no compost and compost treatments at a given N fertilization level, and lowercase letters indicate significant differences in $\delta^{18}O$ between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.

At the ST site, no consistent effects from compost or N fertilization on $\delta^{18}O$ were observed (Fig. 3.10). Of the five sampling dates, significant treatment effects were only observed on 8/3/2022. On this date, compost application resulted in reduced $\delta^{18}O$ compared to the no-compost controls at the medium fertilizer N level. Additionally, in the plots that received no compost, $\delta^{18}O$ was highest at the medium fertilizer N level and lowest at the high fertilizer N level.

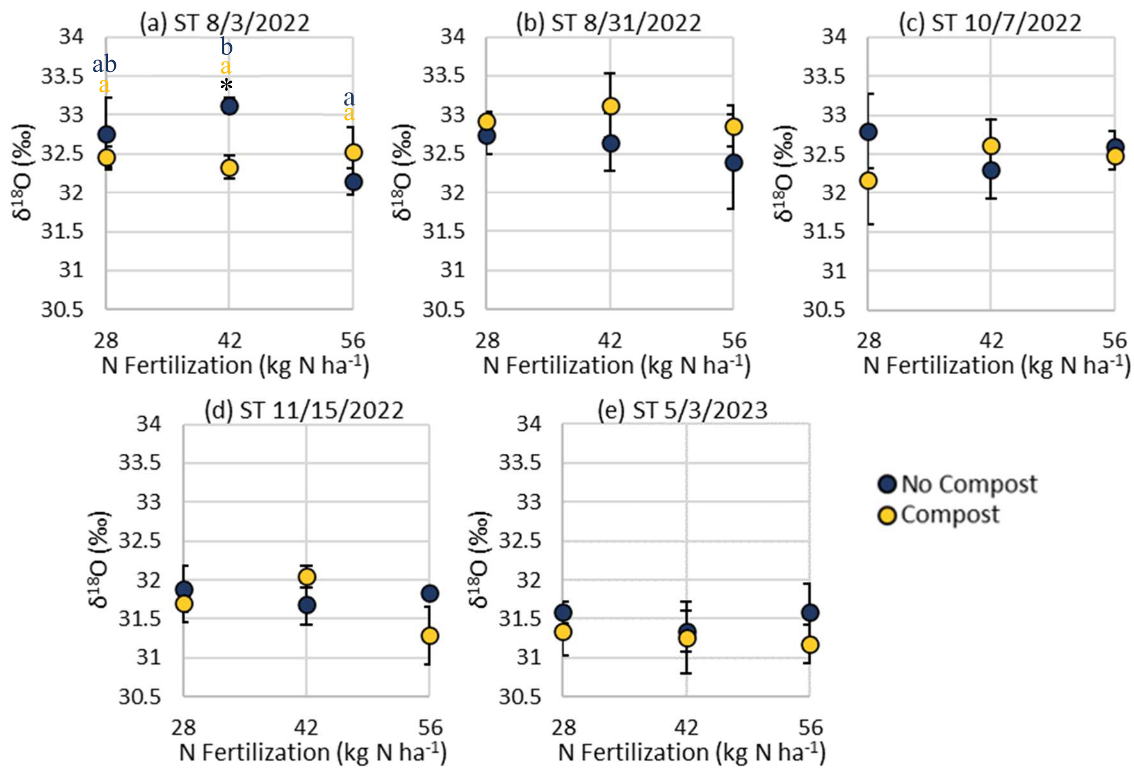

Figure 3.10. Leaf $\delta^{18}O$ on (a) 8/3/2022, (b) 8/31/2022, (c) 10/7/2022, (d) 11/15/2022, and (e) 5/3/2023 for the no compost controls and compost treatments at three fertilizer N rates (28, 42, and 56 kg N ha^{-1}) at ST. The line bars represent standard error (n = 4). Asterisk (*) (8/3/2022 only) indicates significant difference in $\delta^{18}O$ between no compost and compost treatments at a given N fertilization level, and lowercase letters (8/3/2022 only) indicate significant differences in $\delta^{18}O$ between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.

3.3.3. Soil gravimetric water and inorganic N content

The effects of compost and fertilizer N on soil GWC were not consistent at the MR site (Fig. 3.11). The only significant treatment effect was observed on 11/16/2022 when compost increased GWC compared to the no-compost controls at the highest fertilizer N level. Soil GWC was measured at the ST site only on 5/3/2023, and no statistically significant treatment effects were observed (Fig. 3.12).

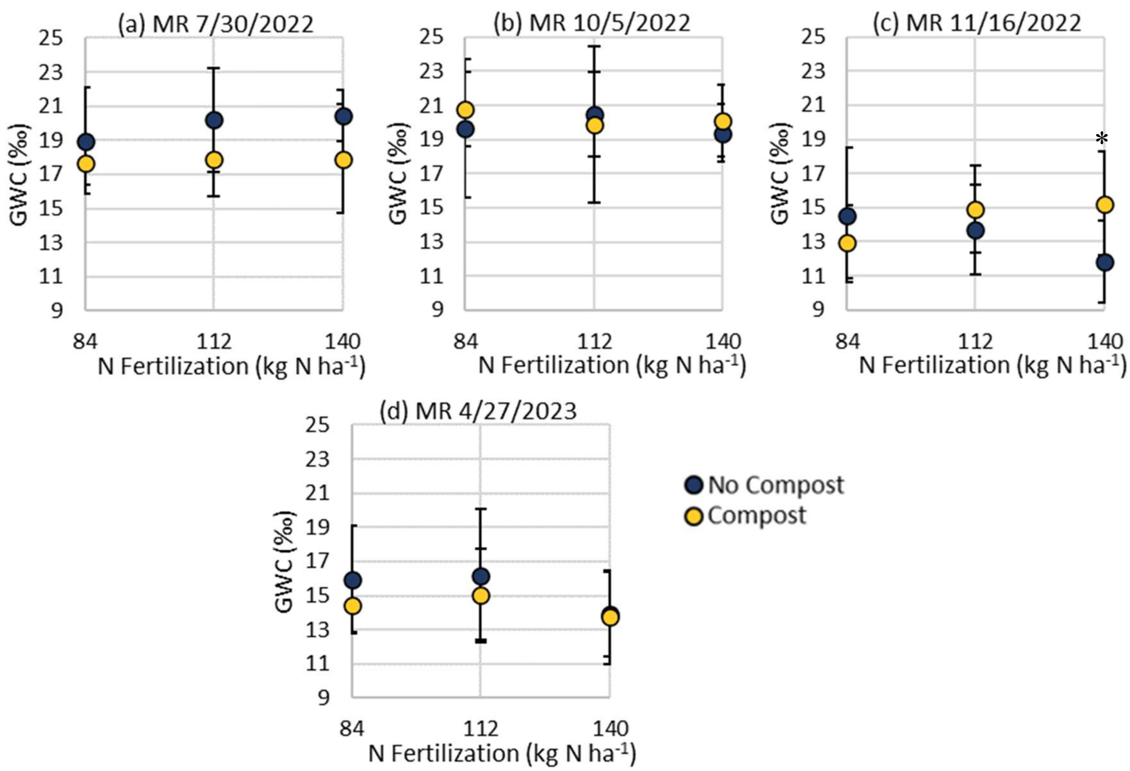

Figure 3.11. Soil GWC on (a) 7/30/2022, (b) 10/5/2022, (c) 11/16/2022, and (d) 4/27/2023 for the no compost controls and compost treatments at three fertilizer N rates (84, 112, and 140 kg N ha^{-1}) at MR. The line bars represent standard error (n = 4). Asterisk (*) (11/16/2022 only) indicates significant difference in GWC between no compost and compost treatments at a given N fertilization level, and no significant differences in GWC between N fertilization levels across a compost treatment were observed at a statistical significance level of alpha = 0.05.

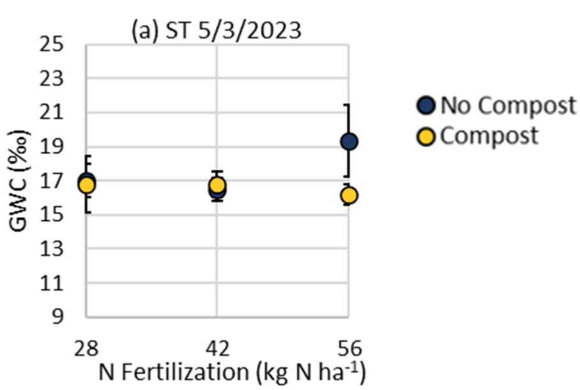

Figure 3.12. Soil GWC for the no compost controls and compost treatments at three fertilizer N rates (28, 42, and 56 kg N ha^{-1}) at ST on 5/3/2023. The line bars represent standard error (n = 4). No significant differences between treatments were observed at a statistical significance level of alpha = 0.05.

Soil NH_4^+ concentrations at the MR site remained low throughout the experimental period (Fig. 3.13). At this site, the only significant treatment effect on NH_4^+ concentration was observed on 7/30/2022. On this date, compost treatments exhibited lower NH_4^+ concentration compared to the no-compost controls when the highest fertilizer N level was applied. Conversely, at the ST site, compost treatments had significantly higher NH_4^+ concentration compared to the no-compost controls at the highest fertilizer N level (Fig. 3.14). Also at the ST site, NH_4^+ concentrations were similar across the three fertilizer N levels when compost was applied. Interestingly, in the no-compost controls, soil NH_4^+ concentrations decreased with increasing fertilizer N levels.

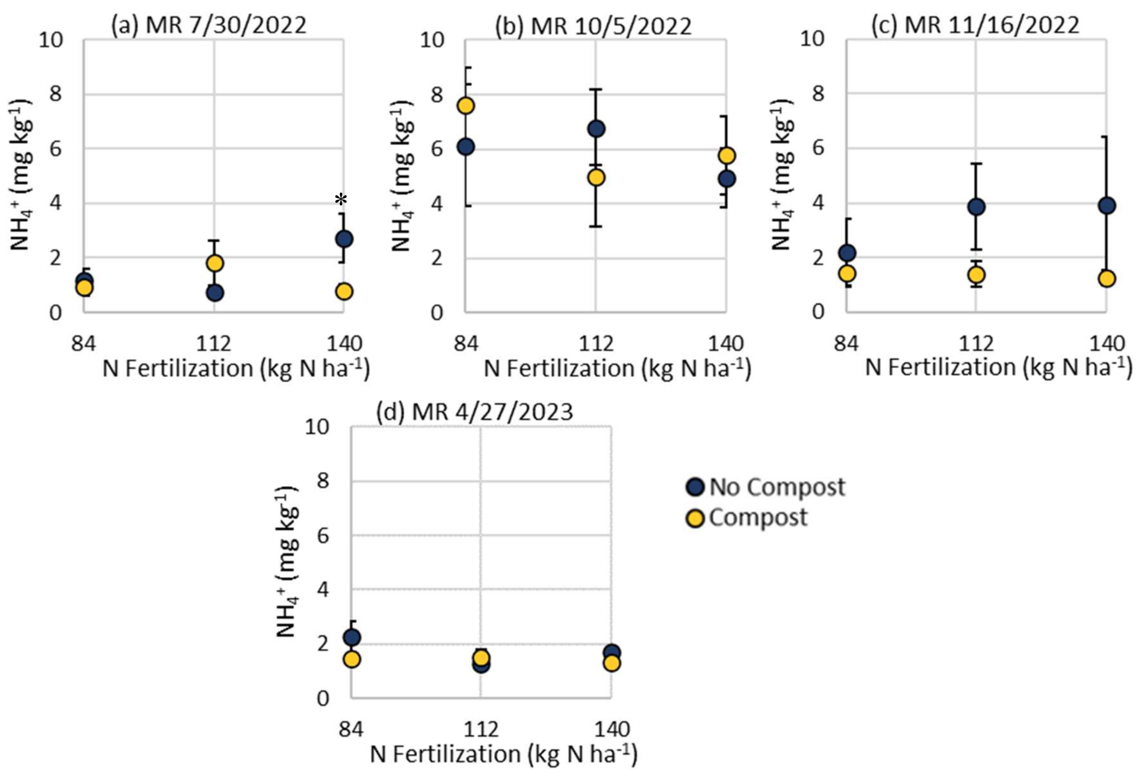

Figure 3.13. Soil NH_4^+ on (a) 7/30/2022, (b) 10/5/2022, (c) 11/16/2022, and (d) 4/27/2023 for the no compost controls and compost treatments at three fertilizer N rates (84, 112, and 140 kg N ha^{-1}) at MR. The line bars represent standard error (n = 4). Asterisk (*) (7/30/2022 only) indicates significant difference in NH_4^+ between no compost and compost treatments at a given N fertilization level, and no significant differences in NH_4^+ between N fertilization levels across a compost treatment were observed at a statistical significance level of alpha = 0.05.

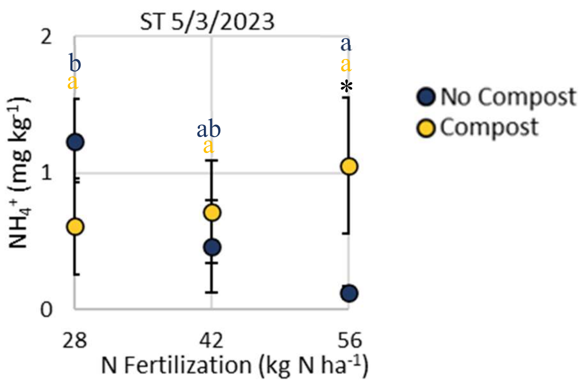

Figure 3.14. Soil NH_4^+ for the no compost controls and compost treatments at three fertilizer N rates (28, 42, and 56 kg N ha^{-1}) at ST on 5/3/2023. The line bars represent standard error (n = 4). Asterisk (*) indicates significant difference in NH_4^+ between no compost and compost treatments at a given N fertilization level, and lowercase letters indicate significant differences in NH_4^+ between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.

Soil NO_3^- concentrations at the MR and ST sites are shown in Figs. 3.15 and 3.16, respectively. At both sites, neither compost nor fertilizer N level had any statistically significant treatment effects on soil NO_3^-, and no consistent trends were observed.

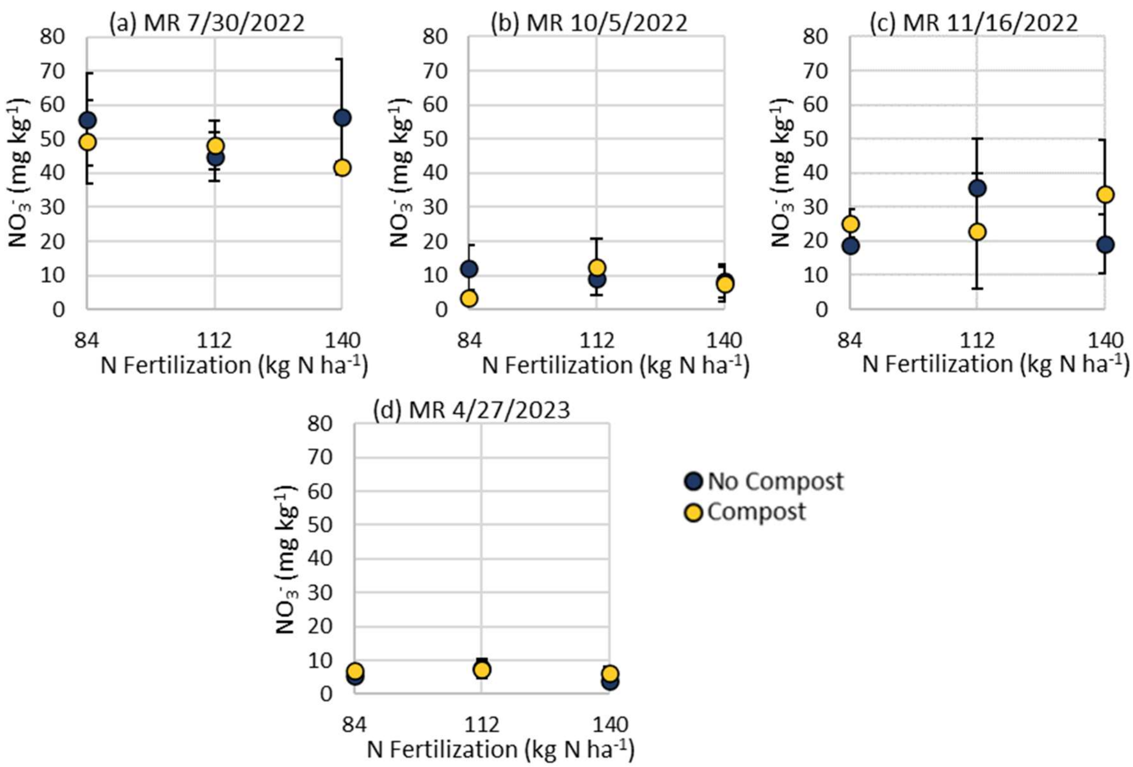

Figure 3.15. Soil NO_3^- on (a) 7/30/2022, (b) 10/5/2022, (c) 11/16/2022, and (d) 4/27/2023 for the no compost controls and compost treatments at three fertilizer N rates (84, 112, and 140 kg N ha^{-1}) at MR. The line bars represent standard error (n = 4). No significant differences between treatments were observed at a statistical significance level of alpha = 0.05.

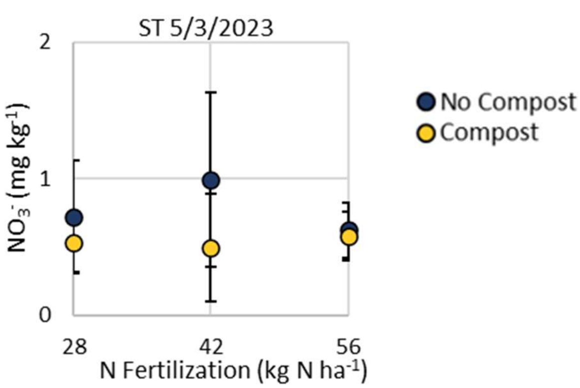

Figure 3.16. Soil NO_3^- for the no compost controls and compost treatments at three fertilizer N rates (28, 42, and 56 kg N ha^{-1}) at ST on 5/3/2023. The line bars represent standard error (n = 4). No significant differences between treatments were observed at a statistical significance level of alpha = 0.05.

ANOVA results with all sampling dates considered for the aforementioned metrics of olive yield, $\delta^{13}C$, iWUE, $\delta^{18}O$, soil GWC, NH_4^+, and NO_3^- are shown in Table 3.3. Compost had no significant treatment effect on any of the analyzed metrics at either site if all dates were considered. Fertilizer N level and the interaction between compost and N level had significant effects on $\delta^{13}C$ and iWUE at the MR site. Fertilizer N level did not significantly change any metric at the ST site. The interaction between compost and N level had significant effects on olive yield and soil NH_4^+ at the ST site.

Table 3.3. Effects of compost and fertilizer N level on olive yield, $\delta^{13}C$, iWUE, $\delta^{18}O$, soil GWC, NH_4^+, and NO_3^-. Pr (>F) values from ANOVA.

Site	Source	Yield	$\delta^{13}C$	iWUE	$\delta^{18}O$	GWC	NH_4^+	NO_3^-
MR	Compost	0.139	0.812	0.809	0.203	0.613	0.340	0.930
	N level	0.739	0.0370*	0.0368*	0.172	0.756	0.962	0.988
	Compost × N level	0.386	0.0211*	0.0211*	0.105	0.795	0.857	0.996
ST	Compost	0.610	0.0572	0.0558	0.607	0.330	0.369	0.513
	N level	0.201	0.426	0.419	0.603	0.665	0.319	0.883
	Compost × N level	0.0346*	0.307	0.311	0.725	0.347	0.0211*	0.434

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

3.3.4. Correlation matrices between plant and soil variables

The correlations between olive iWUE and $\delta^{18}O$, and various soil variables (GWC, NH_4^+, and NO_3^-) measured concurrently with plant sampling throughout the experiment at the MR site, are summarized in Table 3.4. Olive iWUE was significantly positively correlated with $\delta^{18}O$ and soil GWC, but did not show significant correlations with soil NH_4^+ or NO_3^-. Leaf $\delta^{18}O$ did not exhibit significant correlations with any of the measured soil variables. Intercorrelations among soil variables were observed only between NH_4^+ and NO_3^-, which were negatively correlated.

At the ST site, no statistically significant correlations between any plant or soil metrics were observed (Table 3.5). However, it should be noted that soil samples at the ST site were only collected on one date concurrent with plant sampling.

Table 3.4. Correlation matrix (r values) and statistical significance of correlations among the measured variables iWUE, $\delta^{18}O$, GWC, NH_4^+, and NO_3^- at the MR site.

Variable	iWUE	$\delta^{18}O$	GWC	NH_4^+	NO_3^-
iWUE	1.000				
$\delta^{18}O$	0.470***	1.000			
GWC	0.338**	0.125	1.000		
NH_4^+	-0.141	-0.170	0.126	1.000	
NO_3^-	-0.052	0.148	0.078	-0.244*	1.000

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

Table 3.5. Correlation matrix (r values) and statistical significance of correlations among the measured variables iWUE, $\delta^{18}O$, GWC, NH_4^+, and NO_3^- at the ST site.

Variable	iWUE	$\delta^{18}O$	GWC	NH_4^+	NO_3^-
iWUE	1.000				
$\delta^{18}O$	-0.157	1.000			
GWC	0.330	-0.070	1.000		
NH_4^+	0.065	-0.170	-0.025	1.000	
NO_3^-	0.254	0.022	0.129	0.185	1.000

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

3.5. Discussion

The goal of our study was to understand the effects of compost application and N fertilization on olive yield and iWUE. We hypothesized that compost would increase yields when applied with reduced fertilizer N inputs and would reduce water stress, thereby decreasing $\delta^{13}C$ and iWUE. Our results demonstrated that compost and fertilizer N level had a significant effect on olive yield at the ST site, but not at the MR site (Fig. 3.4). At the ST site, yields in the compost treatments decreased as the fertilizer N level increased, while the fertilizer N level did not affect yields in the no-compost control treatments. The only significant difference in yields between compost and no-compost treatments was observed at the lowest fertilizer N level, where compost significantly increased olive yield compared to the no-compost controls. It should be noted that the duration of this study was only one year, and the effects of compost and N fertilizer on yield may take years to develop. For example, in a study conducted in Spain, vegetative growth and fruit yield were not affected by N fertilizer treatments until the third year of experimentation (Centeno et al. 2017). Overall, the yields at the ST site were lower than those at the MR site. This was attributed to the trees at the ST site being in an 'off' year in their alternate bearing pattern (Lavee 2007) and to the site being more impacted by frost during the previous winter season, which decreased bud production.

Evidenced by the similar yield tonnage observed in the no-compost treatments across all fertilizer N levels at both field sites, reductions in fertilizer N below the recommended rate can be achieved without negatively affecting olive yield, regardless of whether or not compost is applied (Fig. 3.4). This finding indicates that in olive orchards, over-fertilization beyond the maximum yield efficiency point and the economic optimum fertilizer rate does indeed exist, as demonstrated by the fact that the addition of extra fertilizer did not increase yield. Based on our experimental design, application rates could be reduced to 84 kg N ha^{-1} at the MR site and to 28 kg N ha^{-1} at the ST site, without negatively impacting yield. Nitrogen overfertilization is common in olive orchards in the Mediterranean basin and California (Fernández-Escobar 2011). It is also recommended to apply N only when the previous year's leaf analysis indicates deficient N levels, in order to optimize the benefits of N fertilizer for yield and environmental sustainability (Fernández-Escobar 2011).

The effect of compost and N fertilizer on the $\delta^{13}C$ and iWUE of olive trees varied by site and sampling date. Compost tended to increase $\delta^{13}C$ and iWUE at the lower fertilizer N levels at both sites and decreased them at the highest fertilizer N level at the MR site, compared to the plots that received no compost (Figs. 3.5, 3.6, 3.7, and 3.8).

The observed treatment effects, with compost increasing $\delta^{13}C$ and iWUE at lower fertilizer N levels, suggest that the use of compost could increase plant water stress when reduced rates of fertilizer N are applied. Conversely, compost decreasing $\delta^{13}C$ and iWUE at the highest fertilizer N level at the MR site indicates that using compost in combination with N over-fertilization may decrease plant water stress.

There was a relatively lower occurrence of significant effects from compost and fertilizer N treatments on $\delta^{13}C$ and iWUE at the ST site compared to the MR site, which was unexpected. This was particularly so because the ST site received deficit irrigation, which was expected to induce greater plant water stress and thus provide more opportunity for treatments to generate a positive effect. The ST site also had lower overall iWUE than the MR site, indicating less plant water stress, which may be due to the higher total precipitation and older tree age and thus larger root zones for water uptake at the ST site, potentially masking treatment effects on iWUE. In contrast, the younger trees at the MR site may have allowed for more observable treatment effects because young trees are still growing and have greater water demands than mature trees (First Press 2006).

The trend of decreased iWUE with increasing fertilizer N rates in the plots that received compost at the MR site indicated that a N surplus can reduce the water stress caused by compost application, though no similar study exists in olive cropping systems for comparison. A study on Mediterranean barley and both bread and durum wheat found that increasing N fertilization increased WUE (Cossani et al. 2012), which contrasts with our results; although, their study focused on WUE which is the overall productivitiy of plants relative to water consumption, while the metric of iWUE used in our study is specifically in the context of photosynthesis and gas exchange within plants. A two-year study comparing different N levels and inorganic N sources on pot-grown olive cuttings found no significant differences in photosynthesis, stomatal conductance/leaf gas exchange, or transpiration (Leskovar and Othman 2019). However, our results suggest that different synthetic fertilizer N levels and compost do impact these tree physiological responses, therefore they changed $\delta^{13}C$ signature and iWUE, which are directly correlated to stomatal conductance (Grattan et al. 2006; Du et al. 2021). Our observations of compost occasionally increasing iWUE, as a proxy of water stress, was unexpected because compost has been reported to reduce direct soil evaporation and preserve water for crop transpiration (Taban and Movahedi Naeini 2006; Nguyen 2013). The observed increases in iWUE following compost application in our study could be potentially attributed to the high water-holding capacity of the compost, as evidenced by others that reported compost can hold between 3.50 to 4.40 g water g^{-1} dry compost

(Khater 2015). In our study, the compost was applied directly onto the soil surface without any tillage, and the irrigation drip lines were positioned above the compost layers. Consequently, this may have caused the water to be retained in the compost and the topsoil layer, limiting its infiltration into the root zone for plant uptake. Overall, if all sampling dates were considered, compost treatment alone did not have a significant effect on iWUE (Table 3.3). It is likely that compost could have a greater or different effect on water and iWUE if compost had been incorporated into the soil, rather than merely applying it on the surface (Nguyen 2013). Additionally, the soil texture at our field sites is worth noting. Both sites consisted of fine-textured clay soils. A previous study has indicated that compost application tends to have a more substantial effect on the water-holding capacity of coarser textured soils, while its impact on finer textured soils like clay is relatively minor (Brown and Cotton 2011). Therefore, replicating our study in olive orchards with coarser textured soils and with compost incorporation into the soil would potentially lead to more significant compost effects on iWUE, including decreased iWUE.

The unexpected effect of compost on iWUE results presents a challenge in interpretation, particularly given that our soil GWC results did not correspond with the typically observed effects of compost in preserving soil water (Taban and Movahedi Naeini 2006; Nguyen 2013). At both of our studied sites, we observed a general absence of significant treatment effects on soil GWC (Figs. 3.11 and 3.12). Nonetheless, it is recognized that WUE can be influenced by complex factors regulating plant growth and fruiting. These factors are generally closely tied to the water status and N availability, both of which are known drivers of plant WUE (Pascual et al. 2016). Therefore, we also examined leaf $\delta^{18}O$ and soil NH_4^+ and NO_3^- in our study to potentially provide insight into the iWUE results. These measures were chosen as leaf $\delta^{18}O$ is indicative of transpiration rates (Barbour 2007), and NH_4^+ and NO_3^- represent forms of N available to plants. We hypothesized that compost might influence leaf $\delta^{18}O$ through variations in stomatal conductance linked to soil water status or evapotranspiration from the soil surface. However, our observations did not reveal consistent effects of compost or fertilizer N on $\delta^{18}O$, aligning with previous findings in peach trees, suggesting that while $\delta^{18}O$ is a reliable indicator of the transpiration environment and biomass production at the tree level, it may not be effective in differentiating the effects of intra-seasonal changes in plant water or N status (Pascual et al. 2016). Soil NH_4^+ and NO_3^- concentrations remained low at both the MR and ST sites, and no consistent effects of compost or fertilizer N on NO_3^- concentrations were observed (Figs. 3.15 and 3.16). At the MR site, a notable treatment effect on NH_4^+ was only observed on the first sampling date, when compost treatments had lower NH_4^+ concentrations compared to the no-compost controls at the highest fertilizer N level (Fig. 3.13a). Conversely, at the ST site, soils treated with compost had significantly higher NH_4^+ concentrations compared to the no-compost controls at the highest fertilizer N level. However, it is important to note that at the ST site, soil was analyzed on only one sampling date (Fig. 3.14). Our findings on NH_4^+ concentration as affected by compost at the MR site align with those of another study on olive orchards, which compared olive waste and biosolid composts to a mineral fertilizer control (De Sosa et al. 2022). In their study, compost treatments also resulted in lower NH_4^+ concentrations than the control. Interestingly, they found that compost increased NO_3^- compared to the control after the second year of application (De Sosa et al. 2022). This contrasts with our findings, along with the differing results between our two field sites, indicating that different pathways in N mineralization may have occurred. It should also be noted that the study conducted by De Sosa et al. spanned three years of compost application (2022), whereas our study was conducted over a single year following a one-time compost application. The minimal treatment effects on NH_4^+ and NO_3^- observed in our study, along with the generally low soil N concentrations, could be attributed to the rapid utilization of plant-available N by the olive trees or microbial N uptake following fertilizer application. This indicates that a longer experimental duration might have been necessary to observe more pronounced treatment effects.

In our study, soil concentrations of NH_4^+ and NO_3^- did not show a significant correlation with plant iWUE or $\delta^{18}O$. However, at the MR site, soil GWC was found to be positively correlated with iWUE (Table 3.4). This finding suggests that soil water content plays a more significant role in regulating the iWUE of olives than the plant-available N content at this site. Additionally, olive iWUE was positvely correlated with $\delta^{18}O$ at the MR site (Table 3.4), which was unexpected since under water stress, stomatal conductance is low, resulting in higher $\delta^{13}C$ and thus higher iWUE, and transpiration is low, resulting in lower $\delta^{18}O$ compared to conditions of no water stress (Barbour 2007; Du et al. 2021). This unexpected positive correlation between iWUE and $\delta^{18}O$ at the MR site may be due to the irrigation water source (Clear Lake) being enriched in ^{18}O (Goff et al. 1993), which would lead to ^{18}O enrichment of plant biomass, thereby attenuating transpiration effects on $\delta^{18}O$. Interestingly, at the ST site, no significant correlations were observed among any of the measured variables iWUE, $\delta^{18}O$, GWC, NH_4^+, or NO_3^- (Table 3.5), although it should be noted that soil variables at this site were measured on only one sampling date. At the MR site, the significant role of soil water content in influecing olive iWUE was supported by the observation that significant compost treatment effects on $\delta^{13}C$, iWUE, and $\delta^{18}O$ were predominantly noted towards the end of the hot and dry summer season, and these effects disminished in November with the onset of the rainy season.

Notably, the lowest values of $\delta^{13}C$ and iWUE at the MR site were recorded on 11/16/2022, out of the five sampling dates, further suggesting that the November precipitation alleviated the water stress accumulated over the dry season, thereby diminishing any treatment effects on water stress.

3.6. Conclusion

Compost and fertilizer N level treatments had variable effects on olive yield, $\delta^{13}C$, intrinsic water use efficiency (iWUE), $\delta^{18}O$, and soil gravimetric water content (GWC), ammonium (NH_4^+), and nitrate (NO_3^-) across our two super high-density olive field sites in California. Compost increased yield compared to no-compost controls when used along with reduced fertilizer N inputs. Compared to no-compost controls, compost tended to increase olive $\delta^{13}C$ and iWUE at lower fertilizer N levels, indicating plant water stress. Conversely, compost decreased $\delta^{13}C$ and iWUE at the highest fertilizer N level, indicating that additional N can decrease the plant water stress caused by compost application. Compost use increasing water stress was unexpected because compost typically improves soil water retention for crop use. However, in our study, there was generally a lack of significant compost effects on soil GWC. There were also no consistent compost or fertilizer N effects on olive $\delta^{18}O$, a proxy for transpiration, and little to no treatment effects on soil NH_4^+ or NO_3^-. Soil NH_4^+ and NO_3^- did not show significant correlation with plant iWUE or $\delta^{18}O$, whereas soil GWC was positively correlated with iWUE at one site. Our results suggest that soil water content played a more significant role in regulating olive iWUE than plant-available N content. Older tree age and higher precipitation also reduced treatment effects on plant water stress outcomes. To our knowledge, this study is the first investigation of compost and fertilizer N management effects on iWUE in olive cropping systems. The information presented in this work may be used to provide guidance on compost and fertilizer N management in super-high density olive orchards in California, with potential extension to other global Mediterranean regions, and improves the understanding of consequences of compost and fertilizer N on soil water status and N availability, both of which impact iWUE through complex regulation of plant physiological responses. Furthermore, this study highlights the need for longer field trials to investigate long-term compost and fertilizer N effects. Future research should also involve compost incorporation into the soil instead of surface application and treatment applications to a range of soil textures to optimize treatment effects and refine understanding of compost impacts on olives grown in other soil types.

3.7. References

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3.8. Supplementary information

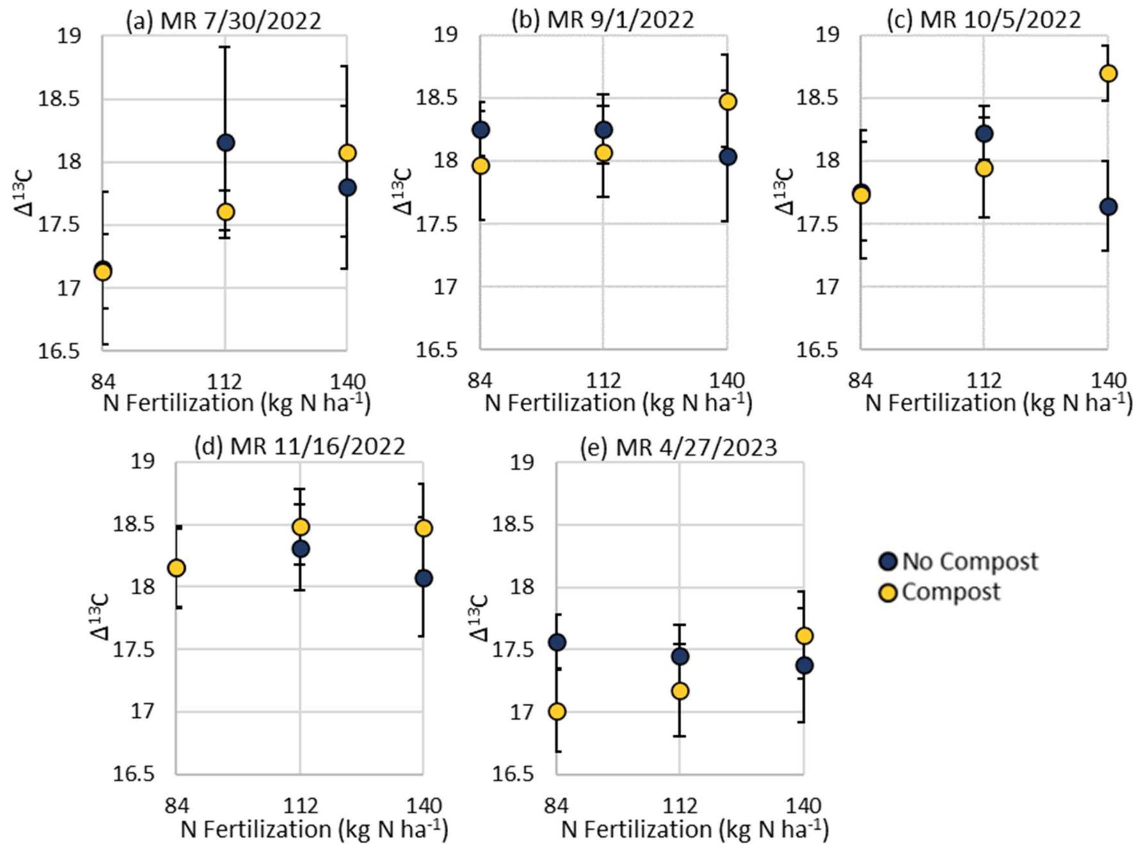

Figure S3.1. Leaf Δ^{13}C on (a) 7/30/2022, (b) 9/1/2022, (c) 10/5/2022, (d) 11/16/2022, and (e) 4/27/2023 for the no compost controls and compost treatments at three fertilizer N rates (84, 112, and 140 kg N ha^{-1}) at MR. The line bars represent standard error (n = 4). Asterisks (*) indicate significant difference in Δ^{13}C between no compost and compost treatments at a given N fertilization level, and lowercase letters indicate significant differences in Δ^{13}C between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.

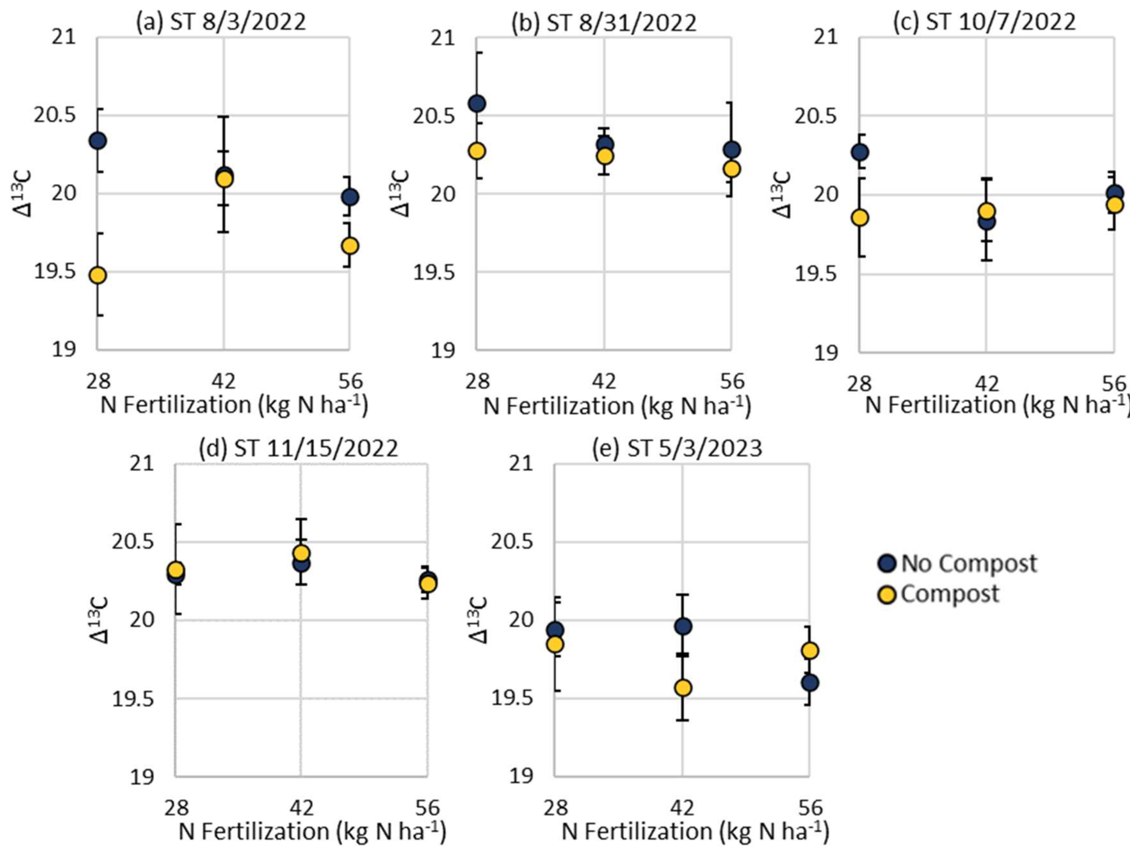

Figure S3.2. Leaf $\Delta^{13}C$ on (a) 8/3/2022, (b) 8/31/2022, (c) 10/7/2022, (d) 11/15/2022, and (e) 5/3/2023 for the no compost controls and compost treatments at three fertilizer N rates (28, 42, and 56 kg N ha^{-1}) at ST. The line bars represent standard error (n = 4). Asterisks (*) indicate significant difference in $\Delta^{13}C$ between no compost and compost treatments at a given N fertilization level, and lowercase letters indicate significant differences in $\Delta^{13}C$ between N fertilization levels across a compost treatment at a statistical significance level of alpha = 0.05.