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ECOSYSTEMS

Impacts of Compost Amendment Type and Application Frequency on a Fire-Impacted Grassland Ecosystem

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Abstract

Composting organic matter can lower the global warming potential of food and agricultural waste and provide a nutrient-rich soil amendment. Compost applications generally increase net primary production (NPP) and soil water-holding capacity and may stimulate soil carbon (C) sequestration. Questions remain regarding the effects of compost nitrogen (N) concentrations and application rates on soil C and greenhouse gas dynamics. In this study, we explored the effects of compost with different initial N quality (food waste versus green waste compost) on soil greenhouse gas fluxes, aboveground biomass, and soil C and N pools in a fire-impacted annual grassland ecosystem. Composts were applied annually once, twice, or three times prior to the onset of the winter rainy season. A low-intensity fire event after the first growing season also allowed us to explore how compost-amended grasslands respond to burning events, which are expected to increase with climate change. After four growing seasons, all compost treatments significantly increased soil C pools from 9.5 ± 0.9 to 30.2 ± 0.7 Mg C ha^{-1} (0–40 cm) and 19.5 ± 0.9 to 40.1 ± 0.7 Mg C ha^{-1} (0–40 cm) relative to burned and unburned controls, respectively. Gains exceeded the compost-C applied, representing newly fixed C. The higher N food waste compost treatments yielded more cumulative soil C (5.2–10.9 Mg C ha^{-1}) and aboveground biomass (0.19–0.66 Mg C ha^{-1}) than the lower N green waste compost treatments, suggesting greater N inputs further increased soil stocks. The three-time green waste application increased soil C and N stocks relative to a single application of either compost. There was minimal impact on net ecosystem greenhouse gas emissions. Aboveground biomass accumulation was higher in all compost treatments relative to controls, likely due to increased water-holding capacity and N availability. Results show that higher N compost resulted in larger C gains with little offset from greenhouse gas emissions and that compost amendments may help mediate effects of low-intensity fire by increasing fertility and water-holding capacity.

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INTRODUCTION

Grasslands comprise more than 30% of the global ice-free land surface (O'Mara 2012; Conant and others 2017). Grassland soils are widely recognized for their potential to sequester soil carbon (C) due to their extensive land area and high belowground allocation of biomass (Conant and others 2001; Conant and Paustian 2002; Lorenz and Lal 2018). Rangeland management practices on grassland soils often lead to a loss of soil C over time (Sanderman and others 2017), making grassland soil management a key target for practices that could increase the movement of C from the atmosphere to the soil and replenish soil C stocks (Sanderson and others 2020).

Grassland C sequestration has been proposed as a mechanism to help mitigate climate change (Smith 2008; Soussana and others 2010; Mayer and others 2018; Mayer and Silver 2022). Compost amendments are one mechanism to increase organic C sequestration in grasslands soils through the direct addition of C in compost and the indirect addition of C due to enhanced primary productivity (Ryals and Silver 2013; Kutos and others 2023). Composting transforms organic byproducts into more biologically stable materials that act as slow-release sources of plant-available nutrients. Compost application also changes the soil physical, chemical, and biological properties that control nutrient availability years after amendment application (Brinton 1985; Shiralipour and others 1992; Ryals and Silver 2013; Ryals and others 2015). Previous studies utilizing composted organic amendments to rangelands have generally only used biosolids, manure and green waste-based feedstocks (Kutos and others 2023).

Food waste may be a particularly valuable feedstock for composted organic matter amendments. Reducing and repurposing food waste is likely to decrease emissions from the global food system, with food waste representing over 10% of food system emissions in high-income nations (Read and others 2020; Crippa and others 2021). Using food waste to produce soil amendments such as compost can provide a mechanism to reutilize C and nutrients as organic fertilizer while lowering greenhouse gas emissions from landfill disposal (Favoino and Hogg 2008). This may also increase soil C sequestration when applied as compost amendments, as well as reduce erosion and improve soil nutrient retention, soil water-holding capacity, drought resilience, and yields (Sullivan and others 2002, 2003; Lee and others 2004; Ohsowski and others 2012; DeLonge and others 2013; Kutos and others 2023). However, previous studies of food waste compost applications generally focus only on N availability and yield (Sullivan and others 2002, 2003, 2019; Lee and others 2004), limiting our understanding of its potential for soil C sequestration.

Food waste compost generally has higher N, phosphorus (P), and potassium (K) concentrations relative to other composted organic matter amendments (Lee and others 2019; O'Connor and others 2021). This increase in nutrient availability may stimulate soil microbial activity, altering soil oxidation–reduction (redox) conditions and increasing the production of nitrous oxide (N_2O) and methane (CH_4), both potent greenhouse gases (Oertel and others 2016). Understanding the impact of food waste compost on greenhouse gas fluxes relative to other composted organic amendments is critical for determining the net climate impact of these practices.

The effects of compost on ecosystem biogeochemical cycling are likely to be affected by the frequency of amendment application. Repeated annual compost applications (that is, more than one application) may stimulate ecosystem productivity through enhanced nutrient availability, and increase soil C stocks by providing additional organic matter and nutrient inputs into soil ecosystems (Paustian and others 1992; Ryals and Silver 2013; Cesarano and others 2017; Mayer and Silver 2022). Repeated compost amendments may also lower emissions from organic waste streams such as landfilling (Hall and others 2022). If added too frequently, compost amendments could potentially increase leaching losses, nutrient imbalances (Li and others 2020; Xu and others 2022), and cause plant yields to plateau or decline (Chen and others 2018; Zhang and others 2022), although likely at much lower rates than raw manure or inorganic fertilizers (Fan and others 2017; Siedt and others 2021). An understanding of the effects of compost application frequency is needed to determine optimal land management practices. This is particularly true in the context of real-world conditions that incorporate background climate and disturbance, particularly the effects of fire in grassland ecosystems.

Fire frequency is expected to increase with climate change in grasslands worldwide, likely altering the magnitude and patterns of C cycling in these ecosystems (Lei and others 2016). Fire is a natural component of many grassland ecosystems, but increased fire frequency can remove surface layer soil organic matter and aboveground biomass resulting in a loss of soil C over time (Pellegrini and others 2018). However, fire also increases soil inputs of pyrogenic C, a form that can persist over long timescales (Pingree and DeLuca 2017). Fire events may also change soil properties, including soil porosity, aggregate formation, hydrophobicity, and pH, all of which can also influence soil C cycling (Certini and others 2011; Jiménez-Morillo and others 2016). Composted organic matter amendments, applied either before or after a fire event, may help grasslands recover from fire by increasing soil organic matter content and associated nutrient availability. Composted organic matter can also improve soil water-holding capacity after fire, important for ecosystem recovery (Conant and Paustian 2002; Teague and Barnes 2017). Compost applied prior to a fire event may lead to additional pyrogenic C formation and nutrient inputs, including increased inorganic nitrate (NO_3^-) and ammonium (NH_4^+) availability, facilitating ecosystem recovery and C storage (Pingree and DeLuca 2017).

In this study, we compared the effects of two composts with similar initial total C and N concentrations but different mineral N (NH_4^+ and NO_3^-) concentrations on soil C storage and biogeochemistry in a fire-impacted annual grassland soil over a four-year period. We hypothesized that food waste compost would stimulate increased net soil C sequestration relative to green waste amendments due to increased aboveground biomass from higher N availability, but that these gains would be significantly offset by higher N_2O emissions. We also hypothesized that increased frequency of amendment application would promote the greatest net increases in soil C and N stocks and aboveground biomass due to greater nutrient and C inputs. Finally, we hypothesized that compost amended sites, applied both before and after a fire event, would have greater overall C stocks following fire than unamended sites due to faster recovery of aboveground biomass with compost additions. We quantified soil greenhouse gas emissions (CO_2, CH_4, and N_2O) during key management and climatic events and measured pH, mineral N dynamics, and C and N stocks in biomass and soil over time.

Methods

Field Site

The site was located at the University of California Sierra Foothill Research and Extension Center (SFREC) in Browns Valley, California. Soils are derived from Mesozoic and Franciscan volcanic rock and classified as xeric Inceptisols and Alfisols in the Auburn-Sobrante complex (Soil Survey Staff 2020). The site had been grazed by cattle for at least 150 years (D. Flavell, personal communication). Average annual precipitation was 700 mm with pastures producing on average 3,300 kg ha^{-1} y^{-1} of biomass dominantly used for livestock forage. The study region has a Mediterranean climate where the growing season typically occurs from October to April or May and is characterized by cool, wet winters and warm dry summers. The site was dominated by naturalized stands of annual grasses and forbs (Bartolome and others 2007; Eviner 2016). The field sites were not seeded, irrigated, fertilized, or tilled. Naturalized annual plant species reseed and replace stands every year. This ecosystem is broadly representative of approximately 12 million hectares of rangeland across California (Eviner 2016).

Experimental Design

In October 2018, nine original 0.15 ha (60 × 25 m) plots were established with 6 m buffers to establish a randomized complete block design with three complete blocks. Each block contained a food waste compost treatment with applications in two years (FW or FW2), a manure and green waste compost treatment with applications in three years (hereafter referred to as green waste compost and GW, GW2, or GW3), and an untreated burned control (UCN); an unburned control (CN) was added in spring 2019 (Table 1). The food waste compost had higher mineral N (80 µg g N^{-1}) than the green waste compost (57 µg g N^{-1}), but the composts had similar total C and N (Tables S1–S2). To apply the second compost treatments in Fall 2020 (FW2, GW2), original plots, except the controls, were split into 0.075 ha (60 × 12.5 m) plots for all treatments (n = 3 per treatment for CN, FW, FW2, GW, GW2). To apply the third compost treatments in Fall 2022 (GW3), the GW2 plots were split in half again to create 0.0375 ha (30 × 12.5 m) plots for GW2 (n = 3) and GW3 (n = 3) treatments. In June 2019 shortly after soil and plant samples were collected, a fast low-intensity grass fire burned all plots evenly.

Table 1. Description of Soil Amendment, Application, and Burned or Unburned Control Treatments

Treatment	Description	Number of applications	Application years	Soil sampling years
Control (CN)	No treatment applied, Fire in June 2019	–	–	1. Fall 2018 2. Spring 2019 3. Spring 2020 4. Spring 2021 5. Spring 2022
Unburned Control (UCN)	No treatment applied, no fire during study	–	–	2. Spring 2020 3. Spring 2021 4. Spring 2022
Green waste compost (GW, GW2, GW3)	Compost derived from plant husks, chicken, horse, and cattle manures. Fire in June 2019	GW: one-time GW2: two-time GW3: three-time	Fall 2018 Fall 2019 Fall 2021	1. Fall 2018 2. Spring 2019 3. Spring 2020 4. Spring 2021 5. Spring 2022
Food waste compost (FW, FW2)	Compost derived from food scraps including fruits, vegetables, egg/clam shells, bones, etc. Fire in June 2019	FW: one-time FW2: two-time	Fall 2018 Fall 2019	1. Fall 2018 2. Spring 2019 3. Spring 2020 4. Spring 2021 5. Spring 2022

Following the burning event, unburned controls (UCN) 0.15 ha (60 × 25 m) plots were established approximately 80 m from the closest burned plot and sampled identically thereafter to further elucidate the effects of fire across treatments. These treatment groups allowed us to simultaneously compare the effects of green waste versus food waste compost, the effects of compost application frequency, and the effects of fire on a composted grassland ecosystem. Fire is a natural and common event in these grasslands (Harrison and others 2003).

Established plots were located on similar slope and aspect; management and soil type (pre-treatment C:N of 12) were consistent across the whole study area. Treatments were randomly assigned within each block. Both green and food waste composts were produced at the West Marin Compost Facility (for compost properties see Tables S1–S2) by maturing in watered piles that were turned (aerated) weekly for three months (Vergara and Silver 2019; Pérez and others 2023). Food waste compost (C:N of 16) was derived from food scraps including fruits, vegetables, egg and clam shells, bones, etc. Green waste compost (C:N of 16.4) was derived from plant residues, chicken, horse, and cattle manures. Both feedstocks were mixed with woodchips in a matrix that was 50% by volume to facilitate proper compost development and maturation. All plots were grazed by yearling steers for approximately three weeks and then mowed to a uniform aboveground cover prior to initial compost application. During grazing, cattle were allowed to graze all plots freely and not isolated to any specific plot. Composts were applied at a depth of 0.65 cm (equivalent to 5.9 Mg C ha^{-1}/0.37 Mg N $ha^{-1}$and 5.5 Mg C ha^{-1}/0.34 Mg N ha^{-1} for food waste and green waste compost per application, respectively) to respective treatment plots in November 2018, October 2019, and October 2021 using a compost spreader (application dates based on best practices by the range manager). Control plots were driven over without amendment application to impart the same soil effect.

Soil Sampling and Analyses

Soil sampling was conducted prior to compost application (fall 2018) and annually at the end of each growing season (end of spring) in 2019, 2020, 2021, and 2022. At every sampling time point, soil samples (n = 9 per plot regardless of plot size) were collected at 0–10 and 10–20, and 20–30 cm depths from nine stratified random locations per plot using a soil auger. When possible, soil samples were also collected from 30–40 and 40–50 cm depths. Due to the inability to sample 40–50 cm in some locations, these depths were not included in soil C stock calculations but are provided in the *Supplementary Information* (Table S3). All compost addition subplots were sampled at least 5 m from the edge of each subplot boundary to minimize edge effects between treatments with application frequency. Samples were transported to the laboratory and processed for analysis within 24 h. Soil pH was measured in a 1:1 volumetric slurry of sample and deionized water using a pH electrode (Mclean 1982). Soil moisture was determined gravimetrically by weighing fresh soil and oven-drying for at least 24 h at 105 °C. Mineral N species and N mineralization rates were quantified throughout the first growing season (early, mid late growing season, and end of growing season) and during the last annual soil sampling (end of growing season 2022). Nitrate plus nitrite (NO_2^-) and NH_4^+ were measured after extraction of 15 g of field-fresh soil in 75 ml of 2 M potassium chloride (KCl) solution (Hart and others 1994). Potential net nitrification and mineralization were measured by comparing fresh 2 M KCl soil extractions with a second subsample was covered and incubated for seven days in the dark, prior to subsequent 2 M KCl soil extractions. Soil KCl extracts were stored at − 20 °C until colorimetrically analyzed using an AQ300 analyzer (Seal Instruments, Mequon, WI). The difference in NO_3^- over time was used to calculate potential net nitrification, and the difference in the sum of NO_3^- and NH_4^+ concentrations over time was used to calculate potential net mineralization, after accounting for the length of the incubation (Hart and others 1994).

For total soil C and N analyses, subsamples were air-dried, sieved to < 2 mm, and had visible roots removed before being ground to a fine powder. Samples were then analyzed in duplicate for total C and N on a CE Elantech elemental analyzer (Lakewood, NJ). Bulk density was sampled in 2020 from three locations in each plot using a 6.5 cm diameter bulk density corer. Samples were sorted in the lab into fine soil (< 2 mm) and coarse rock (> 2 mm) volumes. To quantify soil moisture, soil subsamples from each depth and replicate were weighed before and after drying at 105 °C to a constant weight for at least 24 h. Bulk density was calculated as the rock-free dry volume for total soil fractions, and used to calculate total C and N stocks (g C m^{-2}, g N m^{-2}). Previous work at this site suggested that bulk density was relatively similar following compost amendments over one-to-four-year time scales (Ryals and others 2014).

Soil Greenhouse Gas Fluxes

Fluxes of CO_2, N_2O, and CH_4 were measured across the soil-atmosphere interface using the static chamber method in years one and three; they were not measured in year two (2020) due to complications from the COVID-19 pandemic. Three chambers were sampled in each plot. Gas sampling took place at daily intervals after the initial compost application during the spring wet up period for 8–21 days, and over the entire growing season at bi-weekly intervals. Gas samples (30 mL) were collected from each chamber at 0, 5, 15, 25, and 40 min after the lid closure and stored in pre-evacuated gas vials until analysis (within 72 h) for CO_2, N_2O, and CH_4 on a Schimadzu GC-14 gas chromatograph (Shimadzu Corporation, Kyoto, Japan) equipped with a thermal conductivity detector, flame ionization detector and an electron capture detector. Analyzer detection limits were 0.09 ppm CH_4, 0.49 ppb for N_2O, and 0.09 ppm for CO_2. Fluxes were calculated using an iterative exponential curve-fitting approach with non-statistically significant fits considered as fluxes equal to zero (Matthias and others 1978; Ryals and Silver 2013).

Aboveground Biomass

Aboveground vegetation was sampled for biomass production and C and N content at peak standing crop at the end of each growing season (n = 9 samples per plot per growing season). Aboveground shoots were clipped from 20 cm diameter circles, collected into pre-dried bags, dried at 65 °C for > 24 h, and subsequently weighed. Dried and ground subsamples were analyzed for C and N on a CE Elantech elemental analyzer (Lakewood, NJ).

Statistical Analyses

Response variables were analyzed using JMP 16 (SAS Institute Inc., Cary, NC). Soil and biomass data were analyzed using a linear mixed model with block as a random effect and treatment (green and food waste composts and control) and time (2019: year one, 2020: year two 2021: year three, and 2022: year 4) as fixed effects. For significant treatment effects on soil C and N, plant productivity, and greenhouse gas fluxes, all treatment groups (FW, FW2, GW, GW2, GW3) and control groups (CN, UCN) means were compared using a Tukey's post-hoc test. As aboveground plant biomass exhibited high interannual variability, biomass data were further assessed for significant trends using mean change in plant biomass (treatment–control) and applying a one-sample t-test for the green waste compost treatment and for food waste compost. Values reported in the text are means and standard errors.

Results

Effects of Compost Type

All green waste (GW, GW2, GW3) and food waste (FW, FW2) compost applications increased soil C and N stocks over time relative to both controls. After four growing seasons, all compost treatments corresponded to at least a total of 9.5 ± 0.9 Mg C ha^{-1} (9.5 ± 0.9 to 30.2 ± 0.7 Mg C ha^{-1}) increase in soil C at 0–40 cm depth relative to the burned control (CN) and at least 19.5 ± 0.9 Mg C ha^{-1} (19.5 ± 0.9 to 40.1 ± 0.7 Mg C ha^{-1}) increase relative to the unburned (UCN) control (Figure 1, $p < 0.03$). Similar observations were observed for soil N, all compost treatments corresponded to an increase of up to 2.2 ± 1.0 Mg N ha^{-1} (0.5 ± 0.5 to 2.2 ± 1.0 Mg N ha^{-1}) increase in soil N at 0–40 cm relative to the burned (CN) and up to 2.9 ± 1.0 Mg N ha^{-1} (1.3 ± 0.5 to 2.9 ± 1.0 Mg N ha^{-1}) relative to the unburned (UCN) controls (Figure 1, $p < 0.03$). Soil C and N concentrations and stocks within entire soil profiles (0–40 cm) did not differ significantly by treatment prior to compost amendments or at the end of year 1 (Figure S1). By the end of the second growing season (2020), soil C and N concentrations and stocks were significantly higher than both the burned and unburned control at each soil depth in the FW treatment (Figure 1, $p < 0.05$). The GW treatment showed a similar trend of higher soil C and N concentrations in surface soils (0–10 cm) in the second and third years (Figure 1, $p < 0.05$). Soil C and N concentrations and stocks were significantly lower in 10–20 cm compared to 0–10 cm at all sampling periods and treatments (Figure 1, $p < 0.01$).

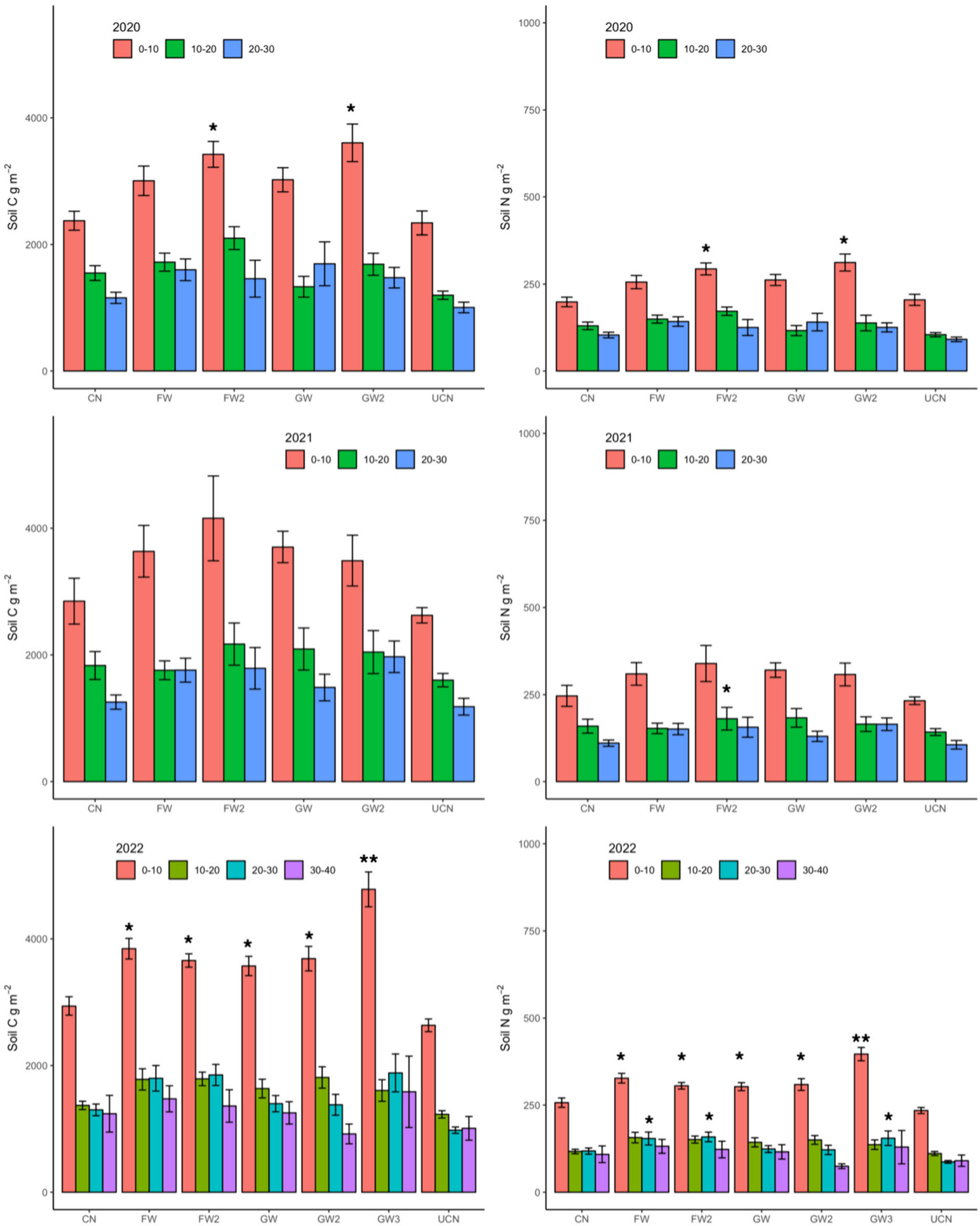
Patterns in soil C and N varied across years and depths. In the burned control (CN) treatment, soil C and N concentrations and stocks in the 0–10 cm depth decreased significantly in year two relative to pre-treatment and year one soils (Figures 1 and S1, $p < 0.05$). In the GW treatment, soil C and N stocks significantly decreased in the top 0–10 cm in year one relative to the pre-treatment values (Figure S1). Soil N stocks significantly increased in year one at 10–20 cm compared to pre-treatment in GW plots and returned to pre-treatment levels by the end of the second growing season (Figure 1). In the FW treatment, soil C concentrations and soil C and N stocks significantly increased over time in both 0–10 and 10–20 cm soil depths (Figure 1). Trends in soil C stocks in 20–30 cm depths were variable across years, with significant differences only observed at the end of four growing seasons.

Figure 1. Soil carbon (C) and nitrogen (N) stocks ▶ (mean ± standard error) in 0–10, 10–20, 20–30, and 30–40 cm depths. Standard error bars are representative of the standard error of the entire depth profile, accounting for standard error associated with each individual sample depth. *FW* food waste compost, *GW* green waste compost, *CN* untreated control, and *UCN* unburned control; *FW2 and GW2* two annual applications for green waste and food waste, respectively; *GW3* three annual applications for green waste. End of year two (May 2020) (top), end of year three (May 2021) (middle), and end of year 4 (bottom). Statistically differences ($p < 0.05$) from CN and UCN across individual depths and years were marked with * and marked with ** when statistically different than all other treatments ($p < 0.05$).

Soil pH did not differ significantly across treatments prior to compost amendments. At the end of the first growing season the FW and GW treatments had slightly higher soil pH than both burned and unburned controls, although the differences were only statistically significant in the GW treatment (Table S4, $p < 0.05$). While we observed a similar trend in the second year, the differences were not statistically significant (Table S4). Soil pH significantly increased in the first growing season compared to pre-treatment in both treatments and control. In the second year, pH returned to pre-treatment values, except for a decrease in the control plots.

In the first two growing seasons, mineral N concentrations did not vary significantly by treatment within sampling dates (0–10 cm, Figure S2). When data were pooled across treatments soil NH_4^+ concentrations were significantly lower in the mid and late growing season, whereas NO_3^- decreased in the early growing season then increased to approximately pre-treatment levels by the mid and late growing season (Figure S3). At the end of the fourth growing season, the one-time green waste (GW) treatments exhibited significantly greater NO_3^- concentrations than all other treatments, while both control (CN) and one-time green waste (GW) treatments had significantly greater NH_4^+ concentrations (Figure S4).

Potential net nitrification and mineralization varied across treatments and over time. Compost amended plots immobilized N early in the first growing season (Figure S5). Late in that growing season GW and GW2 treatments exhibited net NO_3^- immobilization and net N immobilization (Figure S5). Potential net nitrification significantly decreased in the late growing season after the first compost appli-


2020
0-10
10-20
20-30
Soil C g m^{-2}
Soil N g m^{-2}
CN
FW
FW2
GW
GW2
UCN
2021
2022
30-40
GW3

cation in all treatment plots (Figure S5). Similar trends at the end of the first growing season were observed at the end of the fourth growing season (Figure S6).

Summed over four growing seasons, all compost-amended treatments produced significantly more aboveground biomass relative to the control (CN) treatment (increase of 100 to 258 g m^{-2}, Figure 2, $p < 0.03$). Both FW and FW2 treatments produced 138–158 g m^{-2} more aboveground biomass than both GW and GW2 treatments, respectively, over the four-year period (Figure 2, $p < 0.05$). During the first year following compost amendments, GW treatments had significantly more aboveground biomass than the controls (Table S5). At the end of the third growing season, all food waste (FW, FW2) compost plots had significantly more aboveground biomass than both controls (CN, UCN). Within all treatments, aboveground biomass decreased throughout the experiment ($p < 0.03$). There were no statistically significant differences in biomass C and N concentrations across treatments (C = 42.6 ± 0.1% and N = 1.14 ± 0.01%).

We measured daily greenhouse gas fluxes during the first rain event following compost amendments to determine short-term dynamics and biweekly measurements in years one and three of the study. Annual CO_2 fluxes varied over time, with significantly higher CO_2 fluxes observed in GW in year one (Figure 3A, $p < 0.03$). There were no significant effects of treatments on CH_4 fluxes in either year. Soil N_2O fluxes were significantly higher in the first year following compost amendments in the FW treatment (+ 0.23 ± 0.02 mg N_2O m^{-2} d^{-1}) (Figure 3A, $p < 0.001$). During both years, we observed a hot moment of N_2O flux shortly after the first seasonal rain event, particularly in the FW treatment (Figures S8–S9).

Effect of the Number of Applications

At the end of the final growing season, soil C and N stocks were higher in the GW treatment with three applications (GW3). This GW3 treatment contained significantly higher soil C and N stocks (32.6 ± 1.0 Mg C ha^{-1}, 8.2 ± 1.0 Mg N ha^{-1}) than all other treatments and controls (Figure 1, $p < 0.01$). There were no statistically significant differences in soil C and N concentrations and stocks between one-time and two-time compost additions in both food waste and green waste treatments (Figure 1). Multi-year compost amendments consistently increased both CO_2 and N_2O fluxes in the growing season following amendment application (Figure 3, $p < 0.01$). In year two, GW2 and FW2 treatments had significantly higher CO_2 (Figure 3B, $p < 0.01$) and N_2O fluxes (Figure 3B, $p < 0.01$), while both GW and FW treatments had lower N_2O fluxes (Figure 3B, $p < 0.01$) relative to the burned control (CN).

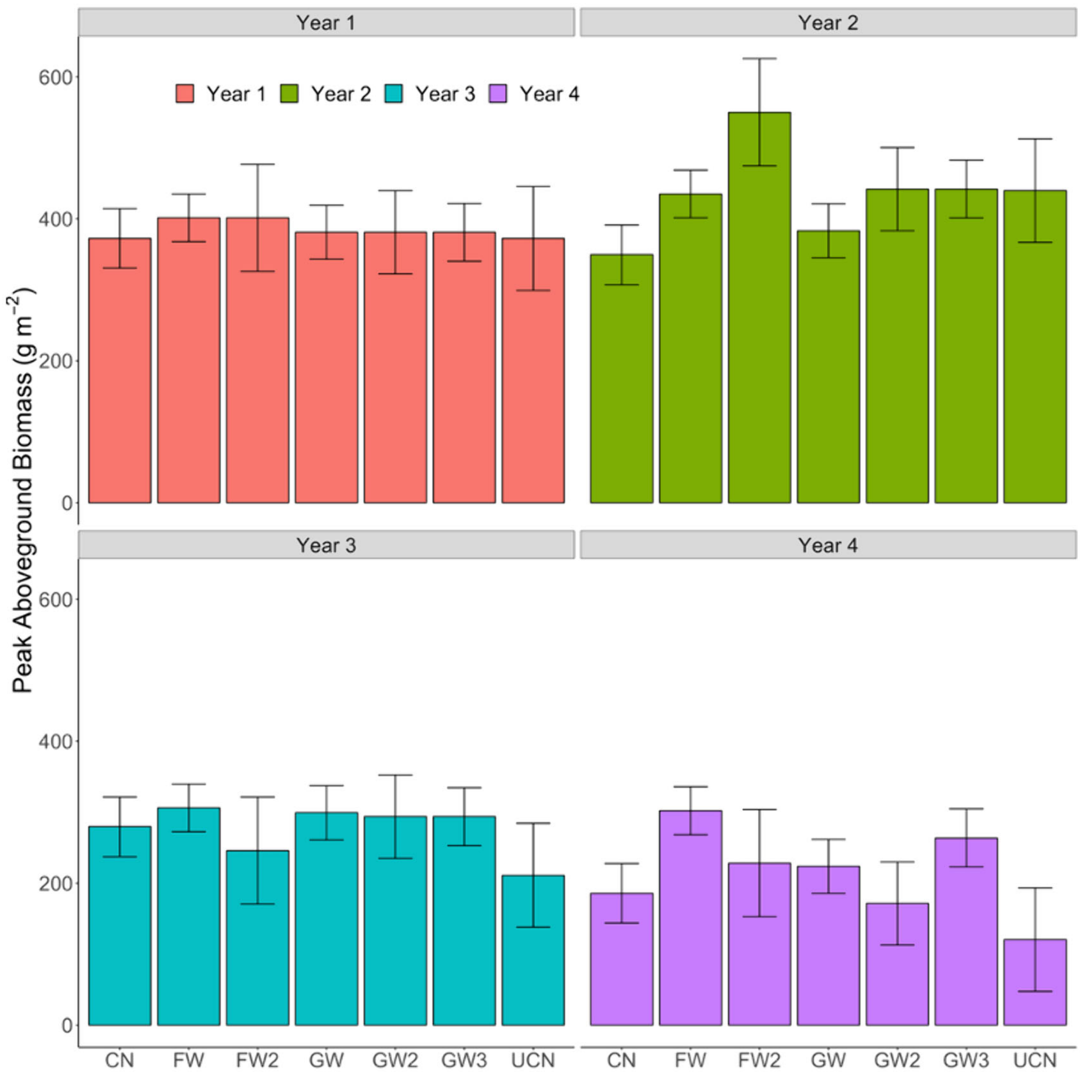
At the end of the fourth growing season, soil pH was significantly higher across depths in the GW2 and GW3 application treatments relative to all other treatments (Table S4, $p < 0.01$). Soil pH was significantly lower across depths in the FW treatment relative to the control and the GW3 treatment (Table S4, $p < 0.01$). The FW2 treatment also exhibited a significantly higher pH at 0–10 cm than all one-time treatments and both burned and unburned controls (Table S4, $p < 0.01$).

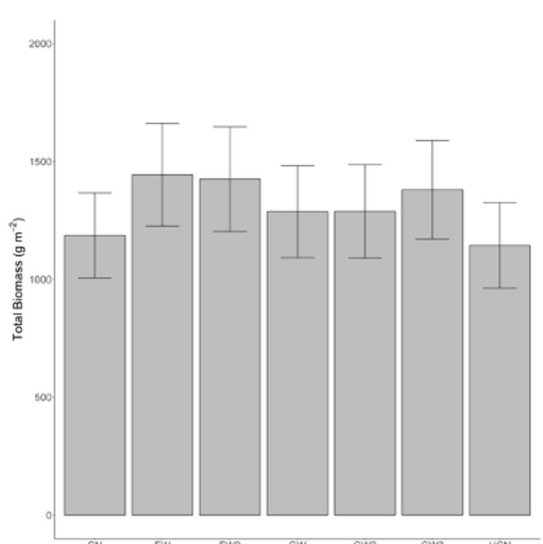
There was high interannual variability in aboveground biomass in all treatments and controls, typical of California annual grasslands (Alexander and others 2023). After the second growing season, the FW2 treatment had significantly higher biomass than both the GW and burned control treatment (CN). However, following the third growing season aboveground biomass in FW2 treatment was significantly lower than both FW and GW treatments. During the final growing season, the FW treatment had the highest biomass observations of any individual treatment and was significantly higher than both the GW2 and burned control treatment (CN).

Figure 2. Annual peak aboveground biomass (g m^{-2}) over four growing seasons (2019 = red, 2020 = green, 2021 = teal, and 2022 = purple) and total over the experiment (grey). *FW* food waste compost, *GW* green waste compost, *FW2 and GW2* two annual applications for green waste and food waste, respectively; *GW3* three annual applications for green waste.

Effect of Fire

The fire event following the first complete growing season allowed us to further elucidate the effect and interactions of fire with compost amendments. Soil C and N stocks and concentrations were significantly higher in all burned treatments relative to the unburned control (Figure 1). In the final growing season, biomass was also significantly higher in all burned compost treatments relative to the unburned control ($p < 0.03$). After four growing seasons, soil pH across depths was significantly higher in all burned treatment plots relative to the unburned control (Table S4, $p < 0.01$).


Year 1
Year 2
Year 3
Year 4
Peak Aboveground Biomass (g m^{-2})
0
200
400
600
CN
FW
FW2
GW
GW2
GW3
UCN


Total Biomass (g m^{-2})
0
500
1000
1500
2000
CN
FW
FW2
GW
GW2
GW3
UCN

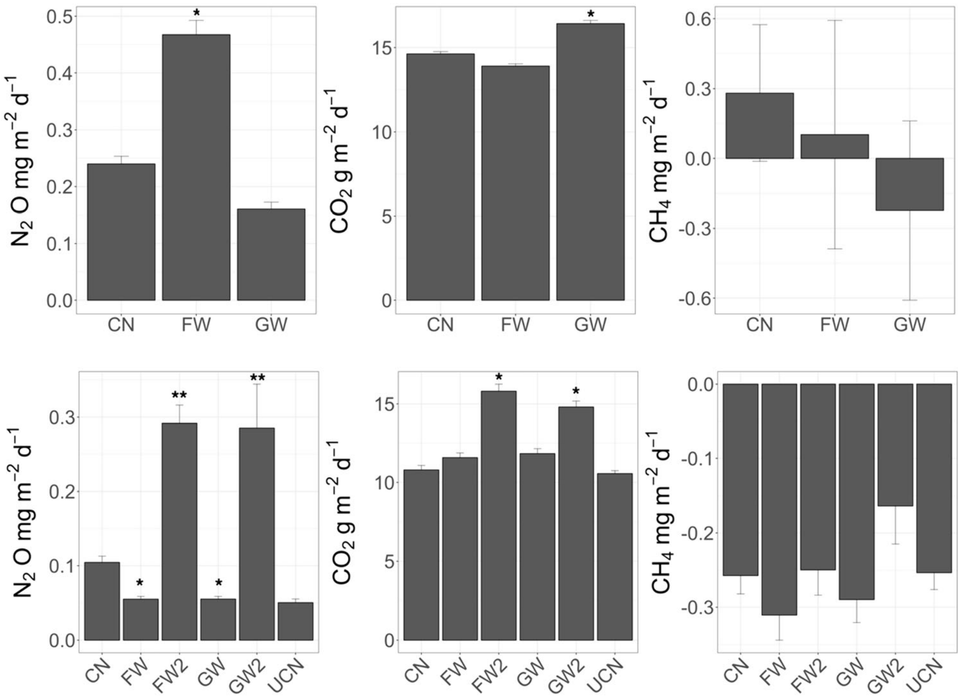


Figure 3. Annual mean nitrous oxide (N_2O mg m^{-2} d^{-1}), carbon dioxide (CO_2 g m^{-2} d^{-1}), and methane (CH_4 mg m^{-2} d^{-1}) fluxes in year one (top) and year two (bottom). *FW* food waste compost, *GW* green waste compost, *CN* untreated control, and *UCN* unburned control; *FW2 and GW2* two annual applications for green waste and food waste, respectively. Statistically differences ($p < 0.05$) from CN and UCN across individual depths and years were marked with * and marked with ** when statistically different than all other treatments ($p < 0.05$).

Discussion

Compost Amendment Type Effects on Soil Carbon and Aboveground Biomass

We found that the food waste compost (higher mineral N) resulted in greater soil C sequestration than the GW compost (lower mineral N) as hypothesized. After four growing seasons, the increase in soil C stocks was 9.5 ± 0.9 to 30.2 ± 0.7 Mg C ha^{-1} (0–40 cm) relative to the control and 19.5 ± 0.9 to 40.1 ± 0.7 Mg C ha^{-1} relative to the unburned control. Although studies varied in length, at 0–10 cm increases in soil C in both one-time compost treatments (6.3 ± 1.2 Mg C ha^{-1} for GW and 9.0 ± 2.0 Mg C ha^{-1} for FW) were similar to a short-term study that measured changes in soil C stocks across CA rangelands with compost amendments (Silver and others 2018) and lower than previous observations at a nearby study that added twice the amount of compost per application (13.8 to 17.7 Mg C ha^{-1}, Ryals and Silver 2013). Values for GW were also similar to a global meta-analysis of green waste composts added to rangelands (Kutos and others 2023). While few studies have explored the effects of composted food waste in rangelands, data from cropping systems showed soil C gain following application for multiple years (Reynolds and others 2015; Baiano and Morra 2017). Increases in soil C pools in all compost treatments in this study were always greater than the amount of C directly applied via compost amendments (5.5–5.9, 11.0–11.8, and 16.5 Mg C ha^{-1} for one-, two-, and three-time GW and FW application treatments, respectively). This increase is likely explained by increases in soil N and moisture availability that stimulate plant productivity and subsequent increases in ecosystem C inputs (Ryals and Silver 2013; Ryals and others 2014).

We also observed similar trends in the effects of compost type on aboveground biomass. Nitrogen availability is typically a limiting nutrient in plant productivity in California grasslands (Grogan and Chapin 2000), and thus increases in slow-release N from both compost additions likely helped drive the observed increases in aboveground biomass. All compost-amended treatments produced significantly more biomass overall than the controls, with both FW treatments producing significantly more biomass than the one- and two-time GW treatments. Compost amendments act as a slow-release fertilizer (Ryals and Silver 2013) which can increase plant production in nutrient-limited grasslands (Borer and others 2017). Increased soil organic matter content can also enhance soil water-holding capacity (Murphy 2015). Increased water-holding capacity (Ryals and Silver 2013; Flint and others 2018) and the availability of N and other nutrients in the FW compost and the GW3 compost application may have stimulated growth of belowground plant biomass (Ryals and Silver 2013; Cleland and others 2019). These increases in both the density and depth of root growth likely promote increases in soil C deeper in the soil profile and may also help explain the annual variability in aboveground biomass across compost treatments.

Aboveground biomass was lower in the last two years of the study relative to the first two years and decreases in N availability in subsequent years following compost addition may partially explain this pattern. However, this region of California experienced an extreme drought, with its three driest years on record occurring during this experiment (California Irrigation Management Information System, 2023). Additionally, declines in aboveground biomass occurred across all treatments, including the controls, suggesting this interannual variability in climate was the main driver of decreases in aboveground biomass observed.

Effects of Single Versus Repeated Amendments on Soil Carbon and Aboveground Biomass

We hypothesized that repeated annual applications of compost would results in greater soil and plant C stocks relative to a single application. A modeling study found that small multi-year amendments over 10 years had a similar effect on soil C sequestration as a single application when controlling for the total amount and quality of the compost (Ryals and others 2015). Here, we found that only the GW3 treatment increased cumulative soil C and N stocks after four years relative to a single compost application when accounting for the whole 40 cm profile. Both FW2 and GW2 compost treatments significantly increased soil C and N stocks in surface soils (0–10 cm) relative to the one-time treatments. This occurred in the growing season immediately following the second application and was maintained in the FW2 treatment after three growing seasons. However, by the fourth growing season GW, GW2, FW, and FW2 treatments did not differ significantly. Other studies have shown high interannual variability in soil C stocks following multiple years of compost amendment. For example, biennial compost amendments to cropland in Sweden led to increased soil C stocks in most, but not all, years of a 13-year field trial, and resulted in large overall increases in soil C stocks relative to controls (Kätterer and others 2014). Two compost applications over six years to irrigated pasture in semiarid rangelands increased soil C stocks in excess of the compost-C supplied by the end of the study, and also showed different levels of effects across years (McClelland and others 2022). Composted biosolid amendments at two time points over 11 years increased soil C stocks in a shortgrass steppe rangeland and showed both interannual variability in soil C stocks and differential effects of the amount of compost added (Ippolito and others 2010). These studies, like the current research, highlight the need for multi-year, long-term field measurements particularly in rangelands that are often characterized by high interannual variability in moisture.

These temporal dynamics in soil C stocks may be explained by the high interannual variability in rainfall and soil moisture in these drought-prone ecosystems that drives large year to year variation in gross primary productivity (Harpole and others 2007; Xia and others 2009). Additional C from repeated compost applications may only elevate soil C pools when there is enough rainfall to support a sufficiently large increase in plant growth relative to C losses via decomposition. It is also possible that after the first initial pulse of plant and soil C gain following compost amendments, subsequent responses were slower and more thus difficult to detect. It is notable that the amount of compost added in this study was small relative to the size of the soil organic C stock, even when considering multiple years of addition.

At the end of the fourth growing season, we observed that aboveground biomass in the FW and GW3 treatments were significantly greater than the unburned control. Composted organic matter additions increased soil C and N pools, likely contributing to increases in plant production through enhanced water-holding capacity and N availability (Ryals and Silver 2013). Our results highlight the potential for sustained increase in aboveground biomass following compost amendments and suggest that the observed effects on soil C and N stocks likely help to buffer drought impacts on aboveground productivity (Kowaljow and others 2010; Zhang and Xi 2021).

It is notable that the typical limitation to applying compost to rangelands is economic (Hall and others 2022), particularly in the absence of a robust carbon market. From a ranching or rangeland management perspective, the financial benefits of compost applications need to outweigh the costs. Here, we found that a single compost application can significantly increase plant growth as well as soil C stocks.

Compost Amendment Effects on Greenhouse Gas Emissions

We hypothesized that the amendments would increase N_2O emissions, particularly the higher-N FW compost, and that CO_2e emissions would significantly offset C gain through soil C sequestration. Compost additions increased both ecosystem respiration and N_2O emissions in the year of amendment application, likely due to higher mineral N availability, confirming the first part of our hypothesis. In previous studies, compost additions increased ecosystem respiration, with approximately 3.5% of total respiration C estimated to originate from compost C (Ryals and Silver 2013). Ecosystem respiration measurements represent both autotrophic and heterotrophic respiration, and increases in both plant biomass or respiration may contribute to these increases. Soil N_2O emissions during relatively warm spring wet-up events are likely hot moments that account for the majority of N_2O emissions (Anthony and Silver 2021). Amendment-stimulated increases in CO_2-equivalent (CO_2e) N_2O emissions were significantly smaller than observed increases in soil C pools across all compost treatments (0.21 mg N_2O m^{-2} d^{-1} = 0.76 kg CO_2e ha^{-1} y^{-1}, < 0.1% of smallest CO_2e soil C increase), and in the year following one-time amendments we did not detect significant differences in N_2O emissions between one-time treatments and controls. We observed no differences in CH_4 fluxes across treatments. Together these results suggest that increases in soil greenhouse gas emissions are low following compost amendments and are dwarfed by increases in soil C storage from organic matter amendments when accounting for the CO_2e balance in these grassland soils.

Effects of Fire on Soil Carbon and Aboveground Biomass

Fire is a natural part of much of the grassland biome, and climate change is increasing the frequency of fire events (Jones and others 2022). Fire generally increases nutrient availability (Stavi 2019), stimulating growth in subsequent growing seasons. Typically, grass fires range from less than 200 °C (Bailey and Anderson 1980) up to 300 °C (Clements 2010). Combustion temperatures directly determine what nutrients may be volatilized and removed from the system. For example, N begins volatilizing at approximately 200 °C, whereas other nutrients might be stable at temperatures well over 1000 °C (Neary and others 1999). Production of pyrogenic C from compost amendments may partially explain the increases in soil C observed, as pyrogenic C is thought to be a particularly stable form of soil C (Bird and others 2015). However, both the temperature and oxygen availability during combustion, the extent of combustion (Hedges and others 2000), and initial organic composition can dramatically affect the physicochemical properties of the remaining organic material, which is well-documented for quantifying differences across biochar feedstocks and methodologies (Sohi and others 2010; Tag and others 2016). Thus, differences across compost types, and the resulting impacts of combustion of applied compost, could theoretically affect subsequent biogeochemical cycling.

In this study, all burned treatments had higher soil C and N pools relative to the unburned controls. Compost amended sites had more soil C than unamended burned controls (CN) suggesting that fire did not negate the impacts of composted amendments for soil C sequestration. Furthermore, rates of C accumulation in soils were similar to previous research at the field site in the absence of fire (Ryals and others 2014). We did not see any discernable differential effects of fire between the two types of compost. While fire can lead to plant N losses via volatilization, it can also increase total C and mineral N concentrations (Augustine and others 2014; Limb and others 2016; Wang and others 2019). The fast-moving low-intensity fire in this study likely did not remove much of the compost and may have stimulated N cycling. Burned treatments had significantly higher soil pH values relative to the unburned plots following the fire event. Soil pH generally increases following fire (Memoli and others 2020; Chungu and others 2020), with pH shifts toward neutral facilitating increases in plant productivity. Higher NH_4^+ availability was also observed for two growing seasons after burning, suggesting the sustained increased in pH and organic matter availability may have favored conditions for N mineralization (Curtin and others 1998; Khalil and others 2005). Combined, compost amendments before or after low-intensity fire events may further increase soil fertility, helping mediate effects of decreased water availability under drought conditions and stimulate plant productivity following fire events.

Conclusion

We found increased soil C storage following compost amendments of different chemical quality in this annual grassland ecosystem. Compost derived from FW exhibited similar or greater increases in soil C storage relative to GW compost. Even with slight increases in soil N_2O fluxes, our results suggest that composting and diverting food waste from landfills can lead to greater C storage in Mediterranean grasslands while reducing emissions from the food system. A three-time GW compost application increased soil C storage relative to one- or two-time compost applications. This suggests less frequent compost applications may be sufficient to maintain C storage and productivity over sub-decadal timescales. Importantly, we also found that fire did not negatively affect, but instead increased soil C pools and aboveground biomass across compost-amended treatments, highlighting the likely resiliency of this C sequestration pathway to climate-induced changes to fire frequency.

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DATA AVAILABILITY

Impacts of compost amendment type and application frequency on a fire-impacted grassland ecosystem submitted to Dryad with DOI: https://doi.org/https://doi.org/10.5061/dryad.hhmgqnkr0.

Declarations

Conflict of interest The authors declare no conflict of interests.

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