

## **Compost Application Effects on Soil Greenhouse Gas Emissions Along the Climatic Gradient of California Grasslands**

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### **ABSTRACT**

To slow global climate change, it is becoming increasingly important to consider carbon (C) drawdown pathways to reduce carbon dioxide (CO<sub>2</sub>) concentrations in the atmosphere. Compost application to grasslands has been proposed as a CO<sub>2</sub> removal mechanism to mitigate climate change. However, compost application may cause exacerbated greenhouse gas (GHG) emissions and significantly offset CO<sub>2</sub> removal. To determine whether compost amendments could act as a net CO<sub>2</sub> source or sink, it is critical to understand patterns of GHG emissions with compost across a wide range of soil conditions. I used a laboratory incubation experiment with soils from five annual grasslands representing a range of soil conditions to determine the potential impacts of compost application on GHG emissions. I found that heterotrophic respiration was elevated under compost application at all sites, and was significantly greater in soils with higher initial C contents. Methane and nitrous oxide production were similar between amended and control soils, though my results showed that site-specific precipitation regimes might influence the production of these gases. My results suggest that, due to the priming effect of organic amendments, less C-rich sites should be a priority for compost addition. Models of long-term soil organic carbon (SOC) storage reinforce this; however, they also demonstrate that each of these sites will still sequester far more C than they will release over short- and long-time scales.

### **KEYWORDS**

microbial activity, carbon sequestration, aerobic respiration, carbon, greenhouse gas emissions

## INTRODUCTION

To mitigate the acceleration of global climate change, it is imperative that there are numerous, effective strategies for carbon (C) drawdown to reduce carbon dioxide (CO<sub>2</sub>) concentrations in the atmosphere. Soils can hold at least three times as much C as the atmosphere (White et al. 2000), making soil C sequestration the target of initiatives like 4 per 1000 (Paris Climate Change Conference 2015). Compost application to grasslands has been proposed as one such CO<sub>2</sub> removal mechanism to mitigate climate change. Although it is well-established that organic amendments bolster soil C and N stock (Ryals et al. 2014), there remains uncertainty regarding the mechanisms controlling variable GHG release and how to accurately quantify long-term SOC storage across sites (Stanton et al. 2018). Compost application to grasslands can provide an array of ecosystem services including higher quality forage and soil carbon sequestration without many of the negative consequences of other amendments such as manure or synthetic N fertilizer (Claassen et al. 2007).

One striking features of compost amendments is the apparent low GHG emissions relative to other organic amendments (DeLonge et al. 2013). This reduction is particularly significant with regard to mitigating GHGs with a greater warming potential, such as methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) (IPCC 2014). Compost application may potentially enable C storage over many decades with fewer associated GHG emissions (Silver et al. 2018). However, few intensive studies of GHG emissions have been conducted across a range of soil conditions following compost amendments. This is even true in California, where field trials have applied compost across a wide range of grasslands throughout the State. California's grasslands occur across gradients in mean annual rainfall and temperature, and are underlain by diverse soils and management—all of which has the potential to influence GHG emissions with compost application (Bond-Lamberty et al. 2010, Leitner et al. 2017). Differences in soil texture, bulk density, and soil organic matter affect water holding capacity, nutrient availability, and microbial activity, all of which could potentially impact responses to organic amendments (Jackson et al. 2017).

In this study, we explored what the influence of soil type was on GHG emissions from Mediterranean grassland soils following a compost application. Using a laboratory incubation experiment, we compared how the organic amendment differentially impacted soils from along the climatic gradient of California's annual grasslands. We then compared the results against the

abiotic components (texture, C and N content, climatic variables, etc.) to determine what site-specific controls were limiting or enhancing GHG emissions.

## METHODS

### Site description

We collected the soils from five different grassland sites in coastal and inland California and we completed all lab work at the University of California, Berkeley. Our study sites were Kaweah Oaks Preserve in Exeter, CA in Tulare County (36.33°N, 119.17°W), the Chamberlin Ranch in Los Olivos, CA in Santa Barbara County (34.71°N, 120.13°W), the Nicasio Native Grass Ranch in Nicasio, CA in Marin County (38.0834° N, 122.7633° W), the Sierra Foothills Research and Extension Center in Browns Valley, CA in Yuba County (39.34°N, 121.35°W), and the Rush Ranch in Covelo, CA in Mendocino County (39.84°N, 123.257°W). Each of these sites have distinctly different soil types reflecting the unique parent material and environmental conditions of the area, making them aptly suited choices for the purpose of this experiment. All of these sites are part of ongoing field experiments on the C sequestration potential of compost applications. Information about the soil types, soil properties, and environmental conditions had all been previously documented (Table 1) (Silver et. al, 2018).

**Table 1. Soil Type Summary.**

<b>Site Name</b>	<b>Soil Type</b>	<b>% Sand</b>	<b>% Silt</b>	<b>% Clay</b>	<b>MAP (cm/yr)</b>	<b>MAT (°C)</b>	<b>% C</b>	<b>% N</b>	<b>pH</b>
Chamberlin Ranch	Gravelly fine sandy loam [Typic Argixeroll (Mollisol)]	63%	30%	7%	38	8.0 - 25.1	1.67 ± 0.41	0.15 ± 0.04	6.31 ± 0.19
Kaweah Oaks Preserve	Calcic Haploxerept (Inceptisol)	43%	47%	10%	28	10.8-24.1	1.72 ± 0.48	0.15 ± 0.05	8.33 ± 0.13
Nicasio Native Grass Ranch	Tocaloma-Saurin-Bonnydoon soil series [Typic Haploxeroll (Mollisol)]	45%	30%	25%	97	8.3- 20.0	2.97 ± 0.38	0.26 ± 0.03	6.77 ± 0.02
Sierra Foothills Research & Extension Center	Auburn-Sobrante complex [Mollic Haploxeralfs (Alfisol and Inceptisol)]	40%	40%	20%	73	10.3 - 24.4	2.85 ± 0.21	0.27 ± 0.02	6.79 ± 0.07
Rush Ranch	Coal Loam Argixeroll (Mollisol)	50%	35%	15%	108	4.6 - 22.3	2.66 ± 0.54	0.25 ± 0.05	6.16 ± 0.08

## **Soil & compost sourcing**

We used a controlled laboratory incubation experiment with soils from each of the five sites and one standardized fully-finished compost. Following the start of the wet season in late November 2019, we collected 2 gallon-size Ziploc bags of topsoil (0-10cm depth) from each site and transferred it back to the lab in coolers. To collect the samples, we removed any remaining plant matter from the surface, placed a ruler 10 cm deep into the soil and only dug to the base of the ruler. We selected sample locations within fields at random. We sourced the compost from the West Marin Composting Facility (38.075°N, 122.707°W). We used the Nicasio blend which was a mixture of green waste and chicken pellet manure. This was the type of compost used on all five sites as a part of a larger C sequestration field study conducted over the past three years.

## **Soil moisture, nutrient content, and pH**

Within a day of collecting the soils, we measured the pH, mineral N concentrations, and soil moisture of these samples. We conducted pH tests for the soils by adding 5mL of deionized water to 5g of fresh soil, vortexing the solution for a minute, and then measuring using a pH probe. We measured inorganic N content using a KCl extraction (Sparks et al. 1996). We added 2M KCl to 15g of fresh soil, left the samples to shake for an hour, filtered, and froze them to be analyzed on a Lachat Quick Chem Flow Injection Analyzer (Milwaukee, WI). We measured soil moisture by weighing 10g of field-fresh soils before and after oven-drying at 105 °C for at least 24 hours.

## **Incubation & GHG measurements**

We measured CO<sub>2</sub>, N<sub>2</sub>O, and CH<sub>4</sub> from incubated soils daily for a month (followed by weekly measurements for another 3 weeks). To set up the incubation, we sieved the soil and removed large stones or plant matter. We added approximately 300g of soil from each site to a 1-quart jar, with 6 replicates for each site. For 3 of the jars from each site, we added 41.8 g of compost and lightly mixed it with the soil. We covered the jars with a small piece of foil with a few aeration holes and kept at room temperature in the dark. We sealed the jars with lids with septa for sampling. During each sampling period, we collected 5mL immediately after sealing the jar and again at 3

hours to determine a flux rate. We did this daily for one month and continued to take samples once a week for another 3 weeks. We analyzed gas samples on a Shimadzu GC-14A gas chromatograph (Pleasanton, CA), equipped with a thermal conductivity detector (CO<sub>2</sub>), a flame ionization detector (CH<sub>4</sub>), and electron capture detector (N<sub>2</sub>O). We ran blanks and standard gases for quality control. After daily sampling, we added DI water to each jar to maintain field moisture content (weight basis).

## Data analysis

We converted gas concentrations (ppm) to flux rates using the equations:

$$\text{CH}_4 \text{ Flux} = [\text{CH}_4 (T_3) - \text{CH}_4 (T_0)] / [T_{\text{elapsed}} * dw] * (12 * 10^3)$$

$$\text{N}_2\text{O Flux} = [\text{N}_2\text{O} (T_3) - \text{N}_2\text{O}(T_0)] / [T_{\text{elapsed}} * dw] * (28 * 10^3)$$

$$\text{CO}_2 \text{ Flux} = [\text{CO}_2 (T_3) - \text{CO}_2 (T_0)] / [T_{\text{elapsed}} * dw] * (12)$$

Where the flux of each gas is equal to the concentration (ppm) initially subtracted from the concentration after 3 hours,  $T_{\text{elapsed}}$  is the time between  $T_0$  and  $T_3$ ,  $dw$  is the dry weight of the soil. We multiplied these values for each gas ( $\mu\text{mol g}^{-1} \text{hr}^{-1}$ ) by the respective conversion factors into concentrations of  $\mu\text{g C g}^{-1} \text{CO}_2 \text{ hr}^{-1}$  for CO<sub>2</sub>,  $\text{ng C g}^{-1} \text{CH}_4 \text{ hr}^{-1}$  for CH<sub>4</sub>, and  $\text{ng N g}^{-1} \text{N}_2\text{O hr}^{-1}$  for N<sub>2</sub>O.

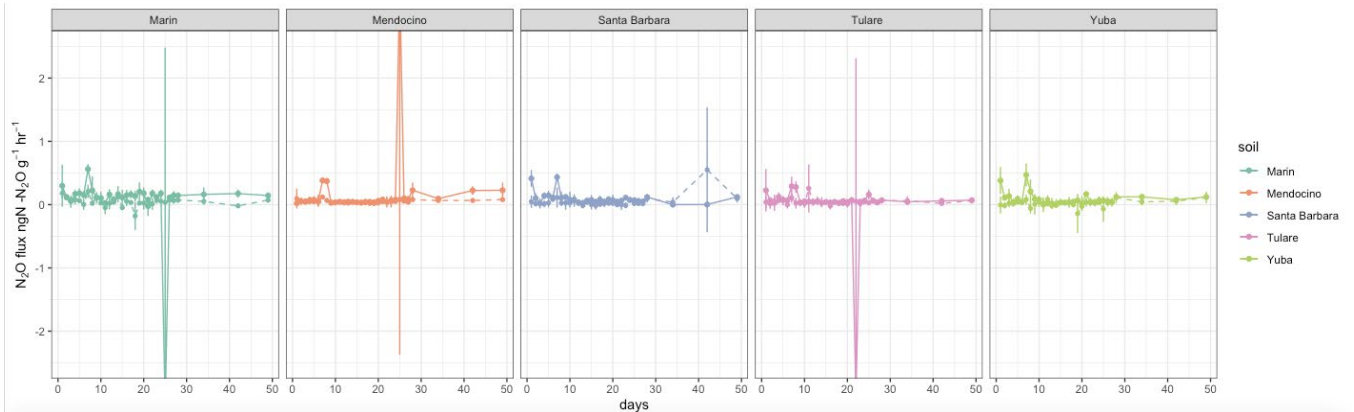
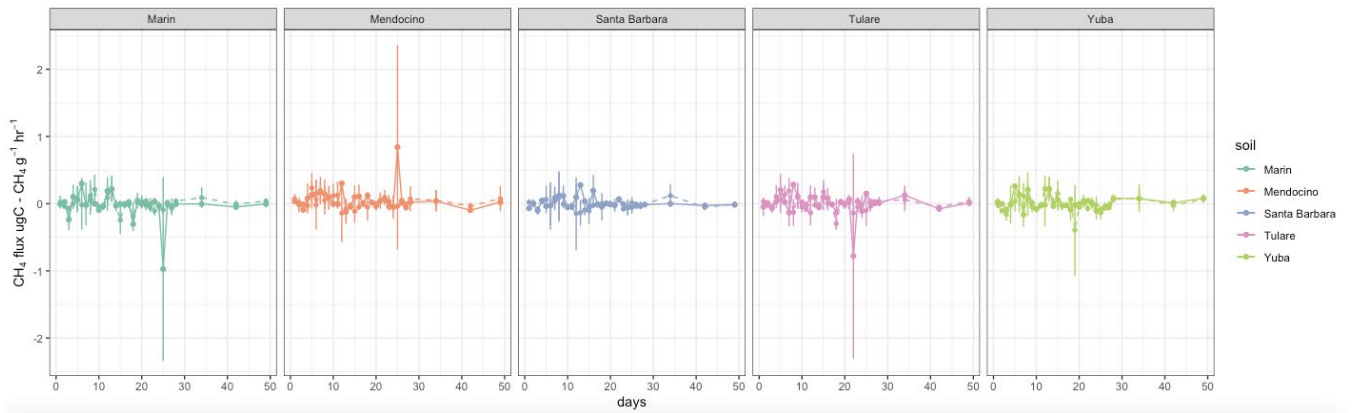
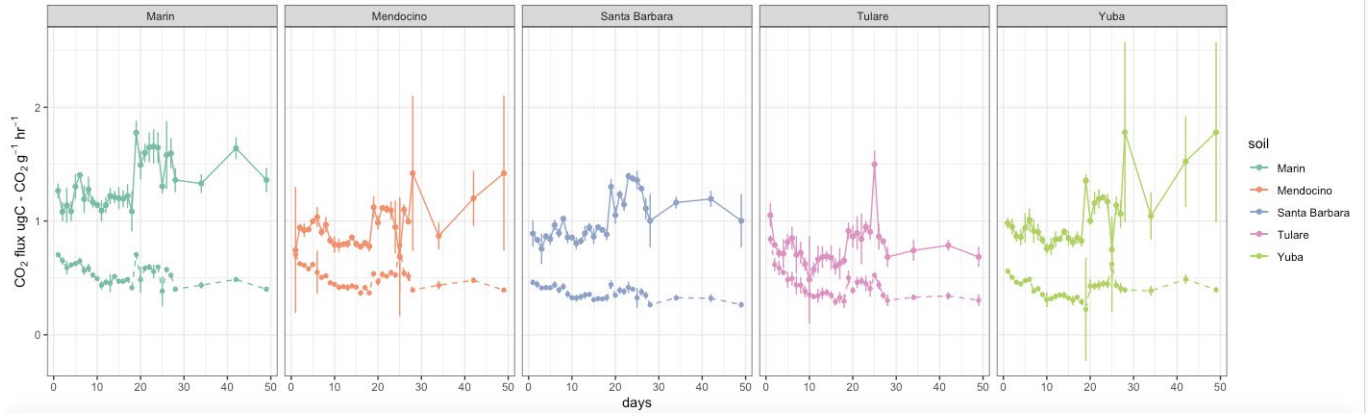
We analyzed fluxes by site and treatment. We compiled daily averages over the incubation period to explore patterns over time. We integrated the curves for total CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O for each site under both compost and control conditions to calculate the total amount of each gas produced over the 7-week period. We used an analysis of variance (ANOVA) and linear regressions to determine if there were a significant treatment and/or site effect for each gas. We then used an ANOVA to study the relationship between gas fluxes and the % clay, the initial C and N concentrations, pH, mean annual temperature (MAT), mean annual precipitation (MAP), and latitude at each site. After determining which variables showed significant relationships, we ran linear regressions on the specific variables to measure which factors were the strongest controls on GHG production. All data analysis was done in R (version 3.6.1).

## RESULTS

### Incubation experiment & GHG emissions

Compost amended soils showed significantly higher respiration rates than control soils across sites (Figure 1). The Marin soils (Typic Haploxeroll, Mollisol) showed the highest daily flux rates of CO<sub>2</sub> under compost addition. The Mendocino (Coal loam Argixeroll, Mollisol), Santa Barbara (Typic Argixerolls, Mollisol), and Yuba (Mollic Haploxeralfs, Alfisols & Inceptisols) soils had approximately similar CO<sub>2</sub> flux rates under the treatment. Tulare (Calcic Haploxerept, Inceptisol), almost always exhibited the lowest emissions of CO<sub>2</sub> over the span of the incubation. There were spikes in CO<sub>2</sub> production on days 20 and 40. The Marin, Yuba, and Mendocino soils fluctuated towards the end of the 7-week period while Santa Barbara and Tulare started to plateau in the compost treated soils towards the end. The control plots showed almost identical daily flux rates in all 5 sites.

Despite the increase in aerobic respiration, compost-treated soils did not alter emissions of CH<sub>4</sub> and N<sub>2</sub>O in any soil type (Figure 2, Figure 3). CH<sub>4</sub> production remained approximately 0 for almost all soil types (Figure 2). There was no clear trend of CH<sub>4</sub> emissions in any of the soils as a result of the compost addition. N<sub>2</sub>O fluxes were also generally low (Figure 3). However, there was a lot of daily variation in both CH<sub>4</sub> and N<sub>2</sub>O flux rates in both the amended and control soils, though there was no clear trend of production of either gas in any of the soils as a result of compost addition.

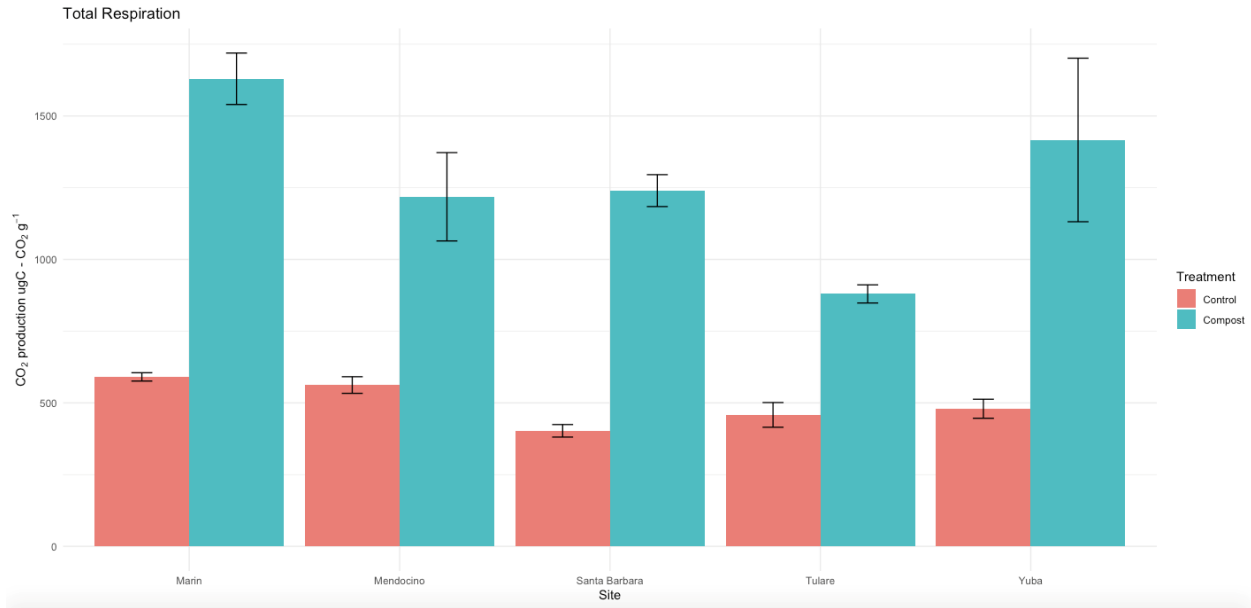


**Fluxes for each site over a 7-week period:** (Figure 1) CO<sub>2</sub>, (Figure 2) CH<sub>4</sub>, (Figure 3) N<sub>2</sub>O. Solid line represents jars with the compost treatment and the dashed line represents jars with the control treatment. Each is presented with a standard error bar and each point represents an average hourly flux for each day.

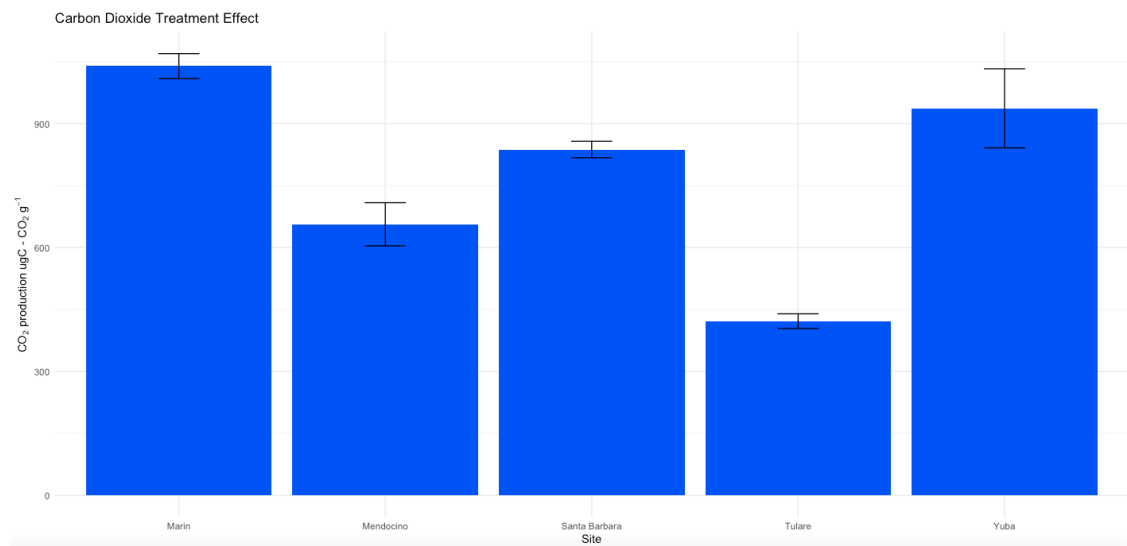


**Total CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O**

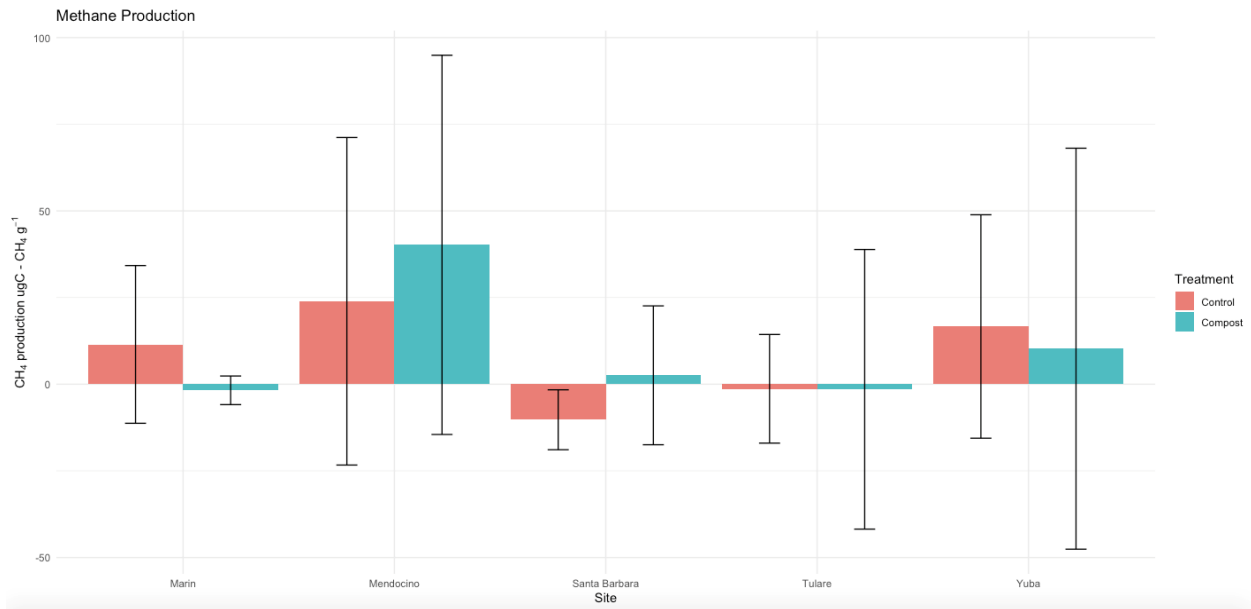
Across all sites, heterotrophic respiration in the amended soils was significantly higher than respiration in the controls ( $p < 0.0001$ ) (Figure 4). Differences among sites were also statistically significantly different ( $p < 0.0001$ ) and there was a significant effect demonstrating interaction between compost addition and site ( $p < 0.001$ ). Under compost application, Santa Barbara and Yuba were significantly higher than Tulare ( $p < 0.05$ ). Marin was significantly higher than Mendocino ( $p < 0.10$ ) and Tulare ( $p < 0.05$ ). CH<sub>4</sub> production was highly variables across all sites during the 7 week period (Figure 6). There was no significant difference between treatment and control fluxes across sites ( $p = 0.8$ ). Differences in the CH<sub>4</sub> flux between sites was also negligible ( $p = 0.4$ ). N<sub>2</sub>O production was highly variable in both treatments (Figure 8). There were no significant differences between the treatment and control jars across sites ( $p = 0.3$ ). There were also no significant difference among sites ( $p = 0.12$ ).



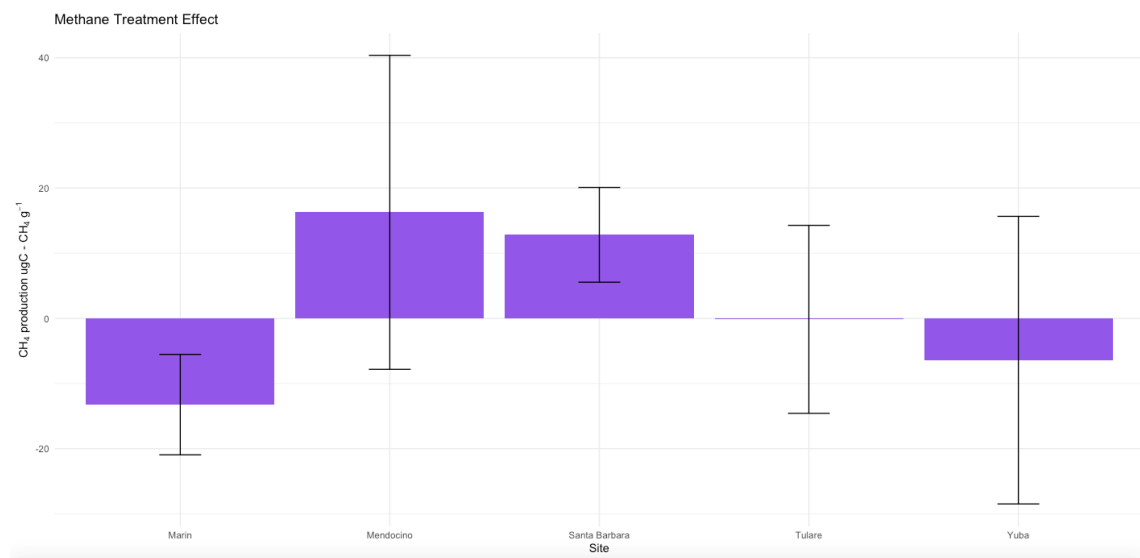
**Figure 4.** Total amount of aerobic respiration over the 7 week period. Bars indicate total CO<sub>2</sub> under treatment (blue) and control (pink) across sites for the 7-week period.



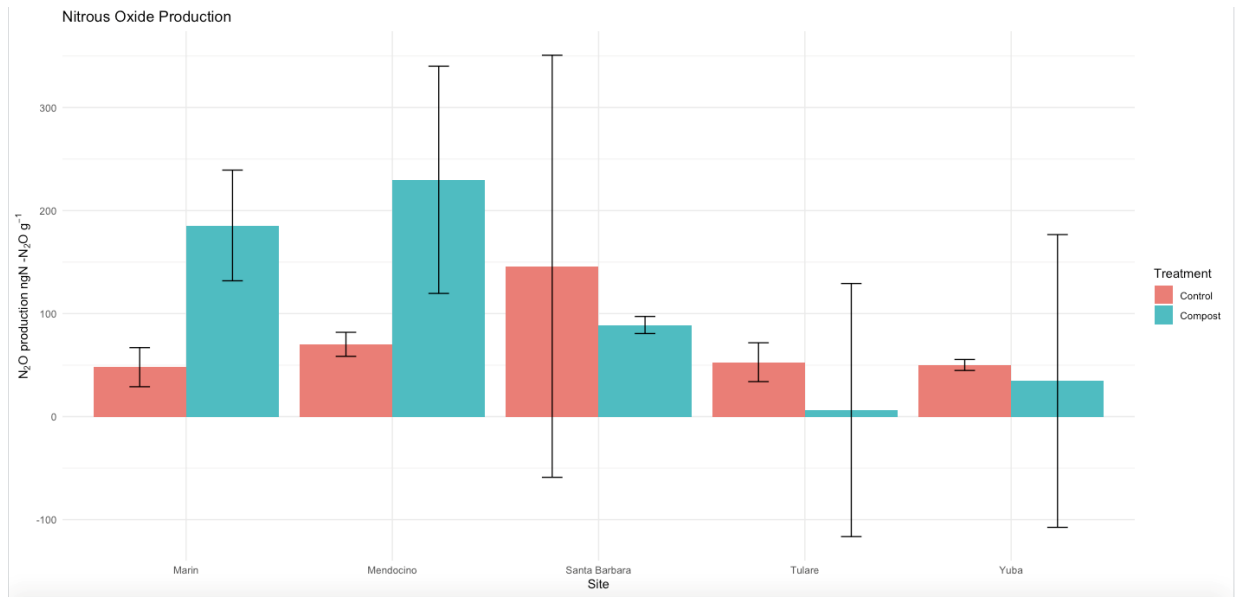
**Figure 5.** Bars indicate the treatment effect (compost-control) on CO<sub>2</sub> produced across all sites.



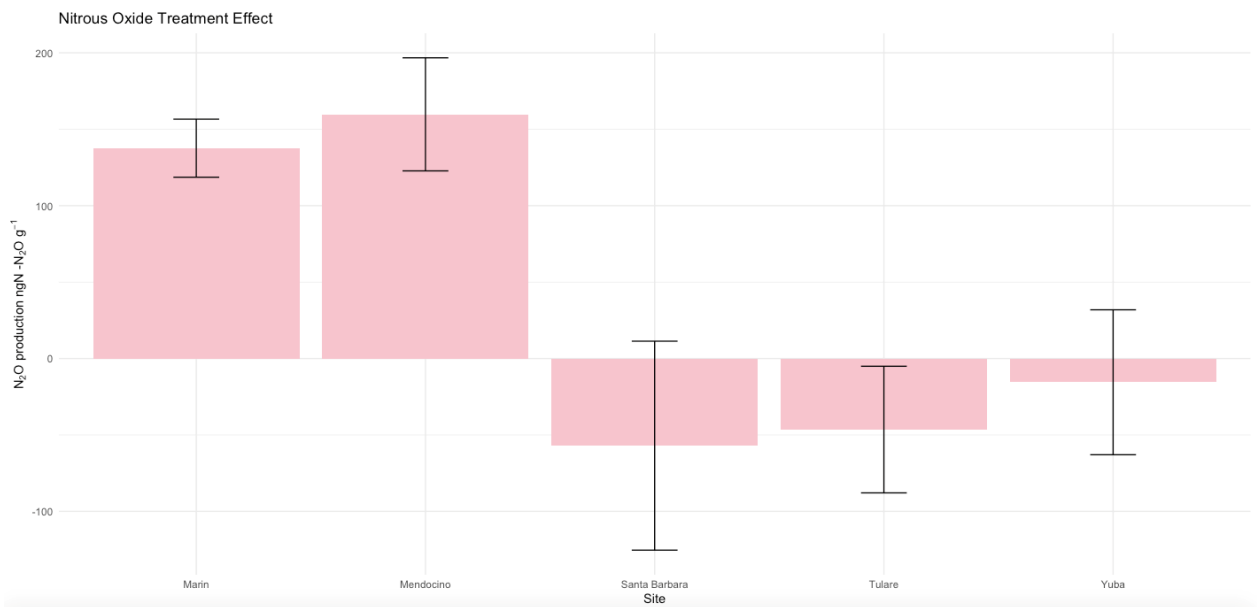
**Figure 6.** Total amount of CH<sub>4</sub> production over the 7 week period. Bars indicate total CH<sub>4</sub> under treatment (blue) and control (pink) across sites for the 7-week period.



**Figure 7.** Bars indicate the treatment effect (compost-control) on CH<sub>4</sub> produced across all sites.



**Figure 8.** Total amount of N<sub>2</sub>O production over the 7 week period. Bars indicate total N<sub>2</sub>O under treatment (blue) and control (pink) across sites for the 7-week period.



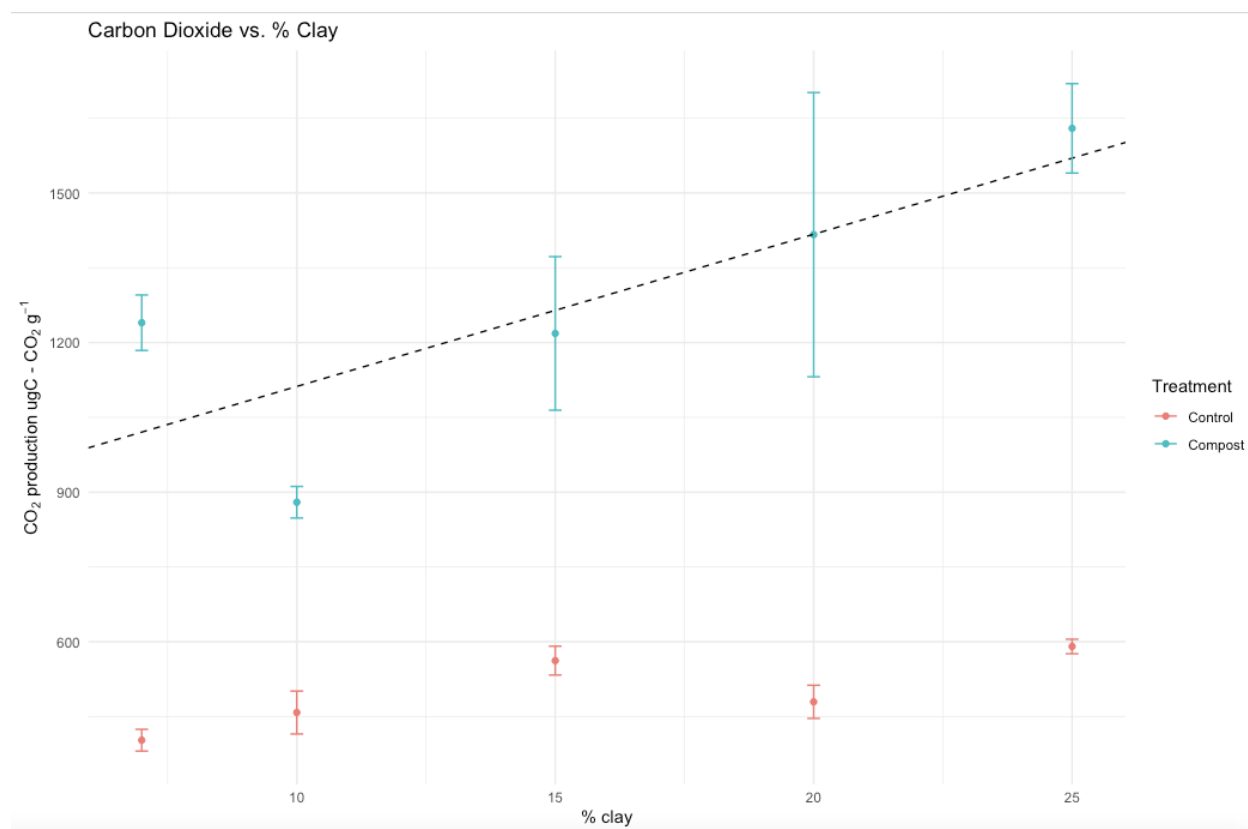
**Figure 9.** Bars indicate the treatment effect (compost-control) on N<sub>2</sub>O produced across all sites.

## ANOVA & linear regressions

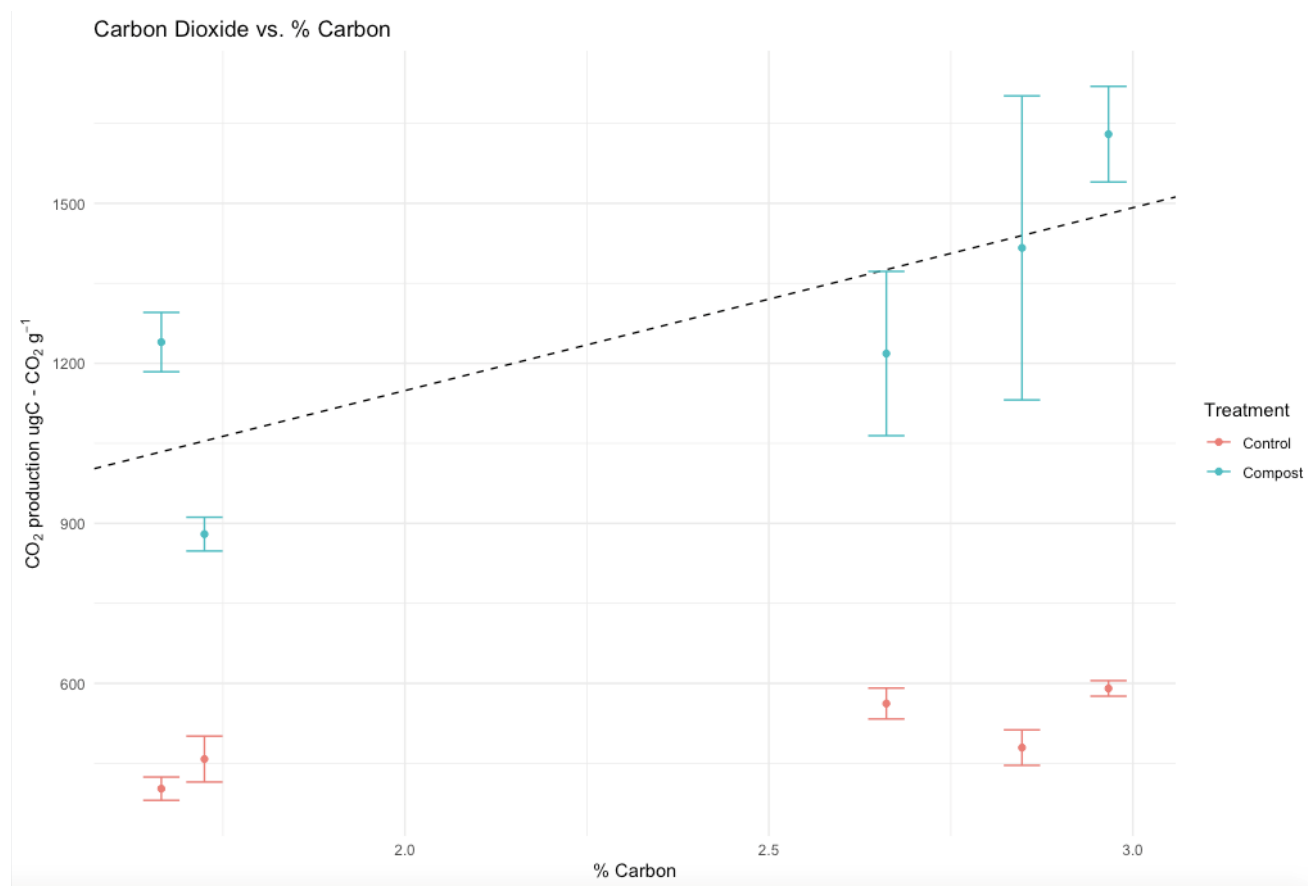
In the linear regressions of each of the gases against different abiotic factors, there was no significant effect of any of the variables on CH<sub>4</sub>.

## Carbon dioxide

The strongest driving factor for CO<sub>2</sub> was the % clay (figure 10) and background C concentration (figure 11) in each soil originally ( $p < 0.01$ ). There was a significant effect on respiration from the interaction between C and clay contents ( $p < 0.01$ ). Other variables which demonstrated a level of significance (pH, max MAT) had correlation coefficients of less than 0.30 and were not considered strong controls on GHG emissions.



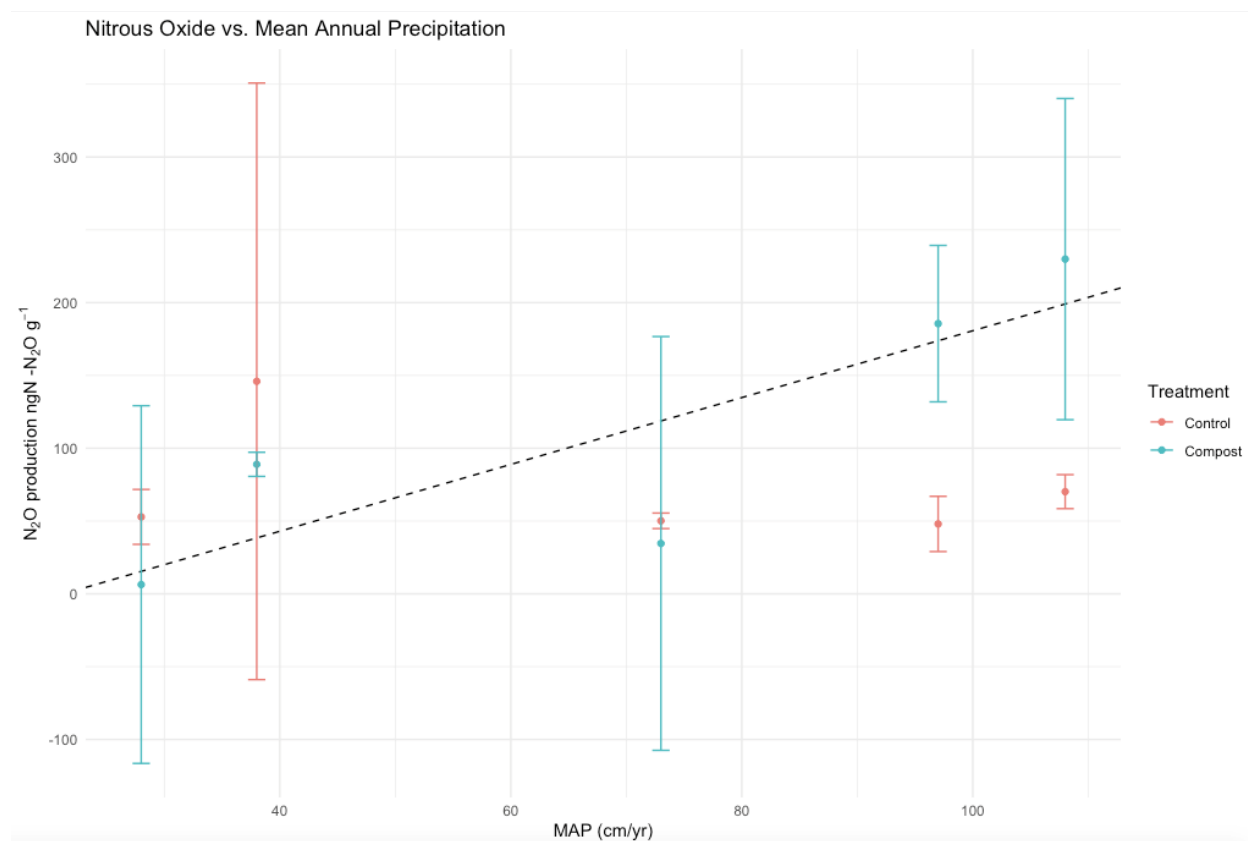
**Figure 10. Mean total CO<sub>2</sub> versus the % clay in each of the sites.** The blue points are the compost treatment whereas the pink points represent the control jars.  $R^2 = 0.4794$ .



**Figure 11. Mean total CO<sub>2</sub> versus the % carbon in each of the sites.** The blue points are the compost treatment whereas the pink points represent the control jars.  $R^2 = 0.4441$ .

### Nitrous oxide

Although there were no significant differences between site or treatment for N<sub>2</sub>O production, there was an apparent precipitation threshold in which production increased under compost addition. There was a significant effect of MAP (figure 12) on N<sub>2</sub>O production ( $p < 0.05$ ) under the compost application whereas there was no significant effect of MAP for the control jars ( $p = 0.53$ ).



**Figure 12: Mean total N<sub>2</sub>O versus MAP in each of the sites.** The blue points are the compost treatment whereas the pink points represent the control jars.  $R^2 = 0.322$ .

## DISCUSSION

Our results largely indicate that, without seasonal variability in play, the initial C content of a site is the strongest control on microbial aerobic respiration. We found that both CH<sub>4</sub> and N<sub>2</sub>O production are both largely unaffected by compost application across sites, though different climatic variables might influence this result in the field. However, which components of soil type are causing increased microbial activity following an organic matter treatment has not been explored with great detail. This is in part because potential driving variables often confound one another making it difficult to separate specific effects in field experiments at individual sites (Oertel et al. 2016), so it can be difficult to isolate drivers of GHG production. In the research described here, climate and plant dynamics have been removed, decreasing the number of potential

confounding factors. While this can also decrease the relevance of results to actual field conditions, it allows us to explore a suite of mechanistic controls on GHG dynamics in more detail.

### **Carbon dioxide fluxes**

Respiration peaked around the middle of the incubation period before starting to plateau at most of the sites. This peak could be due to an increase in microbial biomass (Perucci et al. 1990) and consequently, microbial activity. The observed CO<sub>2</sub> fluxes under the organic matter addition did not follow patterns demonstrated in other experiments (Ros et al. 2003, Perucci et al. 1992). In previous experiments, the highest microbial activity under a compost addition occurred immediately after the application and then plateaued (Ros et al. 2003). The short duration of our experiment may explain why we did not see similar patterns to other longer-term incubations.

The clay content and initial C concentrations influenced microbial respiration more than any other abiotic factor. However, clay and C concentrations were auto-correlated ( $p < 0.0001$ ,  $R^2 = 0.86$ ) and thus, C concentration is likely the driving factor behind the level of activity we observe in different soils. Although clay-rich, finer soils are associated with more stable forms of C, they also often have a larger labile pool of C than coarser soils (Kaye et al. 2002). With an organic matter addition, there can be a priming effect on the microbial communities bolstering C mineralization in the short term following an application (Luo et al. 2011). This may help to explain why soils with a higher initial C concentration were more susceptible to greater rates of mineralization following a compost application than those with less bioavailable C already in the soil. Even relatively stable forms of carbon in clay-rich soils have been demonstrated to be at risk of destabilization following an increase in substrate availability (Hamer et al. 2004). Organic rich soils are generally more nutrient-rich as well (Kaiser and Kalbitz 2012). With microbial communities well-adapted to a Mediterranean climate, respiratory activity was highest in more fertile soils (Zhao et al., 2010).

### **Nitrous oxide & methane fluxes**

Much of the doubt surrounding the effectiveness of organic amendments as a carbon drawdown mechanism has been rooted in the idea that compost will increase CH<sub>4</sub> and N<sub>2</sub>O

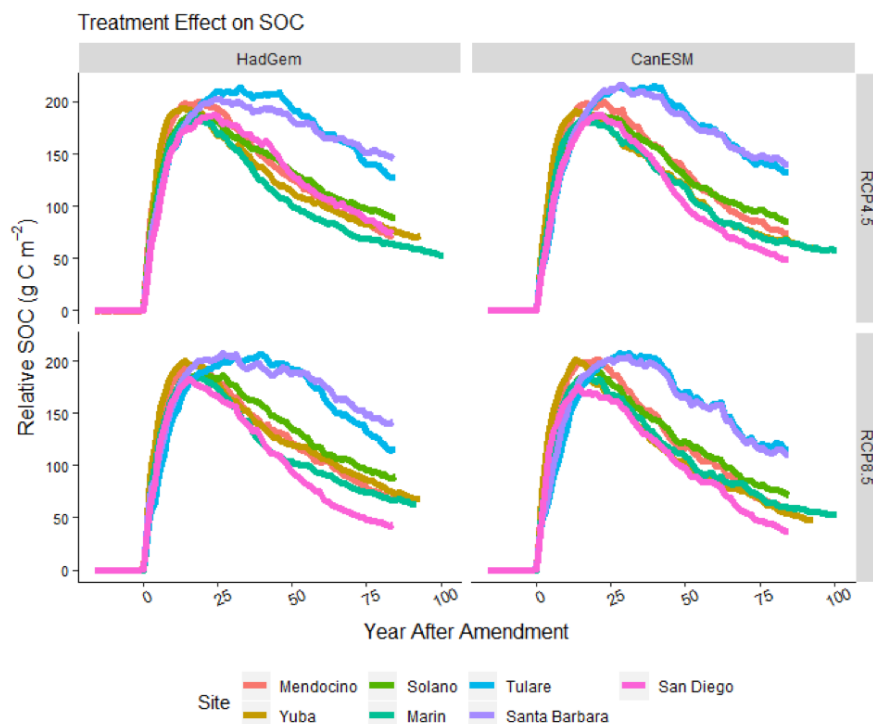


emissions from soils (Powlson et al. 2011). We found that CH<sub>4</sub> emissions were nearly negligible at every site. Others have also found no significant CH<sub>4</sub> emissions under compost addition (Favoino and Hogg 2008). N<sub>2</sub>O production was also not statistically different from the controls in each site. This result can be at least partially attributed to the slow nutrient release of compost, effectively reducing N<sub>2</sub>O production and preventing rapid N mineralization (Shoji et al. 2001).

Nitrous oxide production can often be produced in pulses which are thought to be driven by seasonal precipitation patterns (D'Odorico et al. 2003). Preliminary modeling had found that the net sequestration effect provided by compost application might be offset due to resulting annual N<sub>2</sub>O release in wet environments (Venterea et al. 2012). On grasslands, significant N<sub>2</sub>O releases can also be driven by compost being applied at the start of the growing season when the site is at its wettest (Dobbie and Smith 2003). However, there is some evidence that this fear of extreme seasonal N<sub>2</sub>O pulses under compost application might not amount to much of a cumulative effect on an annual time scale (Suddick and Six 2013). In contrast, compost application offers a significant reduction in N<sub>2</sub>O emissions coupled with high net primary productivity (NPP) compared to many of our current agricultural practices (Alluvione et al. 2010).

### **Implications for carbon sequestration**

Understanding patterns of GHG emissions can be incredibly valuable in trying to quantify or model the C sequestration potential across different geographic locations. Using a combination of earth system and biogeochemical models, Mayer and Silver (in prep.) predicted long term C storage possibilities across the different sites used in this experiment:



**Figure 13.** SOC storage following a single compost amendment over 100 years across California grasslands.

Tulare, the site with the lowest respiration observed in our study, demonstrates the largest C storage potential (Figure 13). The model projection also found Marin to have a slightly reduced storage capacity compared to the other sites, consistent with the results from this study which found the largest CO<sub>2</sub> production in Marin soil. It is notable that in the model projections, each site still shows a significant capacity to be a long-term C sink. Previous measurements have demonstrated that all sites have shown a net increase in SOC content both in the short and long term after a single compost application (Silver et al. 2018).

## Limitations

Incubation experiments inherently miss the effects of seasonality on microbial communities. Data from field experiments across these sites will be incredibly valuable in understanding how daily and seasonal environmental variability interact with soil variables to control microbial activity. Although sites with lower carbon stocks indicate greater SOC storage, this might not translate to an equivalent increase in NPP across sites. Measuring if there are any

differential effects of compost on NPP across sites will be critical for understanding net ecosystem carbon storage potential.

### **Future Directions**

To further understand the effect of soil type on C sequestration on these amended sites, the next step will be to measure and calculate net C and N stored in the compost-treated soils relative to the untreated ones. This can give us a more definitive picture on whether the factors controlling microbial activity are the same as the ones controlling C sequestration. Coupling this with continued measurements of C stocks and GHG release at these sites can help elucidate which abiotic controls are responsible for the patterns of storage observed.

### **Broader Implications**

These results are critical in illustrating the importance of a mechanistic understanding of microbial response to compost application. This understanding is valuable for making projections and scaling to a broader range of climatic zones. This study reinforces projections from biogeochemical models which have simulated minimal N<sub>2</sub>O and CH<sub>4</sub> emissions following a compost application (Mayer et al. 2018). Especially when using models as a source to understand the effects of an amendment, having a complete picture of what factors are controlling mineralization can be valuable in more accurately quantifying the influences of organic matter additions across the world (Wieder et al. 2018). Knowing this information can also guide policy in determining where to prioritize mitigation efforts based on predicted long term C storage potential.

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## REFERENCES

- Alluvione, F., C. Bertora, L. Zavattaro, and C. Grignani. 2010. Nitrous Oxide and Carbon Dioxide Emissions Following Green Manure and Compost Fertilization in Corn. *Soil Science Society of America Journal* 74:384–395.
- Bond-Lamberty, B., and A. Thomson. 2010. Temperature-associated increases in the global soil respiration record. *Nature* 464:579–582.
- Carey, J., and Claassen, V. P. 2007. Comparison of Slow-Release Nitrogen Yield from Organic Soil Amendments and Chemical Fertilizers and Implications for Regeneration of Disturbed Sites. *Land Degradation & Development*:119–132.
- DeLonge, M. S., R. Ryals, and W. L. Silver. 2013. A Lifecycle Model to Evaluate Carbon Sequestration Potential and Greenhouse Gas Dynamics of Managed Grasslands. *Ecosystems* 16:962–979.
- Dobbie, K. E., and K. A. Smith. 2003. Nitrous oxide emission factors for agricultural soils in Great Britain: The impact of soil water-filled pore space and other controlling variables. *Global Change Biology* 9:204–218.
- D’odorico, P., A. Porporato, F. Laio, and I. Rodriguez-Iturbe. 2003. Hydrologic Controls on Soil Carbon and nitrogen cycles. *Advances in Water Resources* 26:45–58.
- Favoino, E., and D. Hogg. 2008. The potential role of compost in reducing greenhouse gases. *Waste Management and Research* 26:61–69
- French ministry for agriculture & food. 2015. Join the 4 per 1000 initiative: Soils for food security and climate change. Paris Climate Change Conference.
- Hamer, U., B. Marschner, S. Brodowski, and W. Amelung. 2004. Interactive priming of black carbon and glucose mineralisation. *Organic Geochemistry* 35:823–830.
- IPCC, 2014: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

- Jackson, R. B., K. Lajtha, S. E. Crow, G. Hugelius, M. G. Kramer, and G. Piñeiro. 2017. The Ecology of Soil Carbon: Pools, Vulnerabilities, and Biotic and Abiotic Controls. *Annual Review of Ecology, Evolution, and Systematics* 48:419–445.
- Kaiser, K., and K. Kalbitz. 2012. Cycling downwards - dissolved organic matter in soils. *Soil Biology and Biochemistry* 52:29–32.
- Kaye, J., J. Barrett, and I. Burke. 2002. Stable Nitrogen of Grassland Soils Carbon and Carbon Variable Pools in Texture Content. *Ecosystems* 5:461–471.
- Leitner, S., P. M. Homyak, J. C. Blankinship, J. Eberwein, G. D. Jenerette, S. Zechmeister-Boltenstern, and J. P. Schimel. 2017. Linking NO and N<sub>2</sub>O emission pulses with the mobilization of mineral and organic N upon rewetting dry soils. *Soil Biology and Biochemistry* 115:461–466.
- Luo, Y., M. Durenkamp, M. De Nobili, Q. Lin, and P. C. Brookes. 2011. Short term soil priming effects and the mineralisation of biochar following its incorporation to soils of different pH. *Soil Biology and Biochemistry* 43:2304–2314.
- Mayer, A., Z. Hausfather, A. D. Jones, and W. L. Silver. 2018. The potential of agricultural land management to contribute to lower global surface temperatures. *Science Advances* 4:1–9.
- Oertel, C., J. Matschullat, K. Zurba, F. Zimmermann, and S. Erasmi. 2016. Greenhouse gas emissions from soils—A review. *Chemie der Erde - Geochemistry* 76:327–352.
- Perucci, P. 1992. Enzyme activity and microbial biomass in a field soil amended with municipal refuse. *Biology and Fertility of Soils* 14:54–60.
- Perucci, P., C. Agraria, U. Perugia, and B. X. X. Giugno. 1990. Biology and Fertility of Soils Effect of the addition of municipal solid-waste compost on microbial biomass and enzyme activities in soil 221–226.
- Powlson, D. S., A. P. Whitmore, and K. W. T. Goulding. 2011. Soil carbon sequestration to mitigate climate change: A critical re-examination to identify the true and the false. *European Journal of Soil Science* 62:42–55.
- Ros, M., M. T. Hernandez, and C. García. 2003. Soil microbial activity after restoration of a semiarid soil by organic amendments. *Soil Biology and Biochemistry* 35:463–469.
- Ryals, R., M. Kaiser, M. S. Torn, A. A. Berhe, and W. L. Silver. 2014. Impacts of organic matter amendments on carbon and nitrogen dynamics in grassland soils. *Soil Biology and Biochemistry* 68:52–61.
- Shoji, S., J. Delgado, A. Mosier, and Y. Miura. 2001. Use of controlled release fertilizers and nitrification inhibitors to increase nitrogen use efficiency and to conserve air and water quality. *Communications in Soil Science and Plant Analysis* 32:1051–1070.

- Silver, W. L., S. E. Vergara, and A. Mayer. (n.d.). 2018. Carbon Sequestration and Greenhouse Gas Mitigation Potential of Composting and Soil Amendments on California's Grasslands.
- Sparks, D.L., 1996. *Methods of Soil Analysis Part 3: Chemical Methods*. Soil Science Society of America and American Society of Agronomy, Madison, WI.
- Stanton, C. Y., K. J. Mach, P. A. Turner, S. J. Lalonde, D. L. Sanchez, and C. B. Field. 2018. Managing cropland and rangeland for climate mitigation: an expert elicitation on soil carbon in California. *Climatic Change* 147:633–646.
- Suddick, E. C., and J. Six. 2013. An estimation of annual nitrous oxide emissions and soil quality following the amendment of high temperature walnut shell biochar and compost to a small-scale vegetable crop rotation. *Science of the Total Environment* 465:298–307.
- Venterea, R. T., A. D. Halvorson, N. Kitchen, M. A. Liebig, M. A. Cavigelli, S. J. Del Grosso, P. P. Motavalli, K. A. Nelson, K. A. Spokas, B. P. Singh, C. E. Stewart, A. Ranaivoson, J. Strock, and H. Collins. 2012. Challenges and opportunities for mitigating nitrous oxide emissions from fertilized cropping systems. *Frontiers in Ecology and the Environment* 10:562–570.
- White, R., S. Murray, and M. Rohweder. 2000. *Grassland ecosystems*. World Resources Institute, Washington, D.C., USA.
- Wieder, W. R., M. D. Hartman, B. N. Sulman, Y. P. Wang, C. D. Koven, and G. B. Bonan. 2018. Carbon cycle confidence and uncertainty: Exploring variation among soil biogeochemical models. *Global Change Biology* 24:1563–1579.
- Zhao, B., J. Chen, J. Zhang, and S. Qin. 2010. Soil microbial biomass and activity response to repeated drying-rewetting cycles along a soil fertility gradient modified by long-term fertilization management practices. *Geoderma* 160:218–224.