CARBON SEQUESTRATION AND GREENHOUSE GAS MITIGATION POTENTIAL OF COMPOSTING AND SOIL AMENDMENTS ON CALIFORNIA'S RANGELANDS

A Report for:

California's Fourth Climate Change Assessment

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Edmund G. Brown, Jr., *Governor*

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PREFACE

California's Climate Change Assessments provide a scientific foundation for understanding climate-related vulnerability at the local scale and informing resilience actions. These Assessments contribute to the advancement of science-based policies, plans, and programs to promote effective climate leadership in California. In 2006, California released its First Climate Change Assessment, which shed light on the impacts of climate change on specific sectors in California and was instrumental in supporting the passage of the landmark legislation Assembly Bill 32 (Núñez, Chapter 488, Statutes of 2006), California's Global Warming Solutions Act. The Second Assessment concluded that adaptation is a crucial complement to reducing greenhouse gas emissions (2009), given that some changes to the climate are ongoing and inevitable, motivating and informing California's first Climate Adaptation Strategy released the same year. In 2012, California's Third Climate Change Assessment made substantial progress in projecting local impacts of climate change, investigating consequences to human and natural systems, and exploring barriers to adaptation.

Under the leadership of Governor Edmund G. Brown, Jr., a trio of state agencies jointly managed and supported California's Fourth Climate Change Assessment: California's Natural Resources Agency (CNRA), the Governor's Office of Planning and Research (OPR), and the California Energy Commission (Energy Commission). The Climate Action Team Research Working Group, through which more than 20 state agencies coordinate climate-related research, served as the steering committee, providing input for a multisector call for proposals, participating in selection of research teams, and offering technical guidance throughout the process.

California's Fourth Climate Change Assessment (Fourth Assessment) advances actionable science that serves the growing needs of state and local-level decision-makers from a variety of sectors. It includes research to develop rigorous, comprehensive climate change scenarios at a scale suitable for illuminating regional vulnerabilities and localized adaptation strategies in California; datasets and tools that improve integration of observed and projected knowledge about climate change into decision-making; and recommendations and information to directly inform vulnerability assessments and adaptation strategies for California's energy sector, water resources and management, oceans and coasts, forests, wildfires, agriculture, biodiversity and habitat, and public health.

The Fourth Assessment includes 44 technical reports to advance the scientific foundation for understanding climate-related risks and resilience options, nine regional reports plus an oceans and coast report to outline climate risks and adaptation options, reports on tribal and indigenous issues as well as climate justice, and a comprehensive statewide summary report. All research contributing to the Fourth Assessment was peer-reviewed to ensure scientific rigor and relevance to practitioners and stakeholders.

For the full suite of Fourth Assessment research products, please visit www.climateassessment.ca.gov. This report advances the understanding of how rangeland management can contribute to climate change mitigation and adaptation by repurposing California's organic waste-stream to compost for emissions reduction and carbon sequestration.

ABSTRACT

Land management offers significant potential to both help lower greenhouse gas emissions and reduce atmospheric carbon dioxide. The goals of this research were to determine the short- and long-term potential of compost amendments to sequester carbon (C) in rangeland soil, and to determine the effects of future climate change scenarios on C storage and loss. The project also explored the emissions from the composting process itself, a poorly quantified component of the waste-to-rangeland lifecycle. Finally, the lifecycle emissions from rangeland compost amendments were compared to those of other fates of waste to determine potential benefits or tradeoffs among a range of common practices. Compost amendments (0.25 inch) to 15 diverse rangelands led to a detectable increase in surface (0-10 cm) soil C stocks (2.1 ± 1.0 Mg C ha^{-1}) over a single growing season. The DayCent biogeochemical model was used to explore long-term effects of compost application in a subset of these rangelands, and to determine interactions with future climate change scenarios. Results showed that the overall climate benefit of compost amendments peaked 15 years after application. The benefit decreased over time, and decreased more quickly in a high emissions scenario. Two 100-day experiments using micrometeorological approaches yielded the first whole-pile, continuous measurements of greenhouse gas emissions from windrow composting. The total methane (CH_4) emission factors were 0.6 and 0.7 g CH_4 kg^{-1} feedstock, and were more sensitive to pile management than initial feedstock chemistry. Nitrous oxide emissions were below the instrument detection limit (25 ppb + 0.05%, or 4.5E-5 g m^{-2}) throughout the experiments. A lifecycle assessment model suggested that diverting organic waste to composted field amendments resulted in greater CO_2e savings compared to anaerobic digestion or incineration for energy, due to the combination of new C sequestration and emission reductions. In sum, results showed considerable potential for repurposing California's organic waste-stream to compost for emissions reduction and C sequestration. Rangeland compost application, where appropriate, can contribute to climate change mitigation, as well as improve ecosystem productivity and sustainability.

Keywords: compost, climate change mitigation, rangelands, lifecycle assessment model, DayCent biogeochemical model, carbon, greenhouse gases

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HIGHLIGHTS

- The addition of 0.25 inch of compost to a diverse set of California's rangelands resulted in a detectable and significant net increase in soil carbon storage (after removal of visible compost fragments) of 2.1 ± 1.0 Mg C ha^{-1} relative to untreated rangeland soils over a single growing season.
- Modeling the long-term effects of compost additions to rangelands showed that increased plant production and associated carbon sequestration are greatest within the first 30 years following a single compost application. Most sites reached a maximum sequestration potential approximately 15 years following a single application.
- Modeling results show that the benefits of compost application are sensitive to climate change. A lower emissions scenario (RCP 4.5) resulted in greater long-term benefits from compost application than a business-as-usual (RCP 8.5) scenario. Net carbon sequestration (both from the compost itself and increased plant growth and associated carbon storage) from a single compost application lasted approximately 85 years, and was dependent upon future climate change and location.
- The first whole-pile, continuous measurement of greenhouse gas emissions from windrow composting of livestock and green waste led to no detectable nitrous oxide emissions, and methane emission factors of 0.6 and 0.7 g CH_4 kg^{-1} wet weight feedstock. Few data sets exist on field-scale compost methane emissions, thus these data sets provide an important benchmark for future work.
- A lifecycle assessment of California's largest organic waste streams — food waste, yard waste, and cattle manure — showed that composting these feedstocks and land applying the compost to California rangelands has lower net greenhouse gas emissions than other waste management approaches, such as landfilling, anaerobic digestion, or incineration.

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1: Introduction

The soil holds the largest stock of organic carbon (C) on the earth's surface, amounting to more than three times the amount stored in the atmosphere (Köchy et al. 2015). Managed grasslands (henceforth rangelands) cover over 30% of the terrestrial land surface (White et al. 2000), 30% of U.S. lands, and approximately 50% of the land area in California (Brown et al. 2004) and have the potential to be important reservoirs for organic C (Jackoby and Jackson 2000). Rangelands are part of a biome that experiences seasonal water deficits. This phenomenon favors plant species that allocate much of their energy to root biomass in search of water and nutrients. High root biomass is often correlated with C-rich soils (Rasse et al. 2005), and thus healthy rangeland soils are frequently characterized by large organic C stocks (Lal 2002). However, much of the world's rangeland soils are depleted in C due to intensive management (Bai et al 2008). This loss of C, most of which has been oxidized to carbon dioxide (CO_2), not only contributes to climate change, but also leads to land degradation. Carbon is a primary component of soil organic matter, which plays a key role in soil and ecosystem health by enhancing soil water holding capacity, nutrient retention and availability, and soil stability (Conant 2011). Thus, increasing soil organic matter content, and by extension soil C stocks, can help mitigate climate change while improving the productivity and sustainability of agricultural soils.

Soil C storage and loss are sensitive to climate. Patterns in net primary productivity (NPP) in rangeland ecosystems, the primary conduit for C input to soils, varies strongly as a result of climate, particularly with interannual patterns in precipitation (Knapp and Smith 2001, Knapp et al. 2002, Chou et al. 2008). Models and long-term data suggest that regions of California may be getting drier as a result of climate change (Cook et al 2014). Increased frequency and severity of drought pose many management challenges, particularly for California's agricultural ecosystems. New management options are needed to maintain and restore healthy soil conditions on working lands in the face of climate change. Increasing soil organic matter content has considerable potential to augment soil water holding capacity (Ryals and Silver 2013) and increase ecosystem resilience to drought, rainfall variability (Haynes and Naidu 1998), and soil erosion (Reganold et al. 1987). Enhancing soil organic matter content also has important co-benefits including greater nutrient availability and improved nutrient retention. The improved soil health associated with increased soil organic matter content can stimulate NPP and crop production, and help mitigate climate change through increased plant C capture and soil C sequestration (Conant 2011, DeLonge et al. 2014). Research in California's Mediterranean grasslands suggested that composted green waste amendments increased soil water retention, NPP, and soil C storage for at least three years compared to untreated grasslands (Ryals and Silver 2013, Ryals et al. 2014, 2015). However, whether these results are transferable to drier ecosystems or composts from different feedstocks is not well understood. Moreover, while a lifecycle model indicated considerable climate change mitigation potential from waste diversion and soil application (DeLonge et al. 2013), uncertainty remains regarding the greenhouse gas emissions from the composting process and the relative controls on those emissions. The net C costs and benefits of composting relative to other fates of organic waste are also poorly understood.

1.1 Goals and Objectives

The purpose of this research was to determine the potential of composted organic waste to increase soil organic matter content, enhance plant growth, and contribute to climate change mitigation in California's rangelands. The research goals included: (1) Quantify the short-term (field) and long-term (model) effects of compost applications on C storage and nutrient cycling in California's rangeland soils and interactions with projected climate change; (2) Quantify greenhouse gas emissions and associated biogeochemical controls from composting organic waste; and (3) Use biogeochemical and life cycle assessment modeling to critically evaluate the climate change mitigation potential of composting and rangeland compost application at the state level. We used field and laboratory experiments in conjunction with modeling to build on an existing research program to determine the potential of compost production and application on rangelands for climate change mitigation and adaptation. We divided our activities into five overarching objectives:

Objective 1. Determine the short-term C and nutrient dynamics following compost application to rangelands across a broad suite of California's bioclimatic zones.

Objective 2. Use the DayCent biogeochemical model to estimate long-term patterns in soil and ecosystem dynamics across California's rangelands, including interactions with projected future climates.

Objective 3. Measure greenhouse gas emissions from the composting process.

Objective 4. Determine controls on greenhouse gas emissions from different feedstocks.

Objective 5. Use lifecycle assessment modeling to critically evaluate the greenhouse gas costs and benefits of composting relative to other fates of waste.

For Objective 1, we applied compost to treatment plots that were paired with untreated controls at each site. These data allowed us to determine the short-term impacts of compost on rangeland C stocks. The field data also allowed us to parameterize DayCent to explore the potential long-term, large scale C sequestration potential and net greenhouse gas dynamics as part of Objective 2. We used an emissions reduction climate change scenario (RCP 4.5) and the business-as-usual scenario (RCP 8.5) to generate weather output from the CanSEM2 Earth Systems Model. The weather data were used to explore the effects of compost applications under different climate change regimes. To better understand the potential emissions associated with the composting process itself, we measured the greenhouse gas fluxes from composting in Objectives 3 and 4. This entailed biogeochemical measurements of environmental conditions in the compost piles, as well as the chemical characteristics of the feedstocks, and associated emissions. Finally, in Objective 5 we used lifecycle assessment modeling to evaluate emissions

and sinks associated with the use of waste for compost relative to other potential fates. The compilation of these projects fills key gaps in knowledge about the conversion of waste to soil amendments for climate change mitigation. In the sections that follow, we provide details of the background for each objective, and the materials and methods used in our research. We then present our results, place them in a larger scientific context, and discuss the relevance of the findings and conclusions for California.

2: The short-term effects of compost applications on carbon storage and nutrient cycling in California's rangeland soils

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2.1 Introduction

Organic amendments have considerable potential to help rehabilitate degraded soils by increasing soil organic matter storage, enhancing water holding capacity, and providing nutrients that stimulate plant growth (Delonge et al. 2014). Livestock manure and green mulches from agricultural waste are common soil amendments (Diacono and Montemurro 2010). Livestock manure is widely used on rangelands in California, which is the largest producer of dairy cattle in the U.S. (https://www.ers.usda.gov/data-products/dairy-data/). While these practices can result in significant soil C sequestration, the C gain is often partly or completely offset by greenhouse gas emissions (Owen et al. 2015, Owen and Silver 2016).

Composting organic matter prior to land application can significantly reduce greenhouse gas emissions relative to the application of uncomposted organic material (Delonge et al. in prep). The composting process results in the complexation of organic molecules with nitrogen (N), effectively slowing decomposition rates and associated N mineralization. Mineral N is a precursor to nitrous oxide (N_2O) production; N_2O is a very potent greenhouse gas with an instantaneous global warming potential that is 298 times that of carbon dioxide (CO_2) over a 100-year timescale (Myre et al 2013). Although composting does result in some greenhouse gas emissions (Chapter 4), the complexation of C and N during the composting process and subsequent land application likely result in lower greenhouse gas emissions than the direct addition of more labile manure or green waste to soil (DeLonge et al. 2013, 2014). Land-applied compost can also lead to sustained increases in plant productivity. Ryals and Silver (2013) found that compost applications increased above- and belowground NPP by 0.54 ± 0.03 Mg C ha^{-1} to 1.45 ± 0.16 Mg C ha^{-1} y^{-1} (mean ± SE) for a coastal and valley grassland sites respectively. At the end of a three-year study, the amended plots had increased total soil C content by

approximately 1 Mg C ha^{-1} y^{-1} (Ryals et al. 2014). Compost amendments also significantly increased soil water content, and did not result in a decline in plant biodiversity or an increase in noxious weeds (Ryals and Silver 2013, Ryals et al. 2016).

While these results are promising, more data are needed from a wider range of bioclimatic zones to better determine the broad scale potential of compost applications for enhancing soil health, including soil C sequestration and climate change mitigation. In this study, we determined the short-term impacts of compost applications on soil C stocks at 15 sites across a wide range of bioclimatic conditions in California rangelands. The research objectives were: (1) determine if the short-term (1 year) effects of compost application varied across broad bioclimatic zones in California, (2) collect soil C data to parameterize the DayCent biogeochemical model for compost applications to California's rangelands, and (3) collect preliminary data from benchmark sites on working ranches to follow long-term effects of compost application on soil C sequestration, forage production, and soil water holding capacity.

2.2 Materials and Methods

The field component of this project was conducted in collaboration with the Natural Resource Conservation Service (NRCS) and Mr. Jeff Borum of the East Stanislaus Resource Conservation District who secured permission to work at the field sites, facilitated access, and participated in the field sampling. In late summer and early fall 2016, field plots were established at 16 grazed annual grasslands located across a wide range of bioclimatic zones in California (Figure A-1). At 14 sites, a 0.4 ha area (62.5 m x 62.5 m) was divided into two adjacent, 30 m x 62.5 m plots with a 5-m buffer area between plots (Figure 2.1). Existing paired 25 m x 60 m plots were used for the Yuba County and Marin sites, where compost application occurred in December of 2008. The Kings County site was damaged by a flood in early 2017 and thus dropped from the analyses.

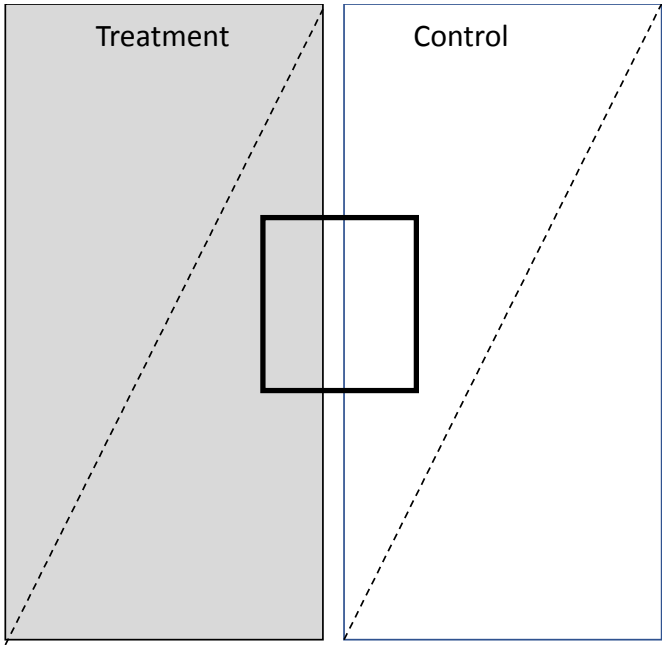

Figure 2.1: Experimental design for compost field trials. At each site, a 0.4 ha plot (62.5 m x 62.5 m) was divided into two equal 30 m x 62.5 m subplots with a 5 m buffer. One side was randomly

assigned as the compost treatment and the other side as a control. Each plot included a fenced grazing enclosure equally distributed across treatment and control (15 m x 30 m). Soil samples were collected across the subplot diagonal (dotted lines).

Initial soils were collected from each plot along a 45.7 m diagonal transect. We chose the diagonal transect in an effort to increase the spatial heterogeneity accounted for by our samples. Samples were collected at 10 m intervals (n = 5 sampling locations) from the 0-10 cm, 10-30 cm, 30-50 cm, and 50-100 cm depths. At some sites, it was impossible to sample below 30 or 50 cm; in these sites, samples were collect to the point of refusal and depths were recorded. Bulk density was sampled from one pit in each plot at 10 cm interval to 1 m following the protocol described in Ryals et al. (2014). To extract samples, a pit was dug with a backhoe, being careful to avoid disturbing the sampling face. Samples were carefully excavated from 5 cm behind a clean pit face to minimize the chances for compaction from pit establishment. Samples were collected using a 6.35 cm diameter metal corer.

Compost was added to one plot at each site (henceforth called the treatment plot) in November of 2016 (except for the two older sites mentioned above). The other plot was used as an untreated control. The same compost was added to all the new sites at a rate of 0.64 cm (0.25 inches) at a rate of 6.4 Mg C ha^{-1}. The compost consisted of a mixture of green waste and goat manure feedstocks with a C:N ratio of 17.6, an average N concentration of 1.7%, and a C concentration of 30% (Appendix A). The compost was produced at the West Marin Compost facility (westmarincompost.org). Information on the compost used at the Yuba and Marin county sites can be found in Ryals and Silver (2013).

Soils were resampled from each treatment and control plot at the end of the growing season in late April and early May of 2017. Soils were collected from the 0-10 cm and 10-30 cm depths along a new 45.7 m diagonal transect using the same procedures as above. Bulk density was sampled for the 0-10 cm depth as above, except the shallow pits which were dug by hand. Soils were processed at the University of California, Berkeley. Soil moisture was determined on 10 g of fresh soil that was oven dried at 105°C to a constant weight. Soil pH was measured in a 1:1 volumetric slurry of distilled deionized H_2O using a pH electrode (Denver Instruments, Bohemia, New York, USA). Three transect locations from each transect were analyzed for soil texture. Approximately 40 g of soil was first treated with hydrogen peroxide to remove organic matter and with sodium hexametaphosphate as a dispersant, and then analyzed using a Bouyoucos hydrometer (Gee and Bauder 1986). The remaining samples were air-dried and sieved using a 2-mm mesh and sorted to remove compost (for the spring samples only), roots, other organic debris, and rocks. Sorting insures that C and N analyses are capturing material incorporated into soil organic matter (both new C and N from plant inputs as well as decomposition of the amendment) and not merely recovery of the added compost. We note that finished compost has a very slow decomposition rate (k values of 0.04-0.05-^{y}, Lynch et al. 2005, Ryals et al. 2015). Subsamples were ground to a fine mesh and analyzed for total C concentration on a Carlo Erba Elantech elemental analyzer (Lakewood, NJ, USA) using atropine as a standard at a rate of one per ten samples. Soils were tested for carbonates using 2M HCl; as no carbonates were found, results reported reflect only organic C concentrations. Bulk density was determined by calculating the rock volume and determining the oven dry (105°C) mass of soil per unit volume. Soil C contents were calculated by multiplying the C and N concentrations

(%) by the oven-dry mass of the fine fraction (< 2 mm) and dividing by the bulk density and depth (Throop et al., 2012).

We used a laboratory incubation experiment to determine potential heterotrophic respiration rates, and ammonification and nitrification potential with and without compost addition. We chose the Santa Barbara County site because it represents an important bioclimatic zone in California and soil amendment impacts are not as well studied as valley and north-coastal grasslands (Ryals et al. 2013, Owen et al. 2015, Owen and Silver 2016). One composite soil sample was collected from each of three treatment and control plots at the 0-10 cm and 10-30 cm depths at the Santa Barbara County site (n = 12). Approximately 5 g composite samples were placed in 1-quart glass vessels and incubated at a constant temperature and field moisture for 9 weeks. Weekly soil CO_2 fluxes were measured using a Licor 6400 infrared gas analyzer (Licor Biosciences, Lincoln Nebraska). Additional soil samples were used to determine net ammonification and nitrification rates over a one-month period. Briefly, 5 g subsamples were extracted with 20 mL 0.5 M K_2SO_4; additional subsamples were incubated in glass vessels for 1 month in the dark at field moisture and then extracted with 20 mL K_2SO_4. A 5 g subsample was used to determine soil moisture content after drying at 105°C to a constant weight. Samples were analyzed on a Latchat auto-analyzer (Latchat Instruments, Loveland, CO) at the University of California, Santa Barbara. Net N ammonification and nitrification potential was calculated per gram of oven-dry equivalent soil according to Hart et al. (1994).

The data were analyzed using Systat 13. Analysis of variance was used to explore patterns across treatments and years. Note that this report focuses only on the 0-10 cm depth as one year was insufficient time to detect statistically significant changes in the subsoil. Subsoil samples will provide a benchmark for future analyses. All data were checked to ensure they met the assumptions for ANOVA; transformations were unnecessary. Statistical significance was determined as $P < 0.1$ unless otherwise noted. Values reported in the text are means plus and minus one standard error.

2.3 Results and Discussion

The study covered a wide range of soil conditions within the typical grassland soil orders of the region (Alfisols and Mollisols). California soils are diverse with respect to underlying geology and alluvial impacts (casoilresource.lawr.ucdavis.edu/gmap/). This was reflected in the texture analysis. Soil clay content ranged from 9 ± 0.7 % in Santa Barbara County to 37 ± 1 % in San Mateo County (Figure 2.2).

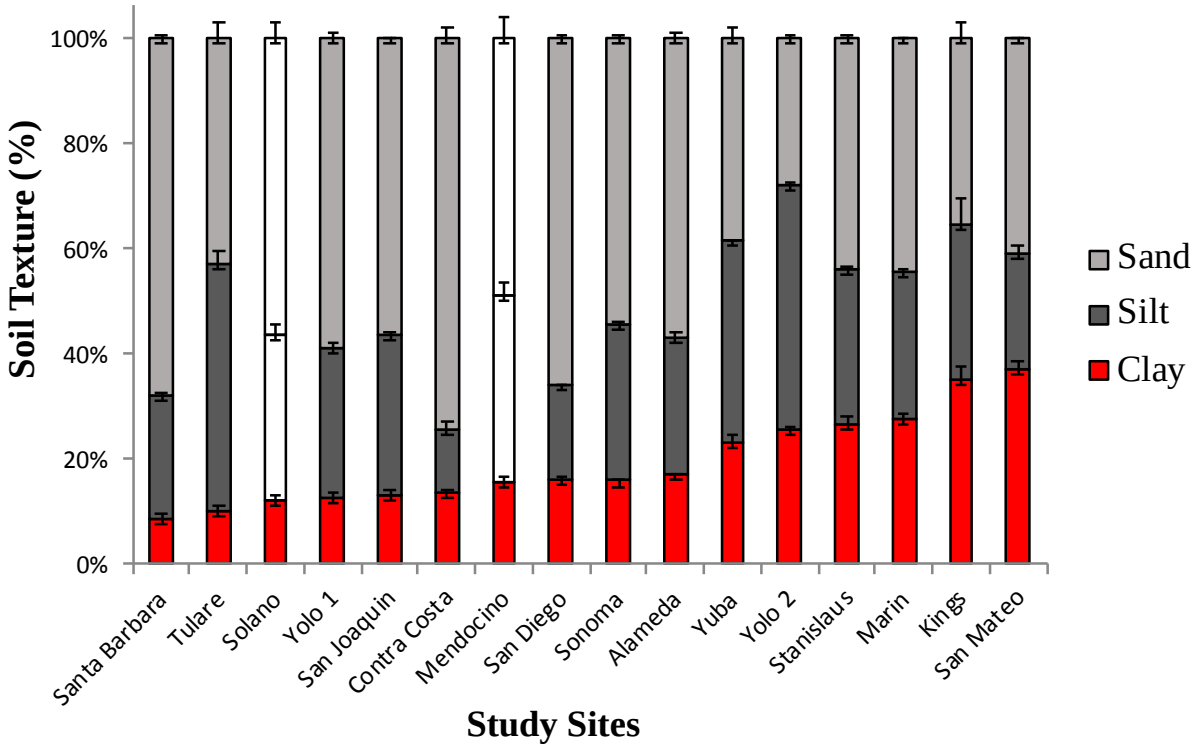

Figure 2.2: Soil textural analyses across the study sites. Samples came from the pre-treatment plots in 2016 for the 0-10 cm depth.

Soil pH ranged from a low of 4.63 ± 0.11 in Solano County to a high of 8.22 ± 0.07 in Tulare County. There was no statistically significant difference in soil pH within sites for the pre-treatment plots in 2016, or in 2017 between compost and control plots (for both new and older sites). In 2017, soil pH was slightly higher in both treatment and control plots than at the start of the experiment ($P = 0.06$ control, $P < 0.01$ treatment). This most likely reflected the different season of the year for the two sampling periods. Soil bulk density in the 0-10 cm depth averaged 1.19 ± 0.03 g cm^{-3} in the new sites and 1.21 ± 0.07 g cm^{-3} in the old sites. There were no statistically significant trends with year or compost application. Compost amendments can decrease bulk density (Celik et al. 2004, Bronik and Lal 2005), although the addition rate here was low and effects were not detectable at this scale of resolution.

Average soil C stocks ranged from 11 to 108 Mg C ha^{-1} across samples in the top 10 cm of mineral soil, with an overall mean across the two sampling periods of 27 ± 13 Mg C ha^{-1}. This is similar to the 33 ± 4 Mg C ha^{-1} reported for a literature survey of California's rangeland soils (Silver et al. 2010). The site with the largest soil C stocks was a peatland soil in Contra Costa County (108 Mg C ha^{-1} in 2016). This site represented an extreme outlier in terms of C concentrations and variability (2 to 6 times higher than all other sites) and was thus dropped from the remaining analyses. Further research at this site with larger sample sizes will be needed to account for the high variability in C stocks and to detect possible treatment effects. There was a weak relationship between soil C content and clay content (pretreatment soil only; $r^2 = 0.24$, $P < 0.01$). Within sites, there was no statistically significant difference between

pretreatment plots. We used the average difference between the treatment and control plots within sites as an index of the short-term effects of compost on soil C stocks (Figure 2.3). Compost amendments led to an average increase in soil C stocks of 2.1 ± 1.0 Mg C ha^{-1} when including all non-peat treatment-control plot comparisons ($P < 0.05$). When only including the newer sites, compost added 1.9 ± 1.1 Mg C ha^{-1} ($P = 0.05$). There were no statistically significant differences in C concentrations between treatment and control plots in 2017 (Figure 2.4), so the patterns in C stocks was due to the combined effects of bulk density and C concentrations.

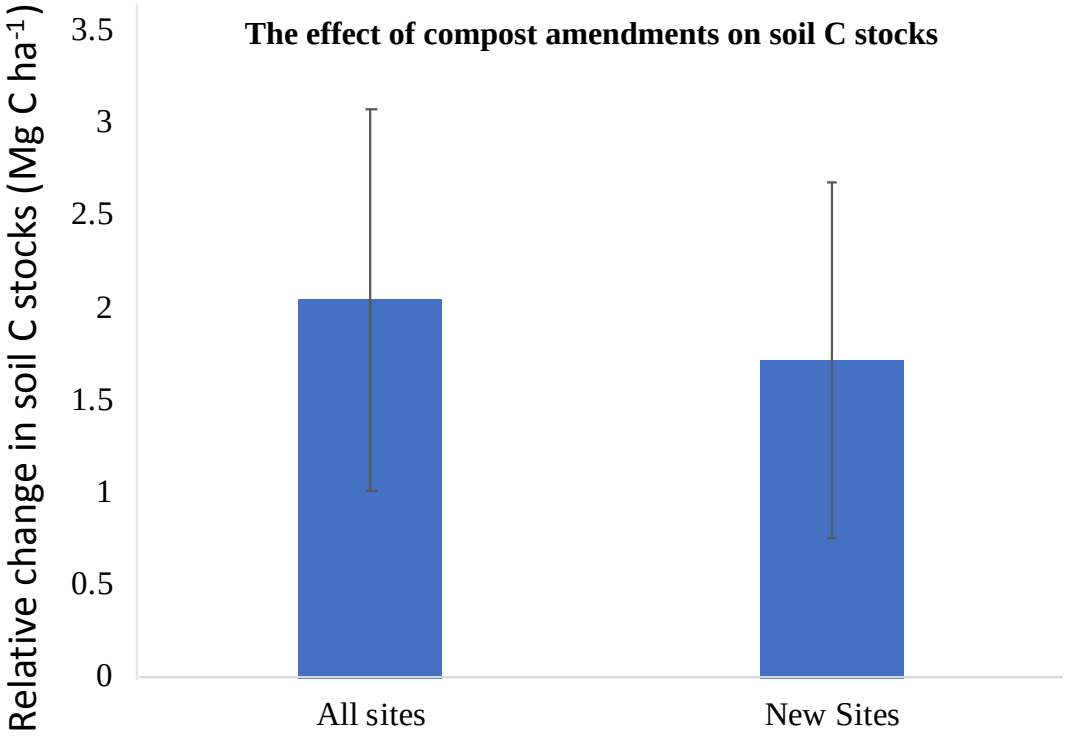

Figure 2.3: The effect of compost amendments on soil C stocks (mean values of the within site treatment minus control) at all non-peatland sites and new sites only. Means for both analyses were significantly greater than zero.

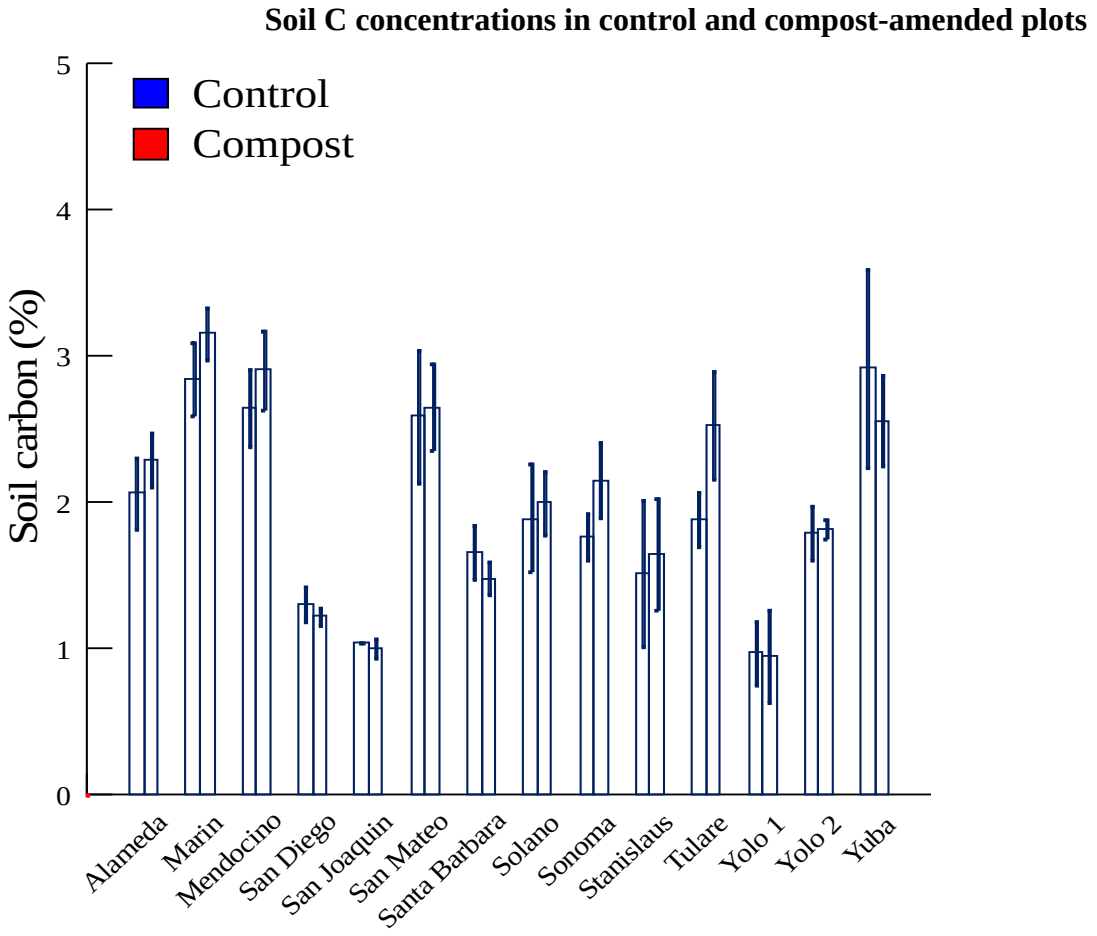

Figure 2.4: Soil C concentrations in control and compost-amended plots at 14 rangeland sites across California. Data are for the 0-10 cm depth. As there was only one bulk density pit per plot we report C concentrations for within-site comparisons.

Previous work in Marin and Yuba Counties also showed significantly greater soil C in compost-amended rangeland soils (Ryals and Silver 2013, Ryals et al. 2014). Our results likely reflect both an increase in plant growth as well as the incorporation of composted organic matter into the soil matrix. It is important to note that soils were carefully sieved and sorted for organic matter fragments, so the majority of the C measured here reflects material that is visually indistinguishable from the native soil matrix.

We used the Santa Barbara County site to explore mechanisms of response to compost amendments in a dry Mediterranean grassland (Marchus and Schimel et al. in prep.). Potential soil respiration rates measured from the Santa Barbara County soils were higher in the soil amended with compost than the control soils for the 0-10 cm depth (Figure 2.4; $P < 0.05$). This is similar to the response measured in more mesic valley and north-coastal grasslands measured by Ryals and Silver (2013) and likely represents stimulation of the microbial community with the addition of new substrate. Potential net nitrification rates (Figure 2.5) averaged 22 ± 1 µg NO_3^--N per g over 30 days in the controls and 17 ± 1 µg NO_3^-N per g over 30 days in the compost-amended plots ($P < 0.05$). There was no significant effect of compost addition on ammonification, which averaged -0.35 ± 0.17 µg NH_4^+-N per g over 30 days. These results suggest that compost was effective at slowing net nitrification in these soils.

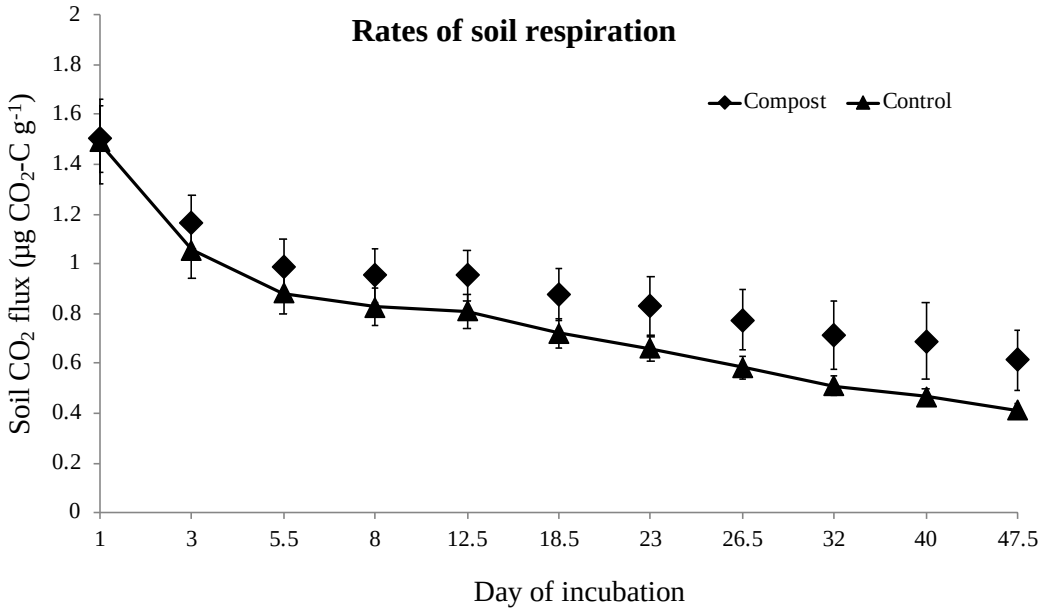

Figure 2.5: Rates of soil respiration during laboratory incubations of soils from the Santa Barbara County rangeland site. Data from Marchus and Schimel et al. in prep.

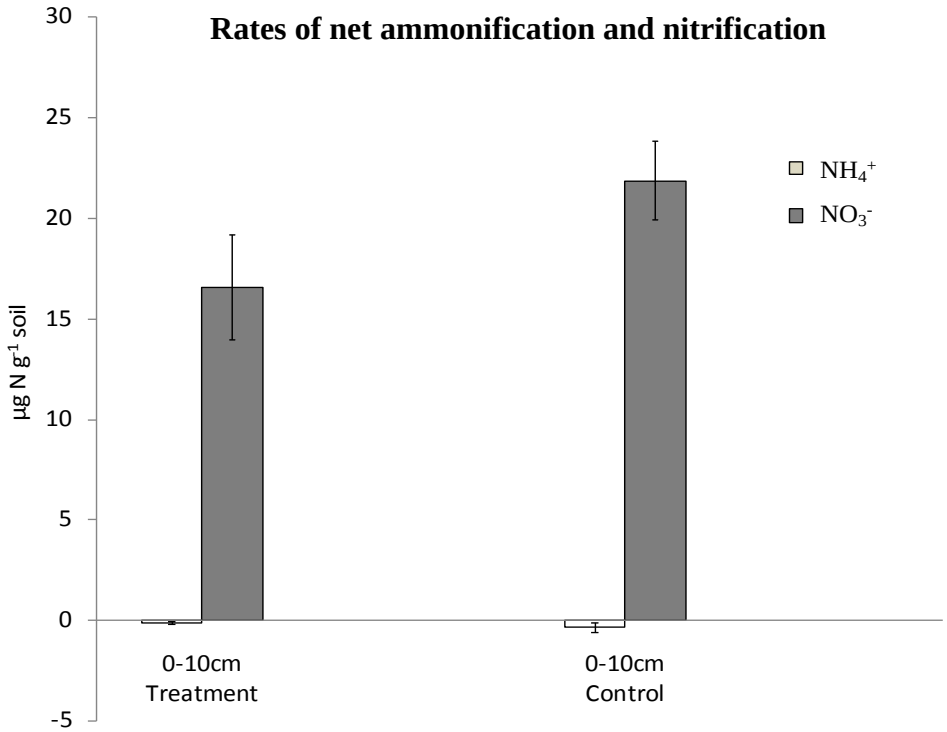

Figure 2.6: Rates of net ammonification and nitrification over a 30-day period during laboratory incubation of soils from the Santa Barbara County rangeland site. Treatment soils are from the compost plots. Data from Marchus and Schimel et al. in prep.

2.4 Summary

Our results show that compost amendments led to a detectable increase in surface soil C stocks over a single growing season across a diverse set of rangeland sites in California. The patterns were most likely due to the combined effects of C concentration and bulk density, as neither of these individual factors differed significantly between the treatment and control plots. We note that these short-term results should be taken with caution because grassland soil C stocks can fluctuate in surface soils as a result of plant litter inputs and decomposition, both of which are sensitive to climate (Parton et al. 1995). However, the overall increase is likely to be a robust result and is in agreement with previous work at a valley and coastal grassland site in California (Ryals and Silver 2013, Ryals et al. 2014). We measured an increase in potential soil respiration rates in the compost-amended soils from the Santa Barbara County rangeland. Soil heterotrophs from this relatively dry site were surprisingly responsive to the addition of compost. While the increase in soil CO_2 production led to an increase in soil C losses from the ecosystem, it represents a small proportion of the C added and does not contribute to the global warming potential of this practice (as the waste C would have degraded anyway). The soils with compost amendments exhibited lower potential net nitrification rates than the controls. In summary, results demonstrate significant soil C sequestration over the short-term with compost amendments to rangeland soils across a wide range of bioclimatic conditions. In the next section, we explore the longer-term potential using a modeling approach, and determine the possible interactions with a changing climate.

3: The long-term carbon sequestration potential from compost amendments in California's rangelands and interactions with climate change

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3.1 Introduction and Objectives

Effective climate change policy for natural and working lands should be based on a solid scientific understanding of ecosystem processes. Field studies can provide critical data on the effects of alternative management options. For example, field studies from managed grasslands in Marin and Yuba counties showed that a one-time addition of compost significantly increased plant productivity and soil C storage for several years (Ryals and Silver 2013, Ryals et al. 2014). Results from the present study yielded similar early-stage results at a statewide scale (see Chapter 2). While field studies provide valuable information over the short-term, few studies

have been conducted over long (>10 y) time periods. The lack of long-term studies makes it difficult to predict future effects of management activities on ecosystem dynamics. The lack of long-term research also hinders our ability to understand the effects of past and current climate change on working lands and interactions with management.

Biogeochemical models are excellent tools for exploring future behavior of ecosystems under specific management practices, and in the context of climate change (Melillo et al. 1995). Here, we used the DayCent biogeochemical model (Del Grosso et al. 2001) to explore the effects of compost application across a latitudinal and climate gradient throughout California. The model simulates grassland productivity and the movement of C between soil, vegetation, and the atmosphere over time and under different climate and management conditions.

This study aimed to explore the scalability of compost amendments on rangeland soils across space and time. Our research addressed the following questions:

- How does compost addition affect long-term net primary production and soil C storage in California rangelands?
- How do environmental variables affect biogeochemical cycling in rangelands, and how does background climate interact with compost impacts?
- How does projected future climate change influence soil C storage, and how does compost application impact C dynamics under potential future climate conditions?

3.2 Methodology

3.2.1 Site Description

We parameterized the model using seven annual grassland sites that are representative of a broad range of California's grassland climates. These seven sites are part of a larger NRCS and UC Berkeley field experiment where compost was applied in fall of 2016 to plots in a total of 15 sites (see Chapter 2 and Appendix B Figure B.1). Compost application at the Marin and Yuba sites took place in 2008. Specific pre-treatment field observations from the early Fall 2016 were used to parametrize the model for each site, and the longer-term field results will eventually be used to validate the model results from this study. All sites were managed rangelands and have been grazed for most of the last century. The four coastal sites (Mendocino, Marin, Santa Barbara, and San Diego) and two inland sites (Solano and Yuba) have a Mediterranean-type climate (cool, wet winters and warm, dry summers), and are dominated by nonnative annual grass and forb species. The third inland site (Tulare) experiences a semi-arid climate, also with annual grass and forb species. The Mendocino site is in Covelo, CA (39.84°N, 123.257°W) with soil classified as Cole loam Argixeroll (Mollisol). The Yuba site is at the Sierra Foothills Research and Extension Center in Brown's Valley, CA (39.34°N, 121.35°W) with soil in the Aubern-Sobrante complex classified as Mollic Haploxeralfs (Alfisol and Inceptisol). The Marin site is in Nicasio, CA (38.06°W, 122.71°N) in the Tocaloma-Saurin-Bonnydoon soil series classified as a Typic Haploxeroll (Mollisol). The Solano site is in Suisun City, CA (38.21°N, 122.03°W) in the Antioch-San Ysidro Complex, with soils classified as a Typic Natrixeralf (Alfisol). The Santa Barbara site is in Los Olivos, CA (34.71°N, 120.13°W); soils are a Ballard gravelly fine sandy loam, classified as a Typic Argixeroll (Mollisol). The Tulare site is in Exeter, CA (36.33°N, 119.17°W); soils are in the Akers complex, and are characterized as Calcic Haploxerept

(Inceptisol). The San Diego Site is in Santa Ysabel, CA (33.15°N, 116.69°W), at higher elevation (1,135 m) compared to the other sites. The soil is Holland fine sandy loam, characterized as an Ultic Haploxeralf (Alfisol). Additional site characteristics are described in Table 3.1.

Table 3.1: Characteristics of modeled sites. ANPP= aboveground net primary productivity; SOC= Soil Organic Carbon; MAP= Mean annual precipitation (1975-2005). Data are from Silver et al. 2018 (above) and local CalClim station data.

Site	Observed ANPP ($Mg\ C\ ha^{-1}$)	Observed bulk SOC (0-30 cm) ($Mg\ C\ ha^{-1}$)	% Clay (0-30 cm)	% Sand (0-30 cm)	Historic 30 yr MAP (cm)	Mean min. daily temp. (°C)	Mean max. daily temp. (°C)
Mendocino	0.6 – 0.9	29.55	16%	49%	108	4.6	22.3
Marin	1.0 – 2.0	40.95	27%	44%	97	8.3	20.0
Santa Barbara	1.8 – 2.0	21.07	9%	67%	38	8.0	25.1
San Diego	0.4 – 1.0	15.03	16%	66%	67	7.2	21.0
Tulare	0.9 – 2.0	23.12	10%	43%	28	10.8	24.1
Solano	1.0 - 1.5	23.75	12%	57%	61	8.8	23.3
Yuba	1.5 – 2.5	22.33	23%	39%	73	10.3	24.4

3.2.2 Model Simulation Methodology

DayCent (Parton et al. 1998) was used to simulate the effects of climate and management in each rangeland ecosystem. The model is driven with site-specific historic climate data, as well as measured soil texture, bulk density, and annual forage production values. DayCent partitions existing and added into three pools: active (<1 year turnover), slow (decadal turnover), and passive (millennial turnover) C stocks. Dead plant material is partitioned into active or slow cycling pools initially, depending upon the structure of the material. Carbon can move among pools through decomposition and stabilization. The movement among pools mimics microbial activity and mineral association of organic matter, but DayCent does not explicitly model specific mechanisms of microbial interactions or mineral stabilization (Parton et al. 1994). Soil C flows and NPP are both strongly dependent on water availability in DayCent. DayCent is a useful tool for this study because it facilitates the simulation of explicit management practices including grazing and compost amendments, and was originally developed, and has been used extensively, for modeling grassland ecosystems.

The model simulations were run for a 3000-year period for each site using the measured soil texture values and assuming perennial grassland coverage to achieve steady state values for the C pools, before running perturbation simulations. Model parameters were adjusted so that the model output matched observed net primary productivity (NPP) for each site under current management conditions. Simulations of future conditions were driven by daily climate data extracted from the CanESM2 Earth System Model. There remains debate as to which Earth system model most accurately represents future weather in California. We used CanESM2 because it was one of the four models recommended by the California 4th Climate Assessment for analyses of climate impacts in California. We used the Representative Concentration Pathway (RCP) 4.5 scenario, which assumes some emissions reductions, and the RCP 8.5 scenario that assumes a business as usual scenario with minimal emissions reductions. Data were extracted for the site-specific (2.8°x 2.8°) geographical grid of CanESM2. The RCP 8.5 scenario differs from the RCP 4.5 scenario in that there is a pronounced increase in daily temperature, especially in daily minimum temperature across all of the sites. The RCP 8.5 scenario also results in increased annual precipitation and interannual precipitation variability in the last half of the century in the Southern California sites. Thus, the RCP 8.5 scenario as extracted from the CanESM2 model simulates a "warmer and wetter" climate for most sites. For each climate scenario, we ran a control run assuming that current management continued throughout the century. We also did a simulation with a compost trial consisting of a one-time 0.25 inch addition of compost to the site. The compost addition replicated the actual management of the field experiment (Chapter 2). The compost amendment added C at a rate of 640 g C m^{-2} (6.4 Mg C ha^{-1}) with a C:N ratio of 17.6. The baseline year for compost amendment was 2016 for all sties except for Marin and Yuba (baseline year 2008), which follows the field trials. Soil C and NPP results are reported as the relative difference between the compost-amended plot and the control plot at each site for each time point unless otherwise noted. The NPP data was smoothed using a Generalized Additive Model. We also calculated 95% confidence intervals. Data analyses were conducted in R.

3.3 Results and Discussion

Under the RCP 8.5 scenario of the CanESM2 climate model, projections for mean annual precipitation exhibited increases across the 21st century (comparing 2000-2010 to 2090-2100), ranging from an additional 3% (36 mm yr^{-1}, $P<0.01$) in Marin County to an additional 33% (180 mm yr^{-1}, $P<0.01$) in San Diego County (Appendix B Figure B.2). The most significant increase in annual precipitation occurred in Tulare County, with an additional 80 mm yr^{-1} by the end of the century ($P<0.001$). Under RCP 8.5, four out of seven sites experienced a substantial increase in precipitation variability at the end of the century (Appendix B Figure B.2**)**. The standard deviation of interannual precipitation increased by 50-70% in Tulare, Solano, San Diego, and Santa Barbara Counties, while Yuba, Marin, and Mendocino Counties experienced a change in standard deviation of interannual precipitation of 25-30%. Mean annual precipitation did not increase significantly under RCP 4.5. Mean temperatures were also affected by climate change (Figure 3.1). Mean minimum temperatures increased by 2.5°C or less in the RCP 4.5 scenario ($P<0.0001$ for all sites), and between 3.6 and 6°C in the RCP 8.5 scenario ($P<0.0001$ for all scenarios). Mean daily maximum temperatures also increased significantly at all sites, between 5.6 and 6.7°C ($P<0.0001$).

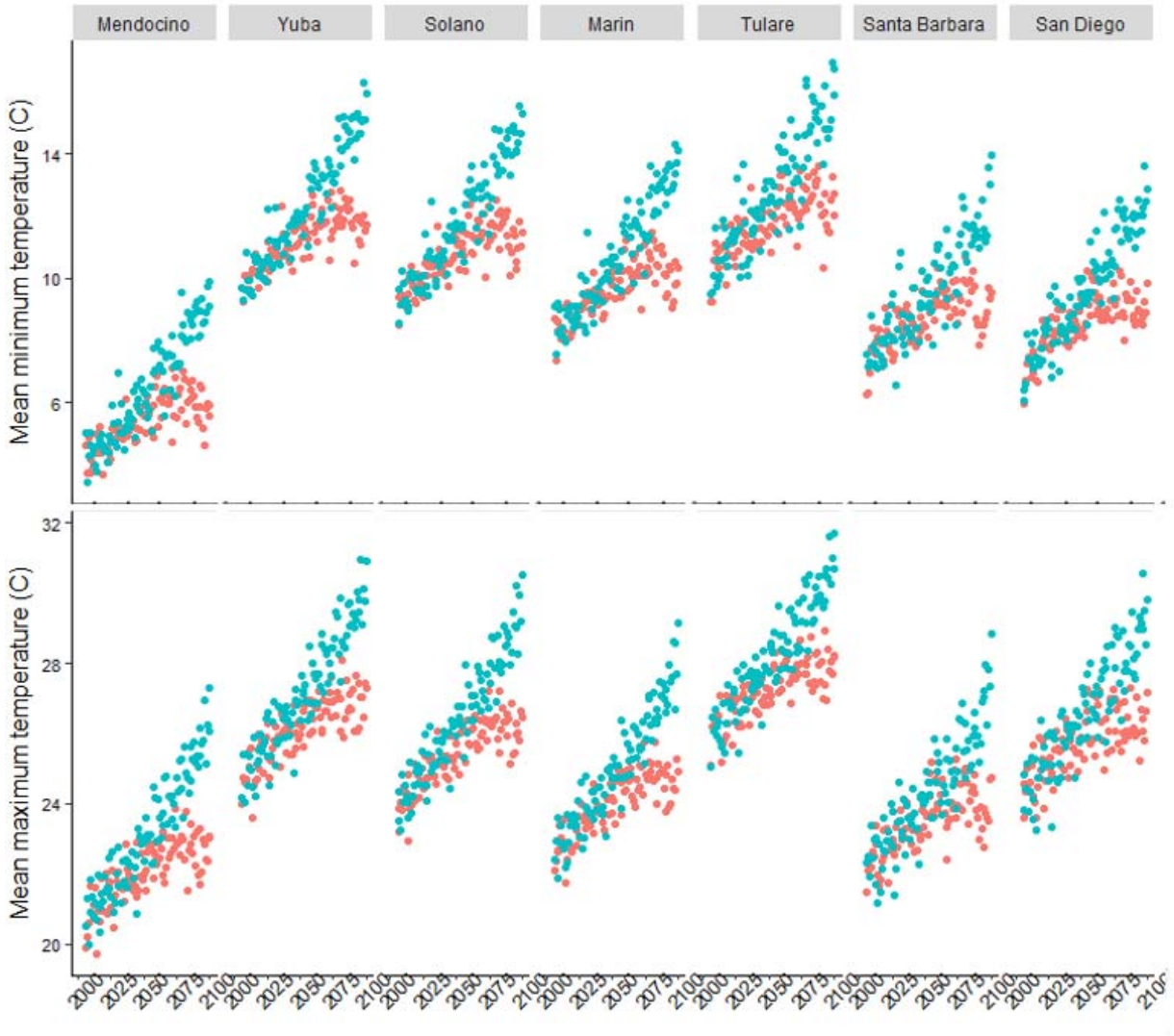

Figure 3.1: Mean daily temperature increased more rapidly throughout the century in the RCP 8.5 scenario (blue) compared to the RCP 4.5 scenario (red).

3.3.1 Net Primary Productivity

A single addition of compost resulted in higher NPP in the compost treated plots relative to the control plots (Figure 3.2). The compost treatment had higher NPP in all seven geographically diverse sites, despite the high interannual variability in NPP due to rainfall. This increase in above- and belowground productivity was largely responsible for the increased movement of C into soil (Figures 3.2, 3,3b). Compost can increase soil water holding capacity and act as a slow-release fertilizer (Diacono and Montemurro 2010), thus vegetation growth received an initial boost in growth rates following the modeled applications. This boost of productivity resulted in higher photosynthetic uptake of atmospheric C and accumulation of C in both above- and belowground tissues. The pattern of increased productivity and soil C storage persisted for more than a decade past the initial compost application. While productivity stopped actively increasing 15 years after the soil was amended, NPP in the compost amended simulations remained higher than in the control simulations at least until the end of the century.

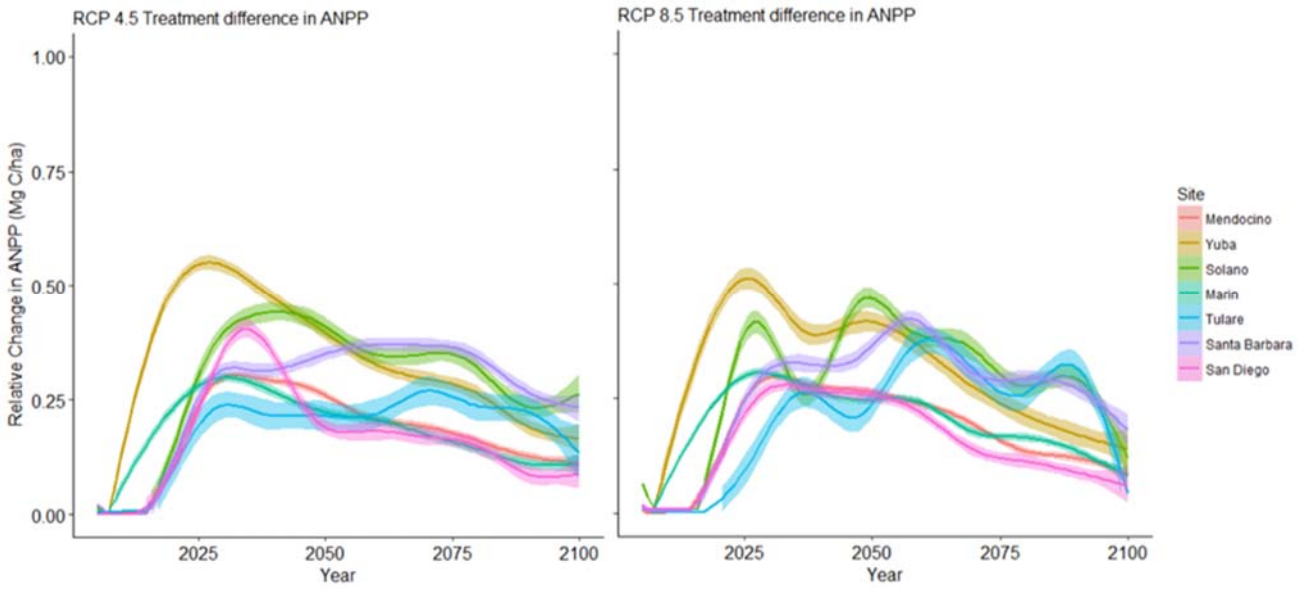

Figure 3.2: Net primary productivity increased in the compost treated plot relative to the control plot in all seven sites. The increase in net primary productivity lasted through the end of the century under both climate scenarios. The results presented are smoothed conditional means using a Generalized Additive Model to fit the data. The shaded areas represent 95% confidence intervals.

3.3.2 Soil Carbon Storage

A one-time application of compost resulted in greater soil C stocks in all three of the modeled soil C pools: the active pool (turnover time of days to one year), the slow pool (turnover time from decades to one century), and the passive pool (turnover time from centuries to millennia) (Figure 3.3a). The effect on bulk soil C was dominated by an increase in the slow C pool. Values exceeded baseline scenarios at all sites and all pools for the entire period of analysis (Figure 3.3a). The increase in the slow C pool was greater in RCP 8.5 than in the RCP 4.5 scenario during the first few decades after compost addition, but the trend reversed as climate conditions diverged in the second half of the century.

The largest gain in soil C occurred in 2031 in Mendocino, where soils gained +1.91 Mg C ha^{-1} and +1.92 Mg C ha^{-1}more in the compost treated soils than in the control for the RCP 4.5 and 8.5 scenarios, respectively. The smallest increase in soil C was in San Diego, which peaked in 2031 under the RCP 4.5 scenario, with a maximum increase of +1.73 Mg C ha^{-1}in the composted compared to the control simulation. For the RCP 8.5 scenario, the peak C gain in San Diego was +1.67 Mg C ha^{-1}. The San Diego site had the lowest initial soil C content, as well as one of the lowest average rates of NPP. The higher altitude of the San Diego site yields a cooler and wetter climate than the other southern Californian sites, making the results more comparable to the northern Californian sites than the South Central California sites.

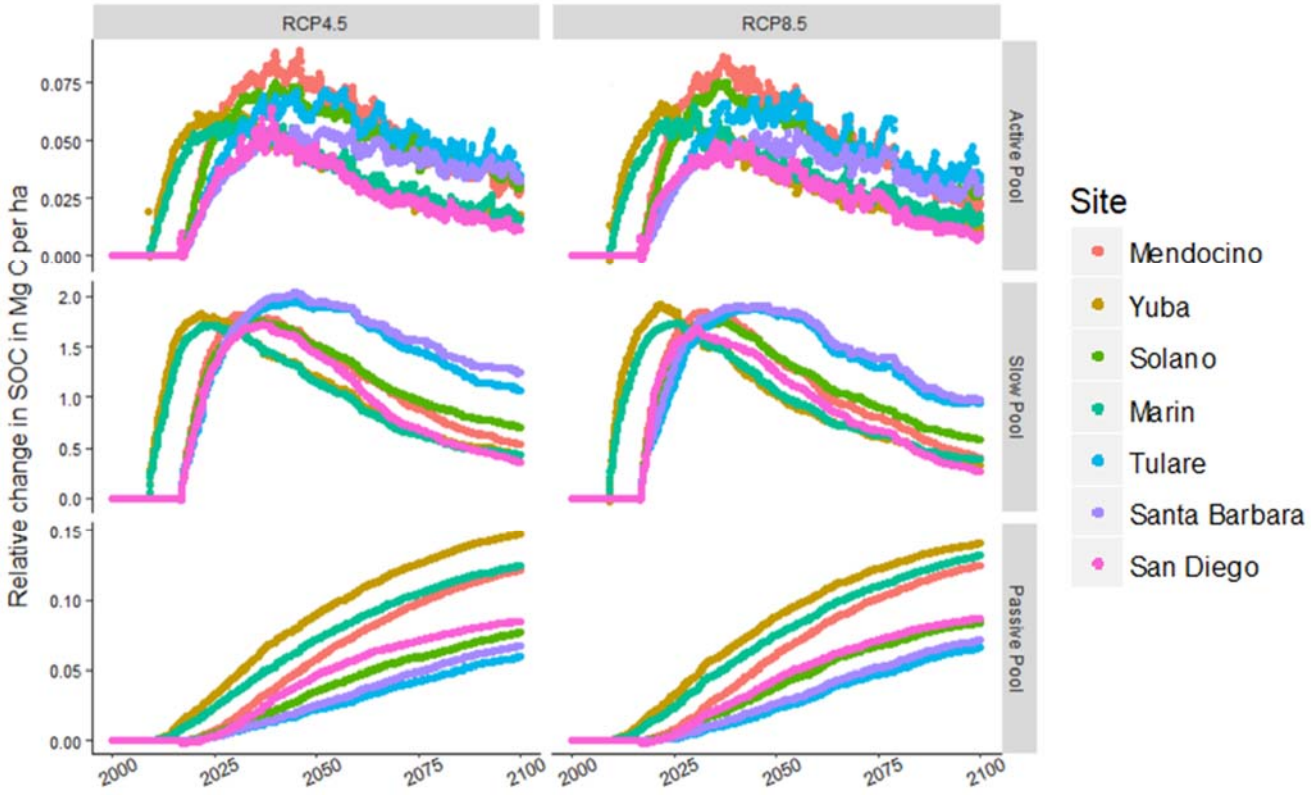

Figure 3.3a: Total soil organic C stock increased in the compost treatment relative to control in all three modeled soil C pools. Soil organic C increased at all seven sites under both climate scenarios.

The increase in soil C was due to both the direct addition of C through the compost amendment as well as an indirect increase in soil C inputs from NPP (Figure 3.3b). Compost C had largely decomposed by the end of the century. The indirect benefit of compost to the ecosystem resulted in additional C drawdown of 0.3 Mg C ha-1 in San Diego County by mid-century, to 0.9 Mg C ha-1 by the end of the century. In the latter half of the century the climate in most sites in RCP 8.5 was wetter and warmer than in RCP 4.5. The fraction of additional C allocated to the slow, decadally cycling pool was greater in RCP 4.5, while the fraction of additional passive pool C was higher in RCP 8.5. This change in C allocation from slow to passive C in RCP8.5 may have been driven by decomposition of slow C due to the more favorable (warm, wet) conditions, accelerating the movement of C from the slow pool to both the atmosphere and to the passive, more stable C pool. The warmer, wetter conditions could accelerate movement of C through the mineral soil and increase instances of sorption to mineral surfaces or could facilitate passive C stabilization through greater soil aggregation from enhanced soil structure.

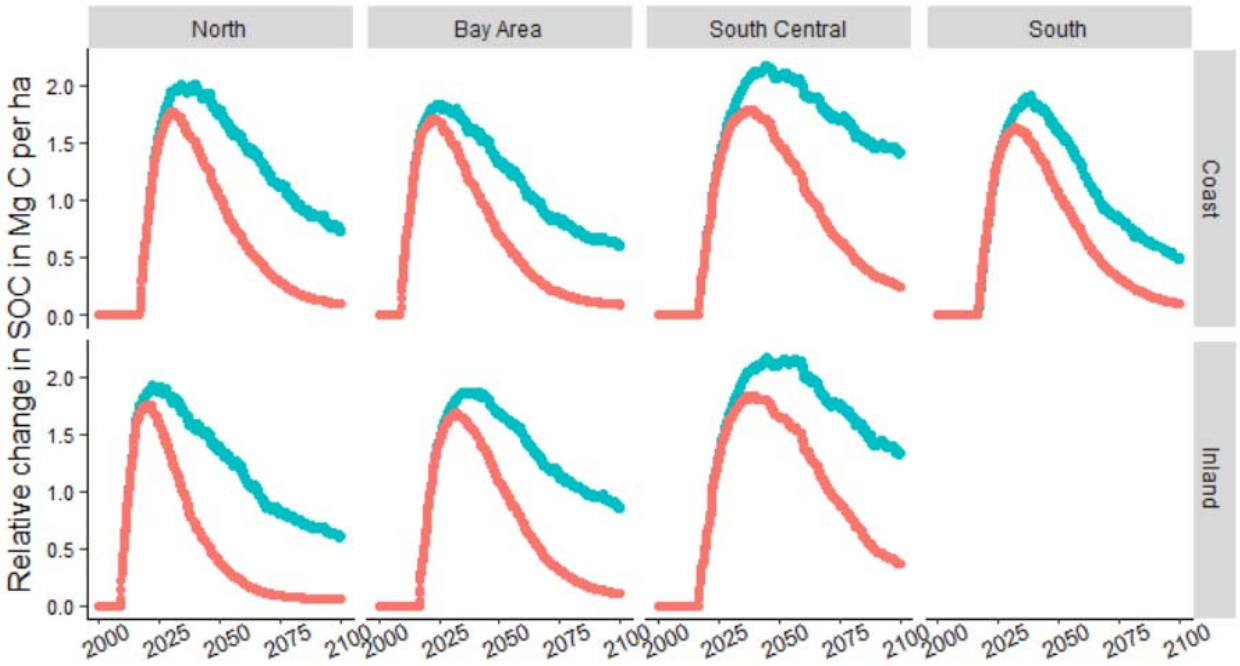

Figure 3.3b: Relative change in total soil organic C stocks (blue lines) including C added indirectly through the boost in net ecosystem productivity, and the C added directly via compost (red lines).

3.3.3 Climate Change Mitigation

The increase in soil C due to compost applications was accompanied by a stimulation of greenhouse gas emissions, but the climate benefit of gross soil C inputs (measured in CO_2 equivalents, CO_2e) outweighed the emissions (Figure 3.4a). Loss of C through CO_2 emissions was accounted for in the total soil C stock, and DayCent for grasslands does not have a module to calculate CH_4 emissions, as CH_4 is normally consumed in grassland ecosystems. Therefore, the emissions represented here are cumulative N_2O emissions due to the addition of compost. The net climate benefit (gross soil C inputs minus emissions) was maximized 15 years after compost application, and remained positive through the end of the century in the RCP 4.5 scenario (Table 3.4). The net climate benefit was highest in the two South Central sites of Santa Barbara and Tulare Counties, while the net climate benefit decreased more rapidly at the other sites, particularly at the wettest and Northern-most site of Mendocino County (Figure 3.4b). Under the RCP 8.5 scenario, precipitation increased over time, resulting in N_2O emissions. By 2100, there was a small source of 0.3 Mg C ha^{-1} in the Mendocino County site (Appendix B Figure B.3). Ryals et al. (2015) compared field observations from static flux chamber measurements every two to four weeks and DayCent output for the Marin and Yuba County sites and showed that the model overestimated N_2O fluxes from both rangelands. We therefore assume that the model overestimated N_2O fluxes here, and thus our C balance likely underestimates the net C sink of the soil associated with compost amendments.

Compared to the RCP 4.5 scenario, the net climate benefit of compost application in the RCP 8.5 scenario decreased more rapidly over time at all sites (Appendix B Figure -3). This indicates that a given C sequestration activity has a greater climate benefit when combined with emissions

reductions, creating a positive feedback of mitigation activities and effect on climate, i.e., a virtuous cycle.

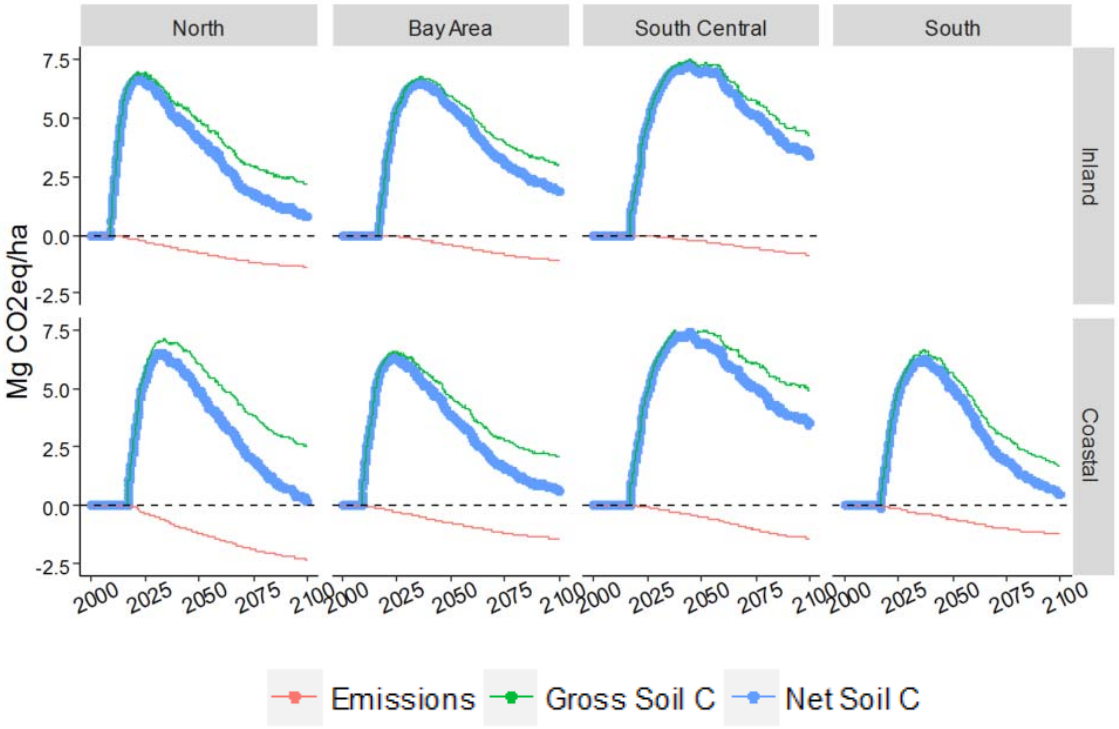

Figure 3.4a: Total enhanced soil C storage due to compost (Gross soil C: green line) was greater than greenhouse gas emissions stimulated by compost application to soil (red line), resulting in a net climate benefit (Net soil C sequestration: blue line) for all sites through the end of the century (RCP 4.5).

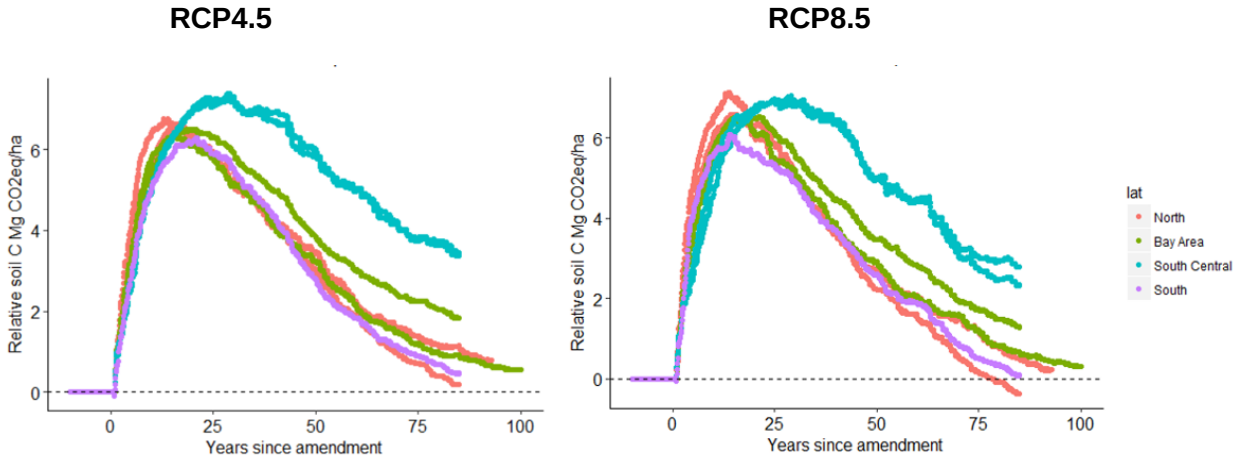

Figure 3.4b: Net climate benefit (Gross soil C inputs minus greenhouse gas emissions) for all seven sites were positive through the end of the century under RCP 4.5. The two northern sites (red), had a similar decreasing net climate benefit as San Diego County in the south (purple), while the Bay Area sites (green) had a slightly longer lasting climate benefit. The two driest sites of Santa Barbara and Tulare Counties in South Central California (blue) had the largest and longest climate benefit due to compost. With greater climate change in the RCP 8.5 scenario, all sites exhibited reduced climate benefit in the latter half of the century, and even a net loss of C from the system by the end of the century in the wet, Mendocino County site (see Figure A-3).

Table 3.4: Model output showed increased relative net climate benefit

Site	CanESM2 (2005-2025) Mean summer (JJA) max. temp. (°C)	Model output mean annual aboveground NPP ± s.e. (Mg C ha^{-1})	RCP4.5 Maximum relative change in net climate benefit (Mg CO_2e ha^{-1})	RCP8.5 Maximum relative change in net climate benefit (Mg CO_2e ha^{-1})
Mendocino	30.4	0.81 ± 0.04	6.48	6.57
Yuba	35.4	1.59 ± 0.13	6.05	5.86
Marin	29.2	1.42 ± 0.05	6.3	6.36
Solano	32.5	1.25 ± 0.06	6.29	6.49
Santa Barbara	26.1	1.79 ± 0.13	6.34	6.36
Tulare	36.4	1.14 ± 0.12	6.27	6.02
San Diego	32.8	0.78 ± 0.08	5.88	6.02

JJA= June, July and August; NPP= Net Primary Productivity; Net climate benefit = C inputs – N_2O Emissions

We used the U.S. Geological Survey Ecoregions to scale the climate benefit from each of these sites to other rangelands within the same sub-ecoregion (Griffith et al. 2016). Assuming that the compost application would have the same climate benefit within each sub-ecoregion, we conservatively estimated that applying compost to only 6% of California rangelands (Flint et al, this volume), would sequester a cumulative 8.4 to 8.7 million metric tons of CO_2 equivalents 15 years after compost amendment. Note that this does not include C and greenhouse gas savings from waste diversion (Chapter 5). The C sequestration achieved through applying compost to this 6% of California rangelands would accomplish about half of the goal set by California's AB32 to avoid 15-20 million metric tons of CO_2e by 2030.

3.4 Summary

A one-time application of compost across a broad suite of rangeland sites in California resulted in a long-term increase in overall soil C storage and NPP. The climate benefit of the compost amendment peaked at 15 years after application. The benefit decreased over time, and decreased more quickly in the RCP 8.5 high emissions scenario. We emphasize that long-term trends in soil C are model estimates and thus the magnitude and duration could vary under field conditions.

Climate change in California is projected to increase the variability of rainfall, and under the CanESM2 model total rainfall is projected to increase as well; these changes are expected to impact greenhouse gas emissions and soil C sequestration. In the wetter Mendocino County site, change in precipitation led to greater greenhouse gas emissions. Soil C sequestration rates were maximized within the first 15 years after addition, and more than offset greenhouse gas emissions for many decades longer. The two driest sites in Santa Barbara and Tulare Counties both had a more positive C balance (net sequestration) in both RCP scenarios, indicating that the climate benefit of compost amendments at drier sites were not as sensitive to the projected increase in both total precipitation and precipitation variability. Our results indicate that emissions reductions at a global scale (i.e. the RCP 4.5 scenario) led to longer term climate benefits of land-based mitigation strategies such as compost amendments, a virtuous cycle.

4: Patterns and controls on greenhouse gas emissions from composting organic waste

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4.1 Introduction

One of the important uncertainties in the greenhouse gas lifecycle of waste diverted to composted soil amendment for C sequestration is the emissions from the composting process. Diverting organic waste from landfills and manure slurries can result in large greenhouse gas

savings (ARB 2015; Bogner et al 2007; Owen and Silver 2014). The emissions from composting these diverted feedstocks is less well understood. Composting is the aerobic degradation of organic matter, mediated by a succession of microbial communities, and used purposely by farmers for centuries to manage their wastes and recycle nutrients into their soil (King 1911). Under ideal, aerobic conditions, organic waste is converted to humified material; microbial respiration is dominated by carbon dioxide (CO_2) and N mineralization predominantly produces ammonium (NH_4^+). Compost piles often have pockets of anaerobiosis, however, and even well-managed compost production will release some methane (CH_4) and nitrous oxide (N_2O) (Bogner et al 2007), powerful greenhouse gases that have heat trapping potentials 34 and 298 times greater than CO_2, respectively, over a 100-year time frame (Myre et al 2013).

The composting process follows four phases; each is distinct in temperature, microbial communities, and state of feedstock. In the first *mesophilic* (25-40°C) stage, primary decomposers break down easily degradable material, releasing CO_2 and heat, causing the compost temperature to rise. Relatively high redox potentials allow for the presence of nitrifying and denitrifying bacteria, so N_2O emissions can occur during this phase (Hellman 1997). As the temperature climbs to 35°C, the *thermophilic* (35-65°C) stage begins, with thermophilic organisms gaining competitive advantage in the degradation of organic material. The high temperatures, if maintained for a sufficient duration, destroy weed seeds and pathogens. High temperatures and microbial activity can favor oxygen (O_2) consumption and methanogenesis (Hellman 1997). The third stage is *cooling* (second mesophilic): the activity of thermophilic organisms slows down when they begin to run out of easily degradable material, and the compost temperature decreases. Mesophilic organisms proliferate again, with an abundance of organisms that can degrade starch and cellulose. The final stage is *maturation,* during which bacterial numbers decline and fungal populations increase as easily degraded material is exhausted, and more recalcitrant material (e.g., lignins and other complex C molecules) dominates. Temperature gradually declines to ambient temperature (Diaz 2007).

Few studies have measured greenhouse gas emissions from organic waste composting (Zhu-Barker et al. 2017; see also Sommer 2004, Wagner-Riddle 2006; both looked at manure only) or aimed to understand the drivers of greenhouse gas emissions from a full-scale process, which is critical to more effectively manage composting for climate change mitigation. One reason that emissions from composting are not well characterized is that the methods require interdisciplinary approaches and technologically advanced instrumentation due to the scale and heterogeneity of compost piles. Current methods used to measure greenhouse gases from composting can be divided into micrometerological approaches including eddy covariance and mass balance measurements (Harper et al 2011), and non-micrometerological techniques including static chamber measurements. Chamber techniques, however, operate at a scale much smaller than a compost pile (< 1 m) and introduce a number of sources of bias, including changing pressure and concentration gradients, physically disturbing the source, leakage, an inability to capture spatial and temporal variation, and the potential for significant human error (Kent 2010, Chapuis-Lardy 2007, Wagner-Riddle 1996). Measuring emissions from an enclosed space (e.g., Amlinger 2008) is easier methodologically, but introduces artifacts from enclosing organic feedstocks under conditions that alter degradation dynamics.

In contrast, micrometeorological approaches use atmospheric physics and engineering principles to measure ambient gas concentrations and relate these to fluxes from a source. These methods allow researchers to study systems in their natural state over a range of spatial scales. A variety

of micrometeorological approaches exist for different spatial scales, but the two approaches most appropriate for use at the compost-scale are mass balance and backward Lagrangian stochastic (bLs) methods (Harper et al 2011). Mass balance techniques measure gas concentrations going into and coming out of a control volume surrounding a source of interest (e.g., a compost pile). The emission rate from the source is calculated by subtracting the output flux from the input flux. bLs is a modeling approach that calculates atmospheric transport, most appropriate for calculating the relationship between concentration and fluxes for ground level sources. Appropriate sensor placement is key to effective modeling (Harper et al 2011).

Amlinger et al. (2008), Sommer et al. (2004), and Kent (MS Thesis, 2010) all used micrometeorological approaches to estimate emissions from a source, and provided a basis upon which the methodologies may be improved. Amlinger et al. (2008) used an open dynamic emission chamber – essentially a tent covering an entire compost windrow – and measured gas concentrations at the inflows and outflows. Though this approach effectively captured emissions, the covering of the pile itself may have altered its behavior by increasing temperature and moisture, and altering the concentration gradient from the source to the atmosphere. Sommer et al. (2004) and Kent (2010) both used a mass balance flux approach. In Sommer et al. (2004), researchers mounted instrumentation on a weather vane to measure actual upstream and downstream concentrations of gases from a circular manure pile, while Kent (2010) measured emissions from a compost pile using wind towers and gas measurement towers to calculate gas fluxes. Kent (2010) was the first to use micrometeorological approaches to measure trace gas emissions from composting, but the study encountered several equipment malfunctions, was short in duration (5 weeks), used only green waste as a feedstock, and the experiment was not replicated.

Understanding the controls on greenhouse gas fluxes is critical for modeling and extrapolation of results. This in turn will support the development of successful policy actions to reduce emissions in California. The composting process is dynamic, varying over time and space. Though it is mostly an aerobic process, the heterogeneity of the feedstock, temperature and moisture, rates of microbial activity, and structural properties of the pile can create variable redox conditions. In the absence of oxygen, a succession of microbes will convert carbohydrates in the organic waste to CO_2 and CH_4 (Bridgam and Richardson 1991). Once CH_4 is produced, it may be emitted to the atmosphere, or oxidized to CO2 within the pile.

Oxygen availability is a key control over CH_4 production and oxidation, and O_2 availability is positively correlated with porosity of the feedstock (pore space in the media; Luo et al 2014, Amlinger et al. 2008) and turning frequency (Hellman 1997), and negatively correlated with the size of the pile. Methane oxidation rates vary as a function of CH_4 production, O_2, water, DOC, pH, temperature (King 1997; Teh et al. 2005, Teh et al. 2008, Teh and Silver 2006, Sullivan et al. 2014, McNicol and Silver 2015), presence of a cover material (biofiltration; Luo et al 2014), and feedstock (Sonoki et al. 2014, Luo et al. 2014), and can be inhibited by higher concentrations of NH_4 (King et al 1997, although see Gulledge and Schimel 1998). Rates of CH_4 oxidation are limited by diffusive transport, which is constrained by moisture content and the shape and size of the pile (King 1997). Addition of biochar to compost piles may enhance CH_4 oxidation by enhancing aeration and gas diffusion, supporting the growth of methanotrophs (Sonoki et al 2014). The balance between CH_4 production and oxidation is likely to be controlled by redox potential (Teh et al. 2008; Teh et al 2005), and both processes are affected by temperature and

moisture, which control O_2 solubility and the rates of biological activity (Treat et al 2014; Olefeldt et al 2013).

Nitrification, the conversion of NH_4^+ to nitrate (NO_3^-), and denitrification, the conversion of NO_3^- to nitrogen gas (N_2 and N_2O), are the major pathways that lead to N_2O production and consumption. The balance between production and consumption of N_2O is dependent on the controls on both processes. Nitrification is largely regulated by NH_4^+ supply, pH, and redox potential; denitrification is dependent on NO_3^- supply, C availability, and redox. Generally, N_2O fluxes are expected to increase as the C:N ratio of organic matter decreases, as the N availability increases (Klemedtsson et al. 2005), and as O_2 concentrations decline (Firestone and Davidson 1989). Kelemedtsson et al (2005) suggested that a C:N ratio of 25 is a threshold level: at higher ratios, N_2O emissions are negligible, and at lower ratios, emissions increase strongly. Nitrous oxide tends to be emitted from microsites of organic substrate where an O_2 gradient occurs, and where concentrations are lower than ambient. For O_2 concentrations between 5-21%, reduction of NO_2^- by nitrifiers (nitrifier denitrification) is generally thought to dominate; at lower concentrations (< 5%), reduction of N_2O to N_2 is generally more favorable (Chapuis-Lardy et al. 2007). A review of emissions from organic waste management strategies (Pardo et al 2015) concluded that pile turning and addition of bulking agents increased aeration and decreased emissions of both N_2O and CH_4; the turning improved aeration and homogenization, and prevented stratification, and thus the oxygen gradients that favor N_2O emission (Pardo et al 2015). Nitrous oxide consumption in soils was associated with low availability of mineral N, and higher residence time of N_2O gas – the longer N_2O remained, the more likely it was to be consumed (Chapuis-Lardy et al 2007).

This study aimed to measure greenhouse gas emissions from the composting of manure and food waste. The specific objectives were:

- To measure real-time, whole pile emissions of CO_2, CH_4, and N_2O from composting green waste and manure.
- To determine biogeochemical controls on greenhouse gas emissions from different feedstocks.

4.2 Methodology

4.2.1 Site Description and Experimental Set-up

We conducted two 100-day composting experiments at the West Marin Composting Facility in Nicasio, California, from February through September of 2016. At the start of each experiment, we formed a windrow pile (15 m long, 2 m tall, 4 m wide) with a mix of locally available manure and green waste; the first experiment used cattle manure, goat manure and yard waste; the second pile included these as well as horse and chicken manure. Each pile was managed as commercially-produced compost piles: with weekly turning events, periodic watering when the compost moisture dropped below 40%, and a composting duration of approximately 100 days. Results for the two piles were similar and thus we present figures for one pile only. The food-waste compost pile was delayed due to permitting issues. These data will be available in 2018.

4.2.2 Micrometeorological Mass Balance Method

Each windrow pile was outfitted with a greenhouse gas and wind measurement system. Four wind towers (Figure 4.1, A-D) were placed along cardinal directions along each edge of the

windrow, 1 m from the edge of the pile; each of these towers held four gas intakes, at 0.25 m, 1 m, 2 m, and 3.5 m above the ground. Air samples were drawn through 16 Teflon tubes, delivering gas samples (1 intake per minute) to a G2308 cavity ring down laser spectrometer (Picarro, Santa Clara, CA), which measured real-time CO_2, N_2O and CH_4 concentrations. While one intake port delivered gases to the greenhouse gas analyzer, the other 15 lines were flushed with ambient air.

Two wind towers, placed on the NW and SE corners of the windrow, held four 3D sonic anemometers each, at four different heights (same as gas intakes) above ground, and measured wind speed and direction continuously (at 0.1 Hz) throughout the experiment.

The high frequency wind and gas concentration data were combined to measure greenhouse gas concentrations upwind and downwind of the pile, and to calculate the flux of CO_2, CH_4 and N_2O from the composting process, using the micrometeorological mass balance method (Denmead 1998, Wagner-Riddle 2006). The flux from a source area can be approximated by:

$$\text{Flux} = \frac{1}{L}\int_0^{\infty} \bar{u}_z \left(\bar{c}_{z,d} - \bar{c}_{z,u}\right) dz \quad (1)$$

Where L (m) is the fetch, or the horizontal distance that air travels over the source, $\bar{u}_z$ represents mean horizontal wind speed at height z (m/s), and $\bar{c}_{z,d}$ and $\bar{c}_{z,u}$ are the gas concentrations at height z downwind and upwind of the emitting source, respectively.

4.2.3 Sensor System and Laboratory Analyses

Throughout the composting experiments, we measured conditions inside the pile with a system of 27 automated sensors. Nine O_2, temperature, and moisture sensors each were placed in three transects along the length of the pile, at three heights (0.50 m, 1 m, 1.5 m), in the center of the pile (Figure 4.2). These sensors were removed briefly for pile turning (< 60 minutes), once a week.

Each week, we collected compost grab samples into 1 gallon Ziploc bags from the three heights corresponding to sensor locations (0.5 m, 1 m, 1.5 m – and 2 m horizontal depth) before and after pile turning (total n = 6). Samples were analyzed for gravimetric moisture content by weighing 10 g samples before and after drying at 105° C for 24 hours. Compost pH was measured by suspending 3 g of sample in 12 g of water (McLean 1982). We measured porosity by filling a pre-marked mason jar with 100 mL of compost, weighing it, and adding distilled water to the 100 mL line, and weighing again to determine the volume of pore space in the original sample (adapted and simplified from Danielson and Sutherland 1986; 5 replicates per sample). Potential N mineralization and nitrification was determined using dark laboratory incubations of 3-4 g compost samples and extracting compost subsamples before and after incubation (7 days) in 75 mL of 2M KCl (Hart et al. 1994; 3 replicates per sample). Total C and N of the compost was measured by air drying the samples, sieving through a 2 mm sieve, removing all roots, grinding, and loading subsamples into small tins for combustion using an elemental analyzer.

4.3 Results and Discussion

4.3.1 Greenhouse Gas Emissions

Methane fluxes varied dramatically over time, from a minimum of 8.2e-7 g m^{-2} s^{-1} to a maximum of 1.7e-2 g m^{-2} s^{-1} (roughly five orders of magnitude), with relatively short periods of time accounting for the majority of emissions. The period of highest emissions occurred at the very start of the pile, and a second "hot period" occurred about three-quarters of the way through the experiment. Most CH_4 – 60% of total emissions -- was emitted during the first fifth (20 days) of the composting process. Sixteen percent of CH_4 emissions were released between days 50 and 75 (1/4 of the experiment duration). The top ten "hottest moments" occurred on days 4-6 (4 of 10), 9-11 (5 of 10), and day 70 (1 of 10). The top fifty hottest moments were clustered into two periods: day 0-20, and day 60-75. While most fluxes measured were small and positive, the distribution of fluxes formed a long tail, with few high fluxes responsible for most CH_4 emissions. The distribution also showed a short tail in the negative direction; we measured few, negative fluxes of small magnitude.

We found a total emission factor of 0.6 g CH_4 kg^{-1} for the first composting experiment. Integrating under the flux curve, we found a total emission factor for CH_4 to be 0.7 g CH_4 kg^{-1} feedstock for the second experiment. Nitrous oxide emissions from the composting waste did not exceed the detection limit of our instrument (25 ppb + 0.05%, or 4.5E-5 g m^{-2}) for the duration of the experiment.

These emission factors bracket the median of emissions factors for windrow composting found in the literature (0.69 g kg^{-1}). We found 55 reported emissions factors from composting, from 16 published studies (Cuhl 2015, Gonauer 1997, Puyuelo 2014, Amglinger 2008, San Joaquin Valley Air PCD 2013, Netherland Enterprise Agency 2010, ARB 2016, Inamori 2001, Colon 2012, Amon 2001, Hellman 1995, South Coast Air Quality Management District, Zhu-Baker 2017, Hellebrand 1996, Adhikari 2013, Anderson 2010). Though we would have expected our experiments to fall on the upper end of these estimates due to the fact that our study used continuous, whole pile measurements, and composted a higher emitting source (manure) than many, this was not the case. That the emission factor was in line with previous estimates suggests that pile management may play a larger role than the feedstock itself in predicting greenhouse gas fluxes from composting.

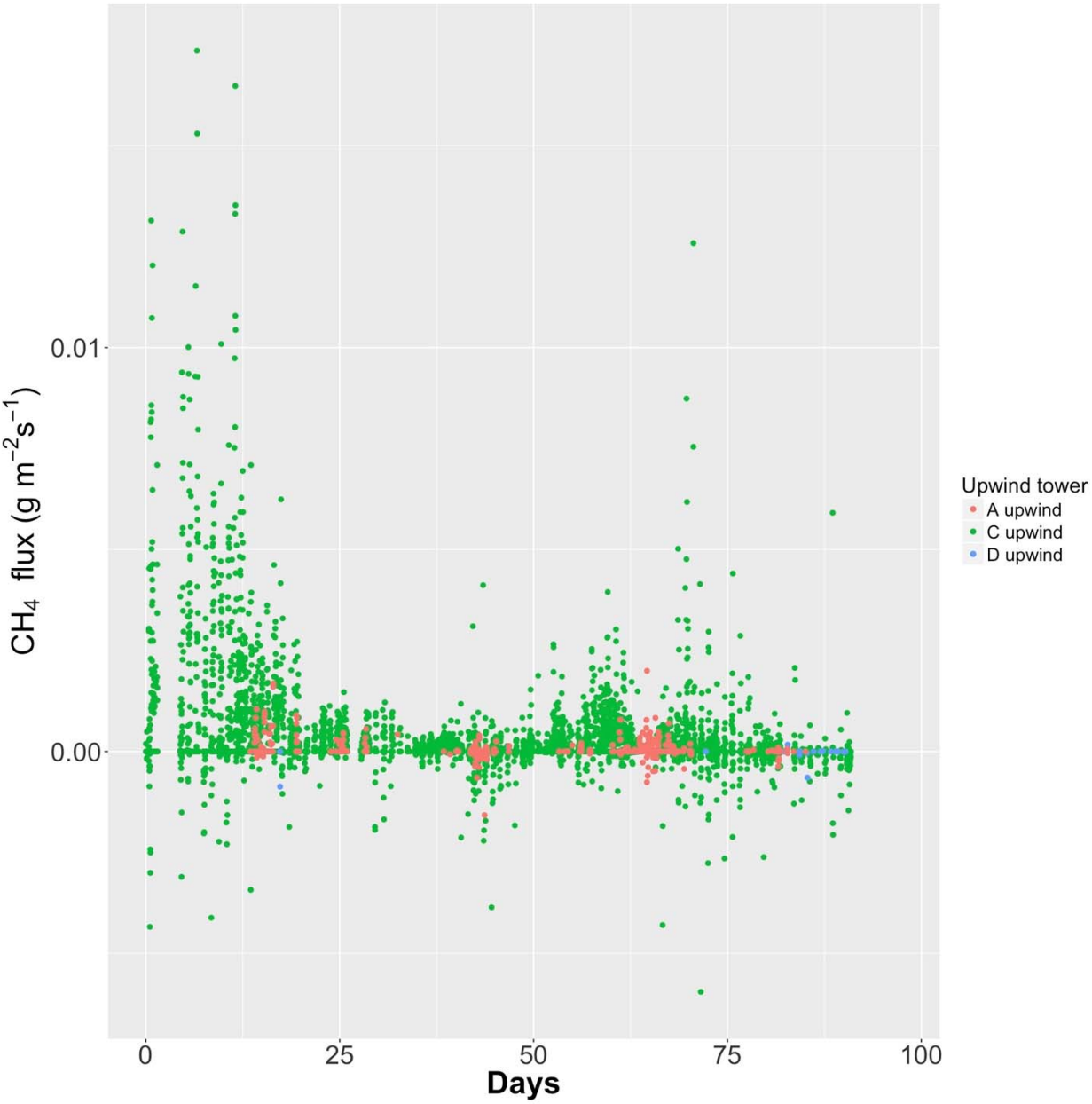


Figure 4.1: Methane fluxes from composting green waste and manure in Marin County, CA. (Pile 2). Watering events occurred on days 29,50,57,64,71, and 78.

Carbon dioxide emissions range over four orders of magnitude, with positive values from 2e-5 to 1e-1 g m^{-2} s^{-1}, and are an indicator of the lability (or accessibility) of C in the composting pile. Similar to CH_4 emissions, relatively short periods accounted for the majority of CO_2 emissions, and there were two periods of high emissions: in the first 12 days of the composting process, and near the end (near day 70).

Integrating under the flux curve, we found a total CO_2 emission factor to be 10 g CO_2 kg^{-1} feedstock for the first experiment, and 7 g CO_2 kg^{-1} feedstock for the second. Because CO_2 emissions from biological systems are considered to rapidly cycle, these biogenic emissions are usually considered to have no global warming potential (Barton et al 2008, Christensen et al

2009, Gentil et al 2009, Rabl et al 2007, Vergara et al 2011). Further, from a greenhouse gas mitigation perspective, it is preferable that the carbon emitted from composting be emitted as CO_2 rather than as CH_4.

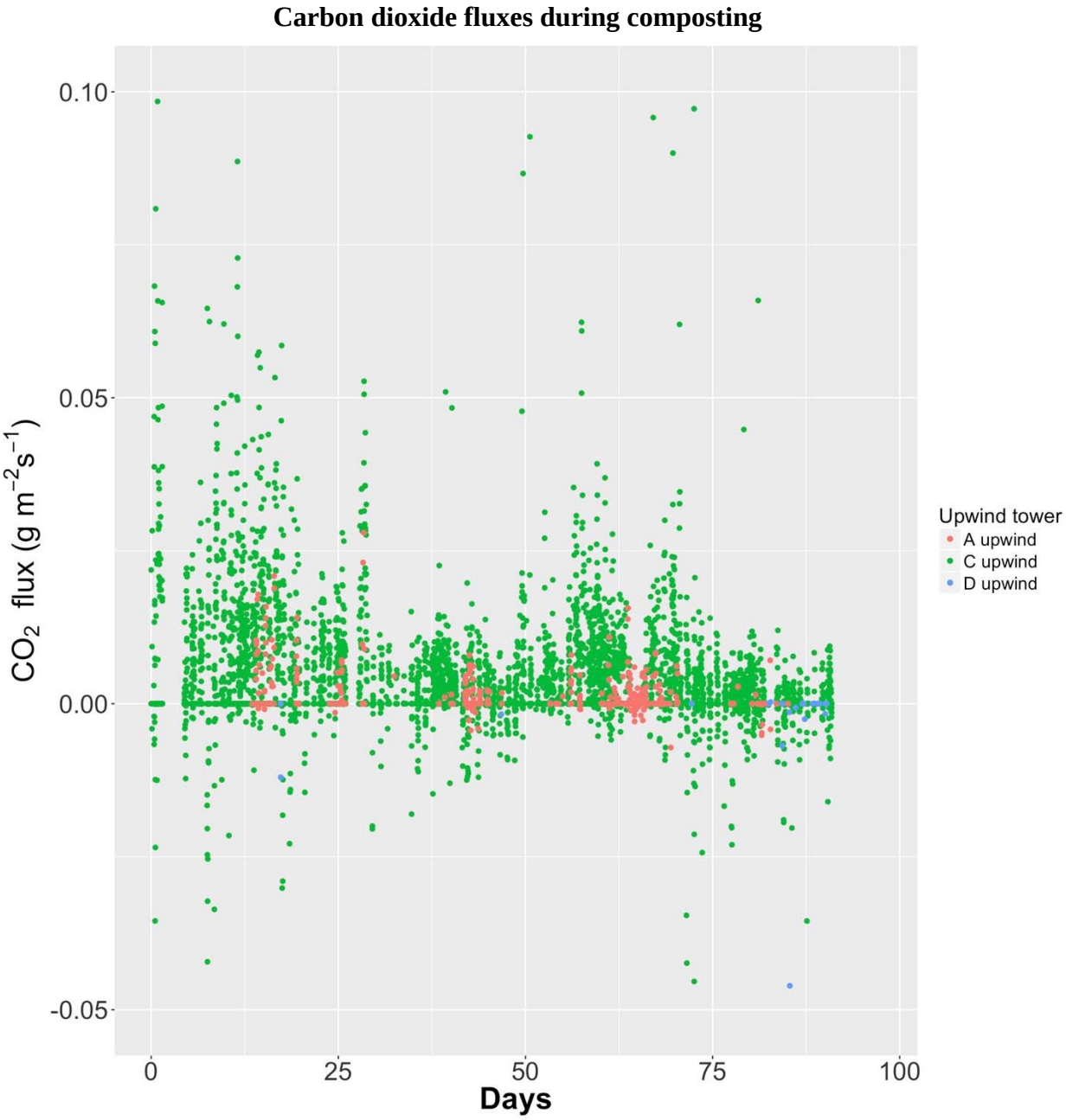


Figure 4.2: Carbon dioxide fluxes from composting green and manure waste in Marin County CA. (Pile 2).

4.3.2 Environmental Conditions: Oxygen, Temperature, Moisture

Oxygen concentrations in both piles followed an inverted "U" shape, with the lowest mean values occurring near the beginning (day 0-1) and near the end of the experiment (day 65-90), and the highest mean values occurring in the middle (Figure 4.3). The lowest instantaneous

mean value was 0.07 ± 0.034% and occurred at the top position on day 0; the lowest daily mean values were 1.6 ± 0.062% (day 65), 2.3 ± 0.086% (day 87), and 0.95 ± 0.37% (day 0) for the bottom, middle and top positions, respectively. The highest O_2 values occurred two to five weeks after degradation began; these high values were 20 ± 0.072% (day 14), 17 ± 0.47% (day 26), and 15 ± 2.4% (day 34), for the bottom, middle, and top O_2 sensors, respectively. The highest daily means occurred on day 19 in the bottom of the pile (15 ± 0.29%), 48 in the middle (15 ± 0.086%), and 34 in the top (14 ± 0.087%).

Oxygen concentrations

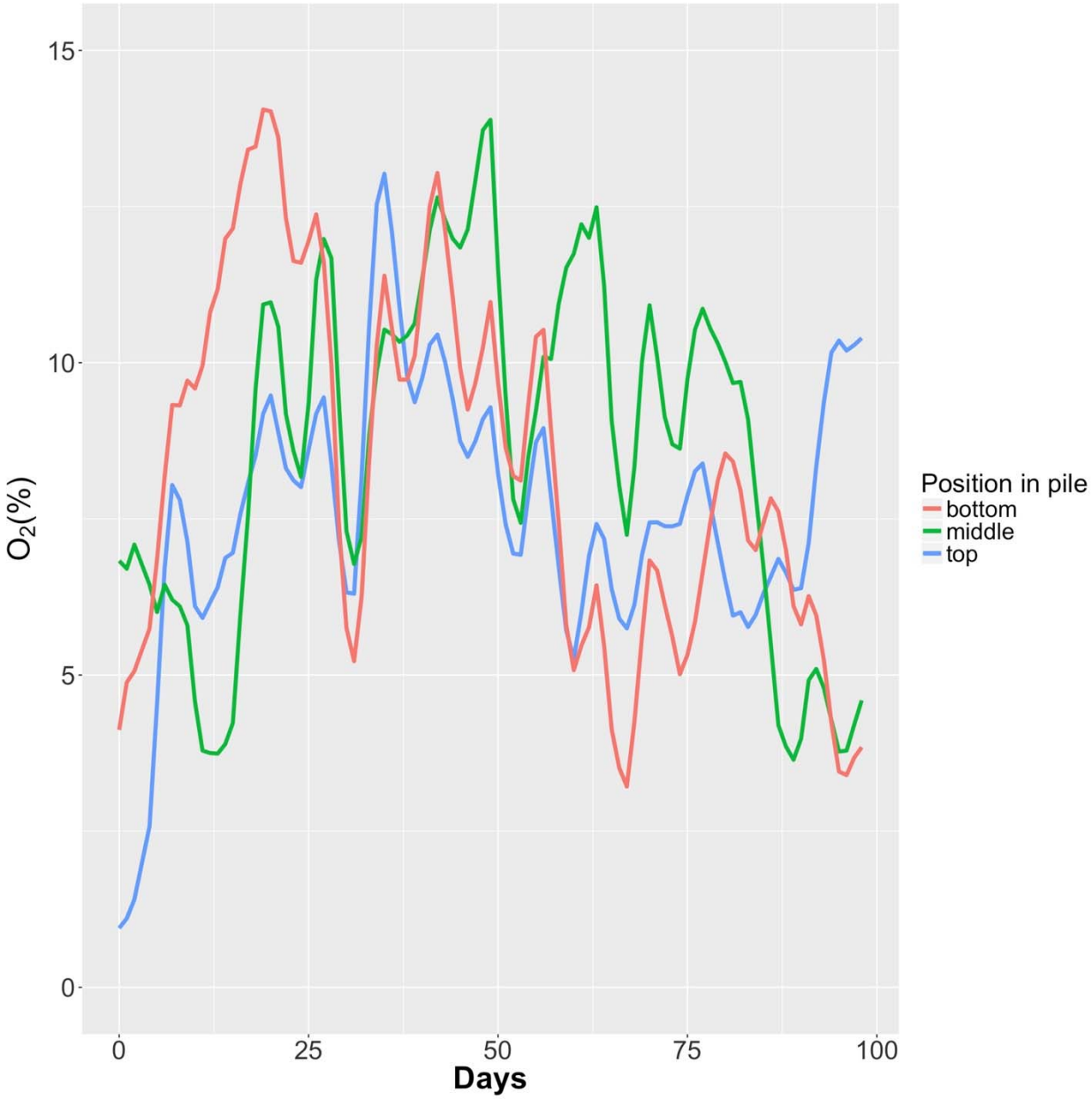


Figure 4.3: Oxygen concentrations within the composting pile show an inverted "U" shape over time. Pile 2.

Temperature varied over time and by position in the composting pile, but followed a predictable pattern: Rising rapidly initially, then slowly decreasing (Figure 4.4). The lowest temperatures recorded occurred at the start of the composting process, on day 0: 34 ± 0.39 °C, 34 ± 0.43 °C, and 35 ± 2.5 °C for the bottom, middle and top sensors. The highest temperatures were reached after a week of decomposition; the bottom, middle and top sensors reached 72 ± 1.6 °C, 73 ± 2.9 °C, and 76 ± 1.3 °C, on day 15, 7.9, and 7.9, respectively. By the end of the experiment (day 98), the pile was still warmer than ambient conditions (57 ± 0.86 °C , 53 ± 5.1 °C, and 61 ± 3.3 °C for the bottom, middle and top sensors, respectively).

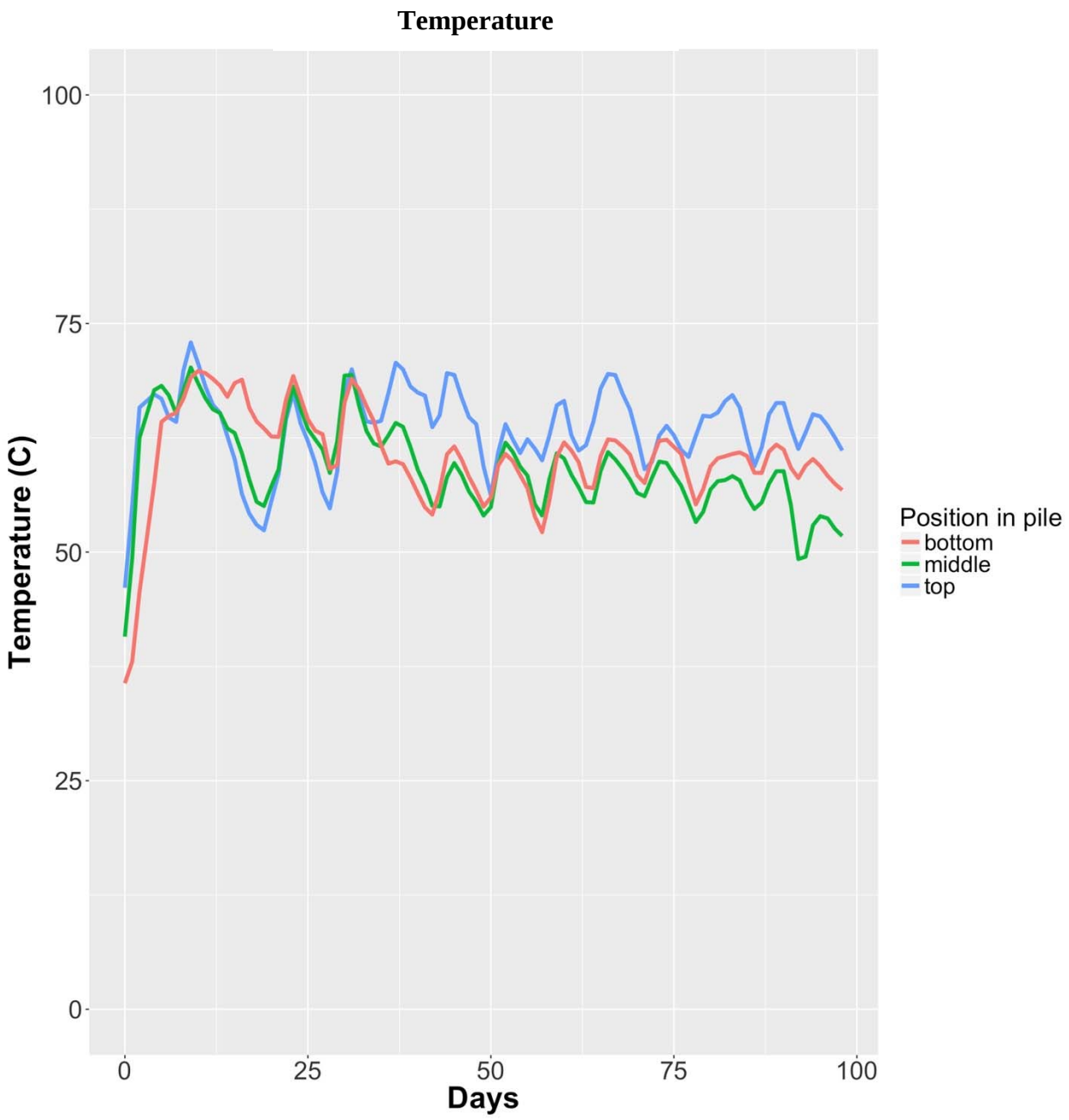


Figure 4.4: Temperature readings from the composting pile over time show a quick rise, and then leveling off, over time. Pile 2.

Moisture levels in the composting pile were variable, due to water consumption and evaporation during the composting process, as well as water inputs to keep conditions favorable for decomposition (Figure 4.5). Moisture generally declined in the pile over the first 50 days, and then increased again (with watering) over the final 50 days. The moisture in the three pile positions tracked each other well throughout the experiment, and all positions show sharp discontinuities with the weekly pile turning. The highest moisture levels at each position

approached 60%; 0.56 ± 0.01% (day 4), 0.57 ± 0.00% (day 64), and 0.56± 0.01% (day 90) were the highest recorded levels in the bottom, middle, and top of the pile, respectively. The lowest moisture reached in the pile dipped below 40% in all positions: 0.28 ± 0.01 (day 7), 0.36 ± 0.01 (day 42), and 0.39 ± 0.01 (day 28), for the bottom, middle, and top sensors, respectively.

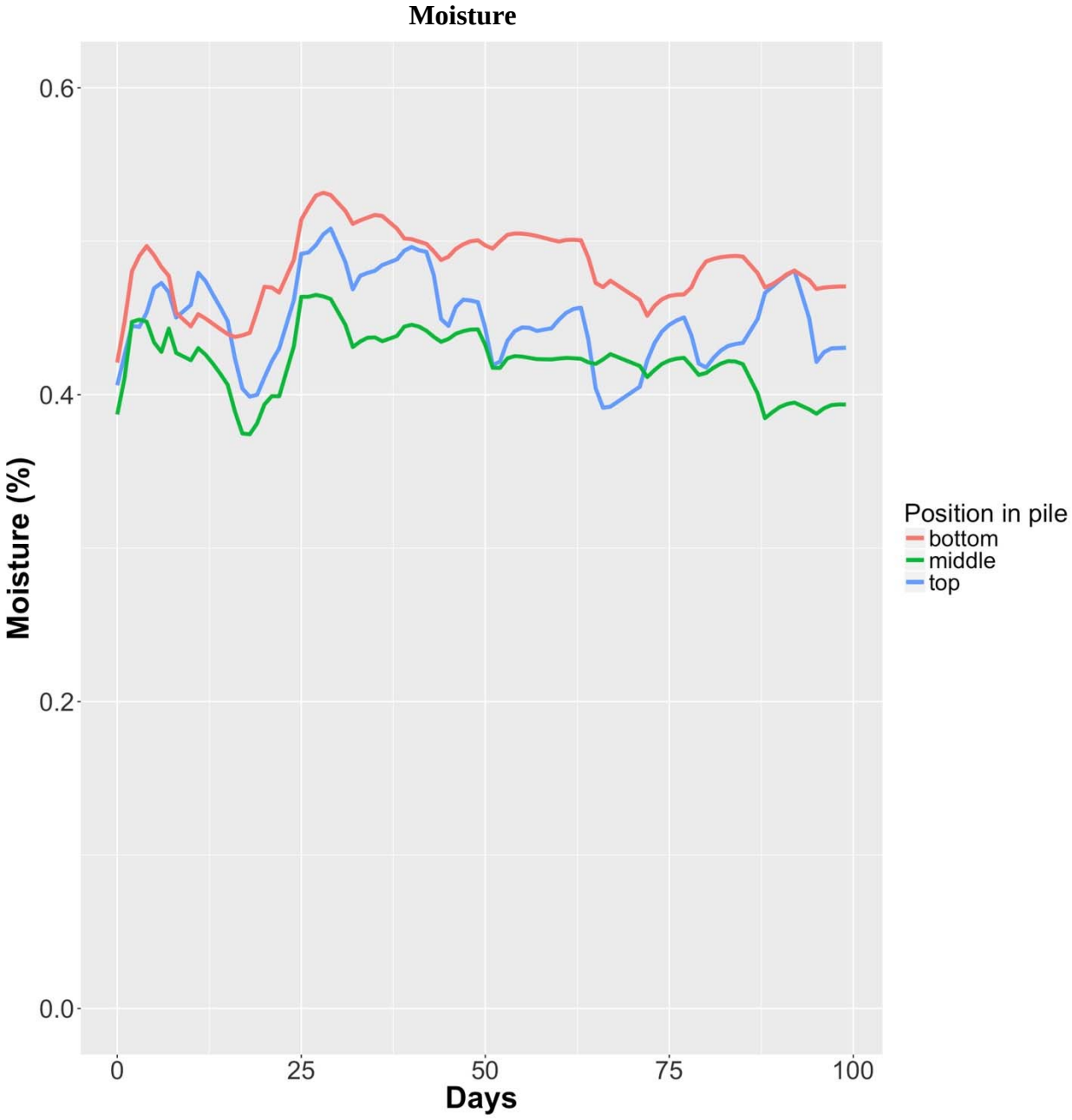
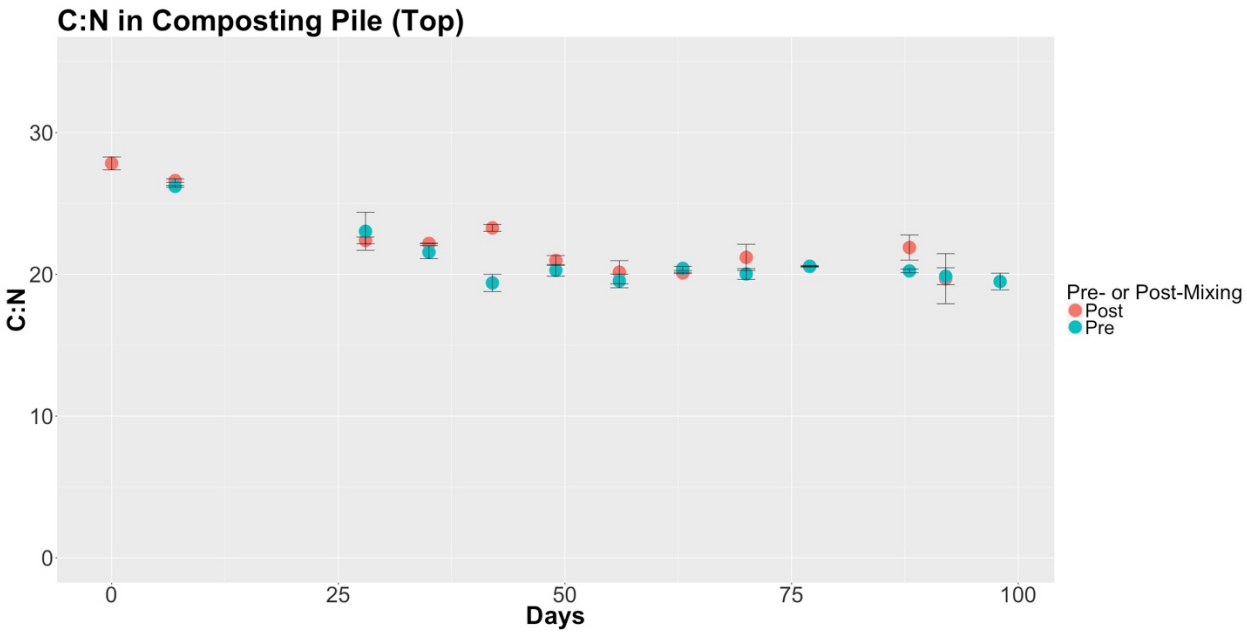


Figure 4.5: Moisture levels in the composting pile over time were variable, but were also adjusted to keep from dropping below 40%. Pile 1.

4.3.3 Laboratory Analyses: pH, C, N

Compost pH remained relatively stable throughout the composting process. Pooling top, middle and bottom together, pH started at 9.0 ± 0.06 on day 0, reached a high value of 9.3 ± 0.01 on day 56, a low value of 7.58 ± 0.07 on day 49, and a final pH value of 9.1 ± 0.11 at the end of the experiment. The C content of the compost declined modestly (pre-mix: slope = -0.058, $p<0.0001$; post-mix: slope = -0.057, $P<0.0001$), the N content remained stable (pre-mix slope not statistically different from 0; post-mix: slope = 0.001, $P<0.01$), and the C:N ratio decreased over the course of the composting process (pre-mix: slope = -0.052, $P<0.0001$; post-mix: slope = -0.064, $p<0.0001$). For the bottom, middle, and top samples of compost, the C concentration declined from above 30% (33.4 ± 0.22%; 32.2 ± 0.08%; 34.5 ± 0.28%) on day 1 to around 30% (29.8 ± 0.52;% 30.1 ± 0.30%; 29.4 ± 0.31%) on day 98 (Figure 4.6).



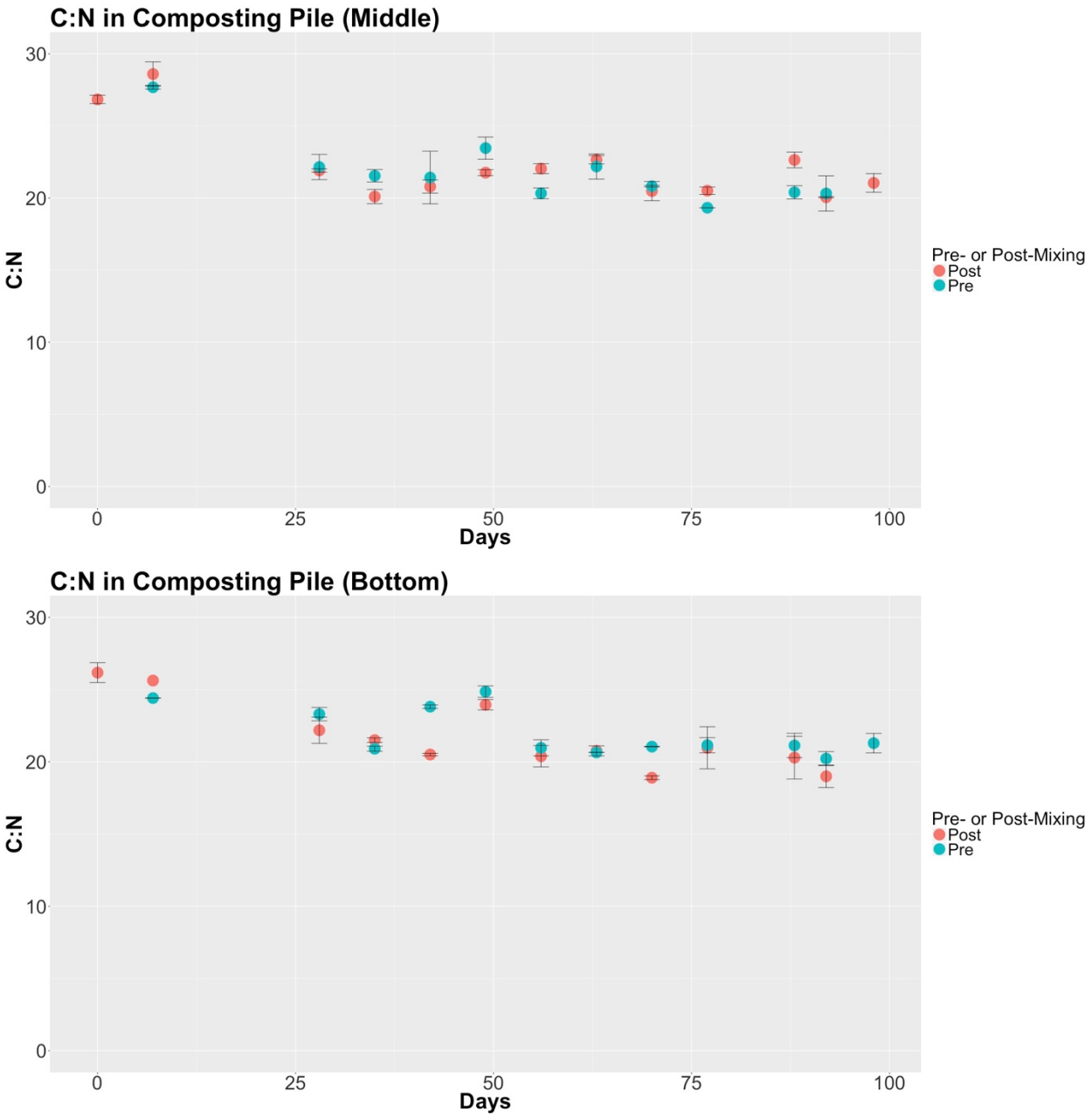


Figure 4.6: Carbon to nitrogen ratio of the composted feedstock, analyzed weekly. Blue dots show pre-mixing samples, and red dots show post-mixing samples. Bars show standard errors. Pile 2.

The N in the compost remained relatively flat, starting at 1.28 ± 0.02%, 1.39 ± 0.03% and 1.24 ± 0.01% in the bottom, middle and top of the pile, and ending at 1.40 ± 0.02%, 1.43 ± 0.03%, and 1.32 ± 0.05%. The C:N ratio at the start of the experiment was 26.2 ± 0.69, 22.1 ± 0.88, and 27.8 ± 0.45, in the bottom, middle and top of the pile, respectively, and was measured to be 21.3 ± 0.67, 21.0 ± 0.65, and 21.9 ± 0.89 at the end of the experiment. Similar results were found for the first experimental pile.

4.3.4 Controls on Emissions

No statistically significant linear or polynomial predictive relationships among O_2, moisture, temperature and CH_4 fluxes from composting were found to exist. The environmental variables instead exhibited threshold effects on CH_4 emissions, with most CH_4 emissions occurring at moisture levels above 40%, temperatures above 60 ^oC, and O_2 below 5%. Though no simple mathematical model emerged from this work to predict CH_4 emissions from composting, the threshold effects observed support the broad literature on temperature, moisture, and O_2 as key controls on methanogenesis. Because moisture levels should exceed 40% to promote microbial activity, the central management implication of the environmental controls is that O_2 levels should remain above 5% in composting piles to inhibit methanogenesis. This can be done through active or passive aeration, or through the addition of bulking agents to the feedstock to increase porosity.

4.4 Summary

Two 100-day experiments yielded the first whole-pile, continuous measurement of greenhouse gas emissions from windrow composting. The total CH_4 emission factor from the two experiments were estimated to be 0.6 g CH_4 kg^{-1} feedstock and 0.7 g CH_4 kg^{-1} feedstock, respectively. These emissions factors bracket the median emission factor found in the literature, suggesting that management may play a larger role in predicting greenhouse gas fluxes than does the feedstock. While there were no linear relationships between CH_4 emissions and other chemical or physical characteristics measured, we found that CH_4 emissions were greatest under warm wet conditions when O_2 concentrations dropped below 5%. The total CO_2 emission factors, generally not considered to have a global warming potential due to its biogenic origin, were estimated to be 10 g kg^{-1} feedstock, and 7 g kg^{-1} feedstock, for the two experiments. Nitrous oxide concentrations were below the instrument detection limit (25 ppb + 0.05%, or 4.5E-5 g m^{-2}) throughout the experiments. Our results highlight the potential to manage composting for low emissions. Low emissions from composting contributes to the overall climate change mitigation benefit of diverting high-emitting waste streams to compost and subsequent land application for C sequestration.

5: The greenhouse gas benefits of composting relative to alternative feedstock fates: a lifecycle assessment

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5.1 Introduction

Lifecycle assessment (LCA) is a leading modeling tool for evaluating the greenhouse gas costs and benefits of alternative waste management practices for California. Composting waste products achieves four waste management objectives: to reduce the total volume of waste to be treated, to stabilize organic matter, to destroy weeds and pathogens, and to produce a valuable product (McDougall 2011, Diaz 2007). The environmental benefits of composting food waste and livestock manure depend on how these feedstocks are managed during the composting process, and on the alternative fates for the wastes (Vergara et al. 2011). Most food waste in California is landfilled (CalRecycle 2015), and most confined livestock manure is stored in anaerobic lagoons for weeks to months, prior to direct land application (Owen et al. 2014).

Waste management is an important contributor to greenhouse gas emissions in California, and alternative treatment methods have the potential to transform waste from a net source to a net sink of greenhouse gas emissions (Bogner et al. 2007, Vergara et al. 2011). Methane emissions from decomposing organic waste in landfills are the biggest contributor to greenhouse gas emissions from solid waste in California and worldwide (ARB 2015; Bogner et al 2007). Municipal waste management emits 2% of California's greenhouse gas emissions, and 94% of these emissions come from landfills. Eight percent of California's greenhouse gas emissions come from agriculture (ARB 2015), and livestock manure management is the greatest contributor (Owen et al. 2014). Municipal waste and livestock manure have the potential to be greenhouse gas sinks rather than sources with appropriate management intervention.

The lifecycle greenhouse gas impacts of composting waste include emissions from processing, transportation, treatment (all part of the "waste management" sector) and application to land (most often included under the "agricultural" sector). The greenhouse gas costs and benefits for transportation, land application, and alternative fates for municipal solid waste are well characterized, and studies show that it is the "treatment" phase of organic waste management (e.g., composting, digestion, etc.) that is the most greenhouse gas intensive (Vergara et al. 2011, Vergara et al. 2014). The baseline management for livestock manure is not well characterized, but the treatment phase – composting, digestion, slurry, lagoon – is also likely to be the most emission-intensive. The net climate benefits from the use of composted waste on rangelands, then, hinge on the emissions from the composting process itself, which are uncertain (DeLonge et al. 2013). Because of the greenhouse gas intensity of the business as usual approaches to waste management, composting is very likely to outperform (under-emit) the baseline case.

This lifecycle assessment focuses on the climate impacts and benefits of different organic waste management strategies. It does not directly consider other potential environmental impacts or benefits of compost or digestate application, including ecological impacts to grasslands (from excess fertilization, or contamination).

This study aimed to estimate the best use of California's organic feedstocks, from a carbon (C) perspective. The specific objective was to use lifecycle assessment modeling to critically evaluate the greenhouse gas benefits of composting relative to alternative feedstock fates for the state of California. Because organic wastes now present a burden to local communities as they aim to dispose of or treat this waste to minimize the negative impact on local air and water quality, this analysis explored what could be gained if these organic feedstocks were instead treated as a resource. We specifically analyzed the C benefits of using organic feedstocks as a resource, relative to their "business as usual" management.

5.2 Methodology

To quantify the potential state-wide greenhouse gas benefits of using composted food waste and livestock manure on rangelands, we used a peer-reviewed lifecycle assessment model (EASETECH) to estimate greenhouse gases emitted from the business as usual waste treatment scenario and compare these to emissions from alternative treatment scenarios for the organic feedstocks, including anaerobic digestion, and composting with land applying the finished compost. Lifecycle assessment is the leading methodology for assessing the environmental impact of a product or decision and comparing it to other uses or products (McDougall et al. 2001).

The study determined alternative fates for cattle manure, green waste, and food waste within the borders of California, and analyzed the environmental implications of the management of these 'wastes.' The system boundary for the analysis began at the moment that the feedstocks are discarded, included all collection, transport, and treatment, and ended when these feedstocks become an emission or a new product. (The production and generation of these wastes are outside the scope of the analysis). Figure 5.1 provide a schematic of the system boundary used.

A consequential (rather than attributional) approach was used to lifecycle modeling; this approach included the modeling of physical flows to and from systems, and also the consequences of those system actions in the broader world (Ekvall and Weidema 2004; Martin et al. 2015). For example, when modeling the energy production from anaerobic digestion, the displacement of the marginal source of electricity in California was used, commonly natural gas (Marnay et al 2002). The functional unit for the analysis was 1 tonne of organic waste (dairy manure, green waste, and food waste) over one year. The total quantity and composition of these wastes were determined from a variety of publicly available sources (CalRecycle 2016, CalRecycle 2015, CalRecycle 2014, Williams 2008, ARB 2014).

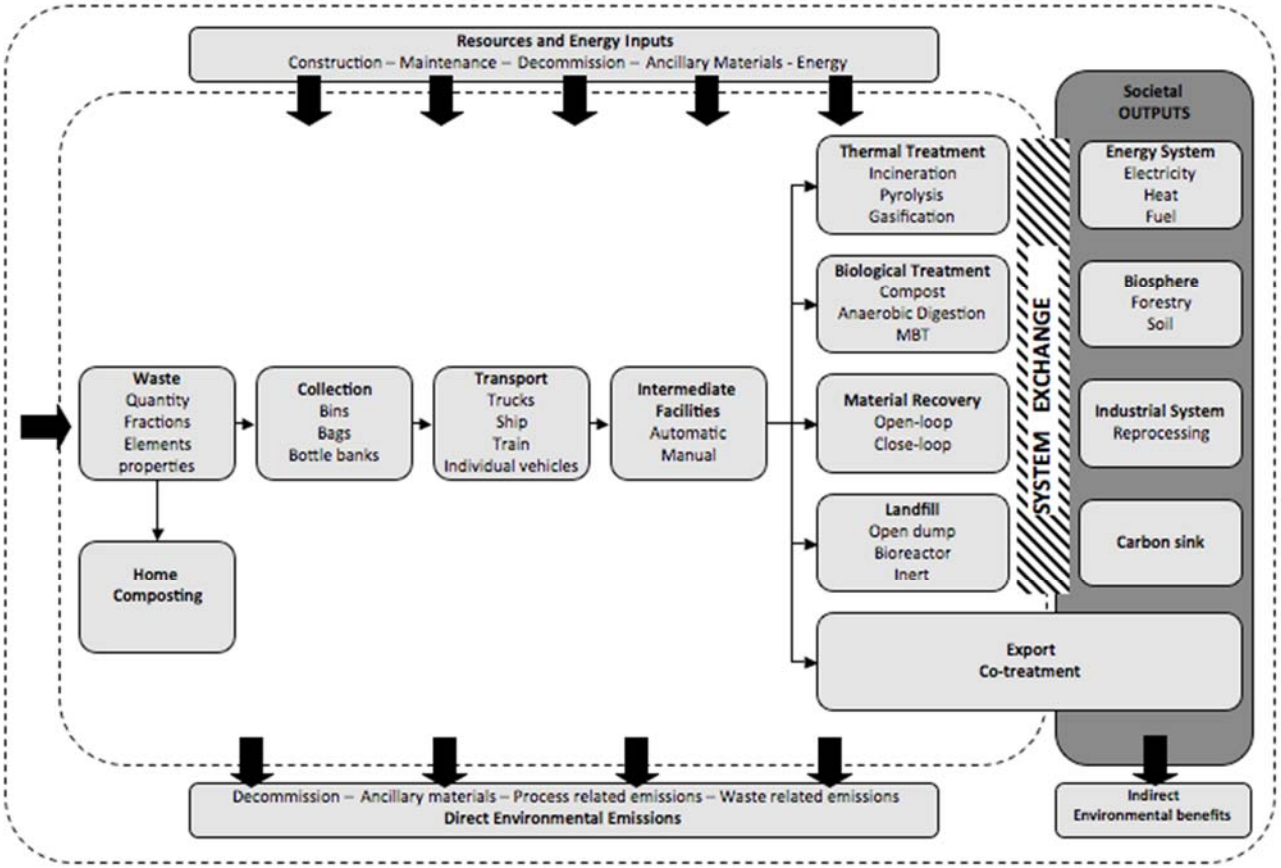

Figure 5.1: System boundary for waste management life cycle assessment (Vergara et al. 2011).

Data were collected to determine the total quantity and composition of the different organic waste streams, and how they were managed in the baseline and business as usual case (ARB 2016, CalRecycle 2016, CARB 2015, Owen et al 2014, Williams 2008; A summary of data sources can be found in Batjiak and Vergara unpublished data). A peer-reviewed lifecycle assessment model developed by the Technical University of Denmark, EASEWASTE (Clavreul et al. 2014) was used. The model allows the user to estimate every process, from generation to emission; there are no default values, as there are in other LCA models (e.g., WARM).

The three alternative scenarios were analyzed in terms of their net greenhouse gas emission (or sequestration), relative to the baseline scenario ('business as usual'). This study analyzed the net C implications of managing these organic feedstocks as follows:

(1) *Business as usual:* Food waste was collected and mostly landfilled. Roughly half of yard waste was composted and about half was used as alternate daily cover in a landfill. Cattle manure was mostly stored in anaerobic lagoons (EPA 2011; CARB 2015; Owen et al. 2014).

(2) *Composting & land application:* The study explored the re-routing of California's food and yard waste, and cattle manure to composting facilities, and the application of the resulting compost to rangelands.

(3) *Anaerobic Digestion*: The anaerobic digestion of all available food waste and cattle manure was modeled. The resulting methane was burned to produce electricity, and the solids were composted.

(4) *Incineration:* Yard and food waste were incinerated to produce electricity. Cattle manure was treated as usual (mostly stored in anaerobic lagoons).

The baseline scenario, against which all other scenarios were measured, was based on how these streams were currently being managed in California. At the time of this report, approximately 5.8 million tonnes of food waste were disposed in California each year, and the majority of this waste (88%) was landfilled (CalRecycle 2016). About 9.2 million tonnes of green waste (yard waste, trimmings) were disposed per year; about 44% was composted, and 40% was sent to landfills as alternate daily cover. Smaller fractions were used for mulch (7%) and used as fuel for electricity generation (8%) (CalRecycle 2016, Williams 2008). Approximately 5.7 million tonnes of cattle manure was produced yearly in California; most was stored in anaerobic lagoons (56%), but a sizeable fraction (20%) was directly applied to land as slurry. Smaller fractions were composted (12%) and land applied in solid form (11%) (Owen et al. 2014). For this analysis, the baseline case was simplified with model food waste being landfilled, yard waste being landfilled and composted, and cattle manure being stored in an anaerobic lagoon, and then spread as a liquid slurry on California grasslands.

Modeling of California's landfills was based on Themelis (2007), which found an average landfill gas collection rate of 64%. Emissions from composting and anaerobic digestion were taken from empirical studies (Komilis and Ham 2004, Komilis and Ham 1999, EPA 2003, Andersen et al. 2010, Boldrin et al. 2011), and methane emission factors from manure storage in anaerobic lagoons were calculated from Leytam et al. (2011) and Owen and Silver (2015). Nitrogen losses were taken from IPCC (2006) and Hamelin et al. (2013), and Dämmgen and Hutchings (2008). Emissions from the application of manure slurries to grasslands were taken from Amon et al. (2006) and Chadwick et al (2011).

Carbon sequestration from application of compost and digestate were taken from Ryals et al. (2013) and Yoshida (2018), respectively. Distances traveled for food and yard waste are taken from Vergara et al. (2011) – food and yard waste travel 16 km to the nearest composting or digester facility, but 50 km to the nearest landfill. We assumed that the resulting soil amendments from composting and digestion were used locally.

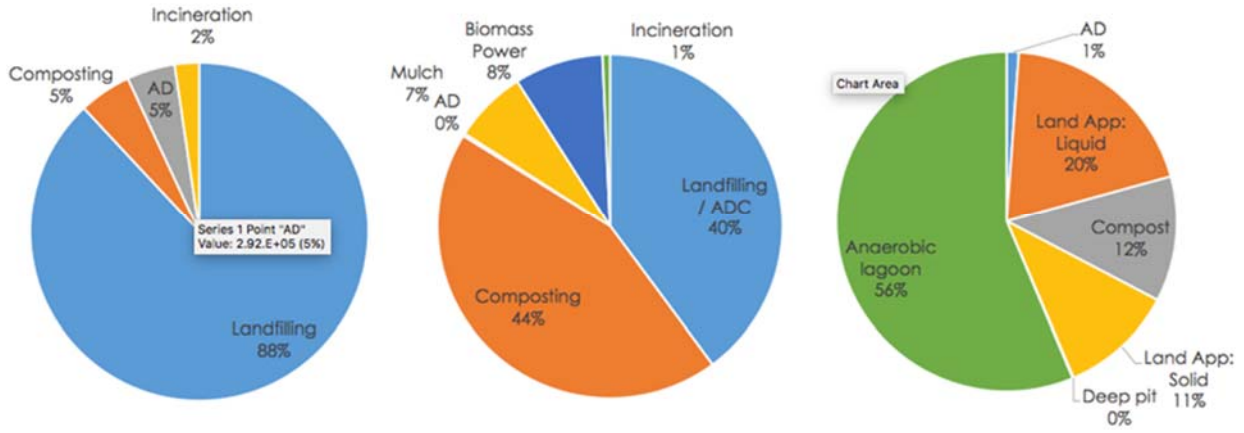

Figure 5.2: Baseline management for three streams of California organic waste. a) Food waste (left), b) Green/yard waste (center), and c) Cattle manure (right).

For the composting scenario (Scenario 2), we routed all the food and yard waste and manure to composting facilities, where the feedstock was composted in windrows, and the soil amendment was applied to California grasslands (as above). For the anaerobic digestion scenario (Scenario 3), all food and yard wastes and manure were digested, and the resulting solids were composted and applied to California grasslands. For the incineration scenario, food and yard waste were burned in modern incineration facilities to produce electricity, and manure was managed as in the baseline case (stored in an anaerobic lagoon, then applied as a liquid slurry to grasslands).

5.3 Results and Discussion

Of all scenarios considered for the management of California's organic waste, the baseline scenario – what is currently being done – performed the poorest in terms of net greenhouse gas emissions. All other scenarios outperformed the baseline case, by emitting fewer greenhouse gas emissions on net (Figure 5.3).

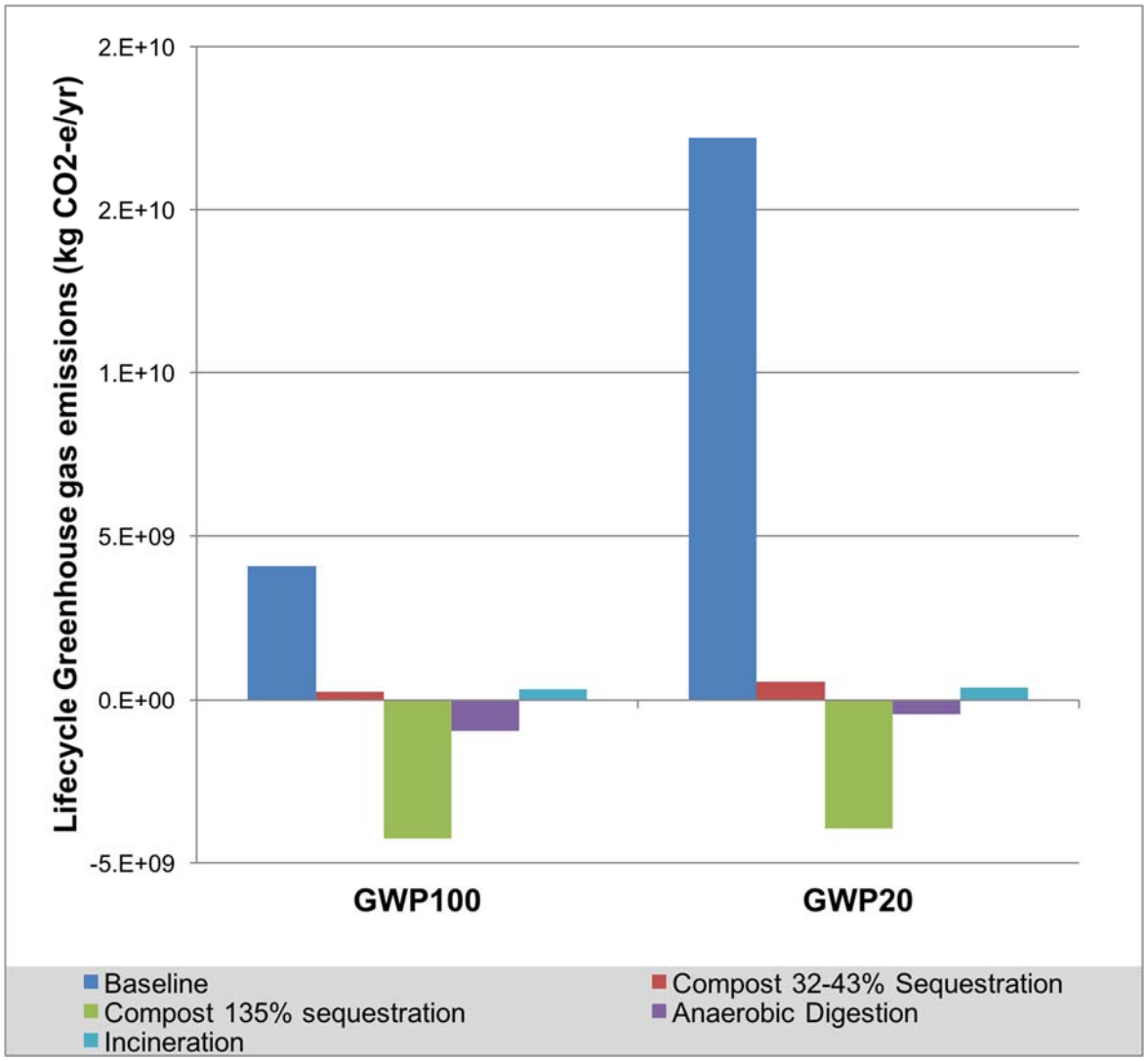

Figure 5.3: Net lifecycle greenhouse gas emissions from alternative ways of managing California's organic feedstocks (food waste, yard waste, cattle manure) for global warming potentials (GWP) of 100 and 20 years. The baseline case is blue, compost-1 is red, compost-2 is green, anaerobic digestion is purple, and incineration is teal. Both compost scenarios were based on empirical data from Ryals and Silver (2013), but compost-1 did not include the C in the compost itself in the estimate for C sequestration; in compost-2, the C in the compost was included in the sequestration benefit.

The raw values for net emissions are less important than the difference between each scenario's emissions and the baseline case; this difference is illustrated by the vertical arrow in Figure 5.3. For both GWP cases, the baseline case emitted the most, compost-2 emitted the least, and the remaining three did not differ from each other dramatically. Table 5.1 shows the difference in emissions between the scenarios and the baseline case. When looking at GWP on a 20-year basis, which is a time horizon that reflects more urgency in combatting climate change and is also a time scale within the lifetime of most people alive today, the performance of the baseline case appears far worse. This is due to the fact that the baseline case emitted a high proportion of CH_4 and N_2O, which are both powerful greenhouse gases; the potency of CH_4 increases when

analyzed on a shorter time horizon. Thus, all alternative scenarios perform even better on a shorter time horizon, as evidenced by the arrow on the right-hand side of Figure 5.3.

Table 5.1: Difference in net greenhouse gas emissions between each scenario and the baseline case (in kg CO_2-e/year).

	GWP100 (kg CO_2-e/year)	GWP20 (kg CO_2-e/year)
Baseline	0.00E+00	0.00E+00
Compost 32-43% Sequestration	-3.85E+09	-1.67E+10
Compost 135% sequestration	-8.34E+09	-2.12E+10
Anaerobic Digestion	-5.06E+09	-1.77E+10
Incineration	-3.76E+09	-1.68E+10

Over a 100-year time-frame, all scenarios emitted at least 3.8E9 kg CO_2-e y^{-1} less than the baseline case – this is equivalent to the electricity use of 570,000 homes in one year. Over a 20-year time frame, the savings are larger, with the minimum savings equivalent to the electricity usage of 2.6 million homes (EPA 2017).

The greenhouse gas savings stemmed from two major processes: avoided emissions and soil C sequestration. All scenarios avoided downstream emissions that would have occurred in the baseline scenario. For food waste and yard waste, the baseline treatment included landfilling, whose anaerobic conditions led to the formation of CH_4. For cattle manure, the baseline case included storage in anaerobic lagoons, which emitted CH_4 and N_2O, both powerful greenhouse gases. These emissions were largely avoided in the alterative scenarios. The anaerobic digestion and incineration scenarios, because they both produced energy on net, both avoided emissions from the marginal producers on the California electricity system, natural gas (Marnay et al. 2002).

The composting scenarios additionally sequestered C on net; the application of compost to California grasslands led to more grass growth, and thus more C uptake from the atmosphere. This C, once taken up by biomass, is largely stored belowground, where it becomes soil organic matter (Ryals and Silver 2013). This study used a conservative scenario, in which C sequestration was measured by only including the enhanced C storage propelled by the compost application – but not including the C added as compost – as well as a scenario in which that C in the compost was also included. We modeled enhanced C sequestration from digestate application in the anaerobic digestion scenario, though it should be noted that rigorous US-based data could not be found.

5.4 Summary

This study is the first to analyze the lifecycle greenhouse gas emissions from managing large fractions of California's organic waste – both municipal and agricultural. Results showed that treating food waste, yard waste, and cattle manure as a resource could yield large greenhouse gas savings from avoiding emissions as well as from enhanced soil C sequestration. This research relied on published studies for data on the various treatment processes modeled. Some of these processes are very well understood (e.g., landfilling, anaerobic digestion), and some are not as well characterized (C sequestration from digestate application, emissions from anaerobic

lagoons). These processes, and particularly the net carbon emissions or storage from these processes, contribute uncertainty to our model, and are areas that could benefit from future research.

6: Conclusions

Emissions reduction alone is no longer sufficient to resolve climate change (IPCC 2014). Removal of CO_2 from the atmosphere is a requisite component of any successful climate action planning. We explored the potential for the diversion of organic waste from high emitting waste streams to composting and subsequent land application to both lower emissions and sequester atmospheric C. Our results showed that compost application sequestered C during the first year after it was applied to a diverse set of rangeland ecosystems in California. These data are a critical first step to determining the broad scale potential of this practice for climate change mitigation. Over the first year, rangelands sequestered an average of 2.1 ± 1.0 Mg C ha^{-1}, similar to what was shown over three years in a valley and coastal rangeland in California (Ryals and Silver 2013). While these preliminary data are promising, additional research is needed to determine if this rate can be sustained, and to document the impact of compost amendments on greenhouse gas emissions, plant productivity, and ecosystem dynamics.

We used model simulations to estimate the long-term impacts of compost applications to California's rangelands under two different climate change scenarios. Our results suggested that compost amendments can lead to a net climate benefit under both an emissions reduction scenario (RCP 4.5) and a business-as-usual emissions assumption (RCP 8.5). Our results suggested that one-time 0.25-inch compost amendments can lead to a net savings of more than 8 MMT CO_2e over 15 y if applied to just 6% of California's rangelands. The actual C sequestration rates may vary, but trends, which are based on an up-to-date understanding of ecosystem behavior, are likely to be robust. The model suggested that the persistence of the C benefit would be greater under the emissions reduction scenario, highlighting a virtuous cycle linking emissions reduction and C sequestration. The climate model used in our analysis assumed a wetter future, particularly under the RCP 8.5 scenario, and this likely impacted the results. We are currently repeating the model runs using a scenario that predicts a drier future. These results will be critical to better understand the possible range of effects of compost amendments for California's future.

The composting process itself is likely to emit greenhouse gases, but the scale and drivers of these emissions are poorly understood. To address this gap in knowledge, we designed and deployed a greenhouse gas monitoring system for composting that facilitates the measurement of windrows under real working conditions. We followed emissions and a suite of environmental variables for two 100-day composting experiments with different starting feedstocks. Our results suggested that compost pile management may be as or more important in controlling emissions as the composition of feedstocks. Both compost piles resulted in CH_4 emissions when using standard practices. Future research should explore practices to lower CH_4 emissions, such as enhanced aeration. Interestingly, we found no simple relationships among emissions, O_2, moisture or temperature. However, CH_4 emissions were greatest when the pile O_2 dropped below 5%. Future work should explore potential additional drivers such as substrate availability and detailed microbial community dynamics.

Finally, the climate benefit of compost amendments to rangelands is dependent upon the assumption that organic waste diversion to composting and soil application results in a lower C footprint than other potential uses of organic waste such as landfilling and manure storage (current practices), incineration for energy generation, or anaerobic digestion for energy generation. Composting and land application had the greatest net climate benefit of the practices explored when the C stored in compost is accounted for. Anaerobic digestion followed by composting also had a large net climate benefit. While incineration exhibited a net C savings relative to waste storage in landfills and slurry ponds, it does not provide the same added co-benefits of field applied compost. The co-benefits of land application of compost are derived from a higher soil organic matter content and include higher soil nutrient content and nutrient retention, greater plant (e.g. crop or forage) productivity, lower erosion rates, more water holding capacity, and lower soil compaction (Conant 2011). An economic analysis of these practices is a necessary next step to determine their relative feasibility for California.

Overall, our research suggests that composting organic waste followed by land application to rangelands has considerable potential to enhance plant productivity and soil C storage over the short- and long-terms. The composting process is likely to emit greenhouse gases, particularly CH_4, and thus careful management will be required to lower emissions and increase the net benefit from the practice. The diversion of organic waste to composting directly, or anaerobic digestion followed by composting, has considerable potential to result in a significant net climate benefit when coupled with rangeland application. Future research should focus on long-term trends, explore a wider range of future climate scenarios, and test a wider range of feedstocks and practices in composting. This, coupled with LCA modeling will provide valuable information and contribute to effective climate change policy for California's working lands.

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APPENDIX A: Compost Information

ANALYTICAL CHEMISTS
and
BACTERIOLOGISTS
Approved by State of California

SOIL CONTROL LAB

42 HANGAR WAY
WATSONVILLE
CALIFORNIA
95076
USA

TEL: 831-724-5422
FAX: 831-724-3188
www.compostlab.com

Account #: 6080899-1/1-9431
Group: Sep16A #26
Reporting Date: September 14, 2016

West Marin Compost (Will Bakx)
P.O. Box 730
Nicasio, CA 94946
Attn: Will Bakx

Date Received: 29 Aug. 16
Sample Identification: WMC-GM,SF:Aug16 Compost
Sample ID #: 6080899 - 1/1

Nutrients-Primary + Secondary	Units	Wet wt. Basis	Dry wt. Basis	TMECC Method
Total Nitrogen:	%	1.1	1.7	4.02-D
Ammonia (NH_4-N):	mg/kg	160	240	4.02-C
Nitrate (NO_3-N):	mg/kg	3.5	5.2	4.02-B
Organic Nitrogen (Org.-N):	%	1.1	1.7	Calc.
Phosphorus (as P_2O_5):	%	0.73	1.1	Calc.
Phosphorus (P):	mg/kg	3200	4700	4.03-A
Potassium (as K_2O):	%	1.8	2.6	Calc.
Potassium (K):	mg/kg	15000	22000	4.04-A
Calcium (Ca):	%	1.2	1.7	4.05
Magnesium (Mg):	%	0.58	0.86	4.05
Sulfate (SO_4):	mg/kg	430	630	4.12-D/IC
Nutrients - Trace elements				
Copper (Cu):	mg/kg	22	32	4.05-Cu
Zinc (Zn):	mg/kg	83	120	4.05-Zn
Iron (Fe):	mg/kg	7200	11000	4.05-Fe
Manganese (Mn):	mg/kg	240	350	4.05-Mn
Boron (B):	mg/kg	27	40	4.05-B
Salts, pH, Bulk Density, Carbonates				
Sodium (Na):	%	0.22	0.32	4.05-Na
Chloride (Cl):	%	0.34	0.5	04.05/IC
pH Value:	units	8.91	NA	04.11-A
Electrical Conductivity (EC5 dw):	mmhos/cm	NA	5.9	04.10-A
Bulk Density :	lb/cu ft	30	20	SCL
Carbonates (as $CaCO_3$) :	lb/ton	17	24	04.08-A
Organic Matter:	%	40.5	59.4	05.07-A
Organic Carbon:	%	21	30	4.01
Ash:	%	27.7	40.6	3.02
C/N Ratio	ratio	17.6	17.6	calc.
Moisture:	%	31.8	0	3.09
AgIndex	ratio	6.5	6.5	SCL

To Calculate lbs/ton: (%Nutrient) x (20)
To Calculate lbs/ton: (mg/kg Nutrient/10,000) x (20)
To Calculate lbs/cu yd: (%Nutrient/100) x B.D. x 27
To Calculate lbs/cu yd: (mg/kgNutrient/1,000,000) x B.D. x 27

Analyst: Assaf Sadeh

APPENDIX B: Additional Figures

Compost Experiment Site Map

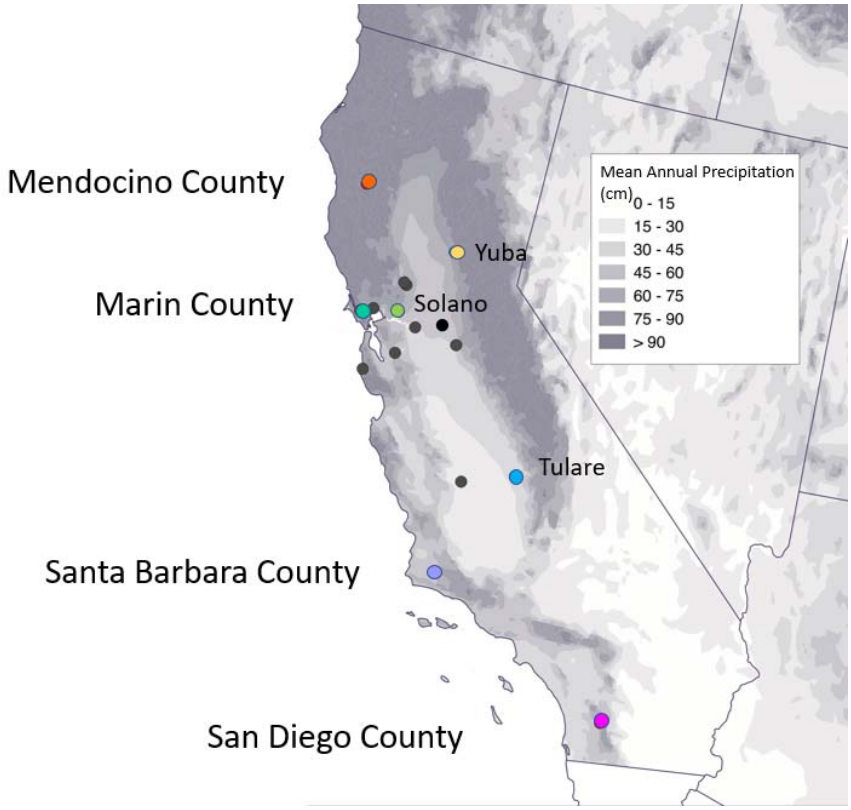

Figure B.1: Site locations for the Statewide Compost Addition Experiment. The seven modeled sites are color coded; the black dots are field sites that were not modeled. Map created by S. Grubinger.

Comparison of Climate Scenarios

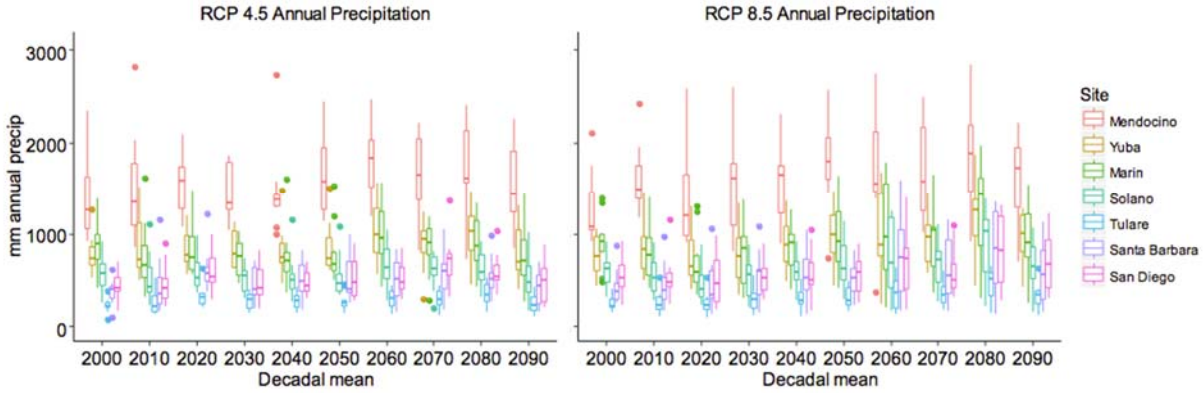

Figure B.2: Projected climate change under CanESM2-ES results in an increase of mean annual precipitation by decade. Annual precipitation varies more in the last half of the century under the high emission scenario (RCP8.5) compared to the reduced emission scenario (RCP4.5).

Climate change impacts

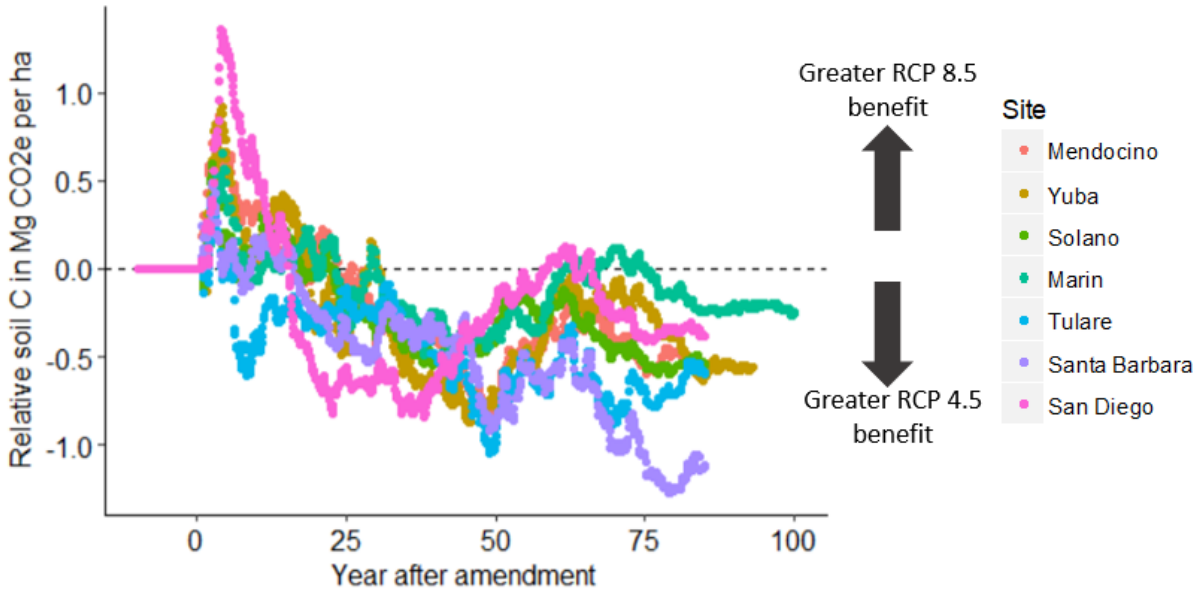

Figure B.3: The same compost amendment results in a greater climate benefit if combined with emissions reduction (RCP4.5) then in a high emissions scenario (RCP8.5). We calculated the difference between the climate benefit of compost for RCP8.5 and RCP4.5 and found that the benefit from RCP4.5 was larger than the benefit from RCP8.5 (points below the dotted line) in all sites throughout most of the century.