Title

Carbon and Greenhouse Gas Dynamics in Annual Grasslands: Effects of Management and Potential for Climate Change Mitigation

Author

Ryals, Rebecca

Publication Date

2012

Carbon and Greenhouse Gas Dynamics in Annual Grasslands: Effects of Management and Potential for Climate Change Mitigation

by

Rebecca Ryals

A dissertation submitted in partial satisfaction of the

requirements for the degree of

Doctor of Philosophy

in

Environmental Science, Policy, and Management

in the

Graduate Division

of the

University of California, Berkeley

Committee in charge:

Professor Whendee L. Silver, Chair
Professor Margaret S. Torn
Professor Todd E. Dawson
Professor Robert C. Rhew

Fall 2012

Carbon and Greenhouse Gas Dynamics in Annual Grasslands:
Effects of Management and Potential for Climate Change Mitigation

by Rebecca Ryals

Abstract

Carbon and Greenhouse Gas Dynamics in Annual Grasslands: Effects of Management and Potential for Climate Change Mitigation

by

Rebecca Ryals

Doctor of Philosophy in Environmental Science, Policy, and Management

University of California, Berkeley

Professor Whendee L. Silver, Chair

Ecosystem management practices that sequester carbon (C) may play an important role in mitigating climate change. Grasslands managed for livestock (*e.g.,* rangelands) constitute the largest land-use area globally. Critical components of the long-term sustainability of rangelands are the maintenance of net primary production (NPP) and soil organic carbon (C) pools. However, overgrazing, plant invasions, and climate change have led to significant C losses from many rangeland ecosystems. Thus, management practices may have considerable potential to restore or increase grassland C storage and help mitigate climate change. Practices that promote C sequestration may have valuable co-benefits, including increased forage production and improved soil water holding capacity. Despite the potential for C sequestration through management interventions, the question remains largely unexplored in grassland ecosystems.

I used a combination of laboratory experiments, field manipulations, and modeling simulations to examine the effects of rangeland management practices on C sequestration and greenhouse gas emissions. The specific goals of this research were to 1) assess the immediate and carry-over effects of management practices on the net C balance and greenhouse gas emissions in grasslands amended with compost, 2) measure changes to soil C and N stocks following amendment, 3) investigate the long-term fate of compost C and net climate change mitigation potential, and 4) explore the extent of tradeoffs between C sequestration strategies and vegetation characteristics.

In the first chapter, I conducted a three-year field manipulation replicated within and across valley and coastal grassland sites to determine the effects of a single application of composted organic matter amendment on net ecosystem C balance. Amendments increased C losses through soil respiration, and estimates of net C storage were sensitive to models of respiration partitioning of autotrophic and heterotrophic components. Over the three-year study, amendments increased C inputs by stimulating net primary production by 2.1 ± 0.8 at the coastal grassland and 4.7 ± 0.7 Mg C ha^{-1} at the valley grassland. Carbon gains through above- and belowground NPP significantly outweighed C losses, with the exception of a sandy textured soil at the coastal grasslands. Treatment effects persisted over the course of the study. Net ecosystem C storage increased by 25 to 70 % over three years, not including direct C inputs from the amendment.

The purpose of chapter two was to further investigate changes to rangeland soil C and N stocks three years after a one-time application of composted organic material. Increases in bulk soil C, though often difficult to detect over short timeframes, were significant at the valley grassland study site. Physical fractionation of soil revealed greater amounts of C and N in the free and occluded light fractions by 3.31 ± 1.64 and 3.11 ± 1.08 Mg C/ha in the valley and coastal grassland, respectively. Analysis of the chemical composition of soil fractions by diffuse reflectance infrared Fourier transform (DRIFT) showed chemical protection and inclusion of compost C into the light fractions. The combination of physical and chemical analyses suggests that the newly incorporated C was physically protected and less available for decomposition.

In the third chapter, I employed the ecosystem biogeochemical model, DAYCENT, to investigate the short (10 yr), medium (30 yr), and long-term (100 yr) climate change mitigation potential of compost amendments to grasslands. Climate change mitigation potential was estimated as the balance of total ecosystem C sequestration minus soil greenhouse gas emissions and indirect emissions of N_2O via nitrate leaching. The model was parameterized using site-specific characteristics and validated with data from the three-year field manipulation. Model simulations included variations in the applications rate and C:N ratio of the composted material. Above- and belowground NPP and soil C pools increased under all amendment scenarios. The greatest increase of soil C occurred in the slow pool. Ecosystem C sequestration rates were highest under low C:N scenarios, but these scenarios also resulted in greater N_2O fluxes. Single or short-term applications of compost resulted in positive climate change mitigation potential over 10 and 30-year time frames, despite slight offsets from increased greenhouse gas emissions.

Finally, chapter four examined important tradeoffs between rangeland C sequestration activities and vegetation characteristics. I measured aboveground biomass, plant N content, vegetation communities, and the abundance of noxious weed species for four years following single management events of compost amendment, keyling plowing, and a combination of amendment and plowing. During the first year, plant N content and aboveground biomass was significantly higher in the amended plots and lower in the plowed plots. In the amended plots, forage quantity and quality increases were sustained over the four-year study. During spring grazing events, cows consumed more forage from amended plots without adversely increasing grazing impacts on residual biomass. Plant communities at both grasslands were relatively resistant to management events, however there were short-term declines in the abundance of a noxious annual grass at the valley grassland and increases in a noxious forb at the coastal grassland.

Grassland management practices, such as the application of composted organic matter, have considerable potential to mitigate climate change while improving plant production, soil fertility, and diverting organic wastes from landfills. This research illustrates the potential for grassland management to sequester while explicitly considering impacts on greenhouse gas emissions, plant production, and vegetation communities over multiple time frames. Overall, my dissertation contributes toward a better understanding of the role of ecosystem management interventions in climate change mitigation.

Table of Contents

Acknowledgements

I express my deep appreciation to my advisor, Whendee Silver, whose unwavering support and guidance has contributed greatly to my development as a scientist. Her commitment to conducting novel research, applying basic science to explore solutions to environmental problems, and communicating science beyond academia is an inspiration. There are few people that have affected my life so profoundly and so positively. I am forever grateful to be a part of the scientific community she has created at UC Berkeley and beyond. Sincere thanks also goes to my other committee members, Margaret Torn, Todd Dawson, and Rob Rhew, who provided valuable insight, thoughtful discussions, and critical feedback which greatly improved the quality of my dissertation.

I appreciate my scientific collaborators who helped me learn new scientific methods to extend the scope of this research. Bill Parton, Melannie Hartmann, and Cindy Keough were incredibly generous with their time and patience during my modeling research. Asmeret Berhe and Michael Kaiser provided access to and assistance with the FTIR analysis. Cristina Castanha and Rachel Porras gave advice on soil carbon fractionation methods and helped me navigate the rules at LBL. Val Eviner and Claudia Stein helped design and execute vegetation sampling.

Special thanks go to my partners in the Marin Carbon Project, including the following individuals: John Wick, Peggy Rathmann, Jeff Creque, Nancy Scolari, Bob Berner, Particia Hickey, David Lewis, Stephanie Larson, Stacy Carlsen, Torri Estrada, and Charlotte Epifano; and agencies: Marin Agricultural Land Trust, Marin Organic, Marin Resource Conservation District, UC Cooperative Extension, and the USDA Natural Resource Conservation Service. It has been a true pleasure to interact with my MCP partners, to listen to their perspectives, and to have a venue for applying for communicating and applying science. I am especially grateful to the heart of the Marin Carbon Project - John Wick, Peggy Rathmann, and Jeff Creque. Jeff's expertise in compost helped carve the path for this research. John and Peggy kindly committed their time, property, and creativity to this research. Their enthusiasm for the carbon cycle and vision for a better world inspire action and optimism.

I would like to thank my fellow graduate students and postdocs in the Silver lab, past and present, for being wonderful colleagues and friends. In particular, I would like to thank Wendy Chou, Daniela Cusack, Marcia DeLonge, Steven Hall, Dan Liptzin, Gavin McNicol, Erika Marin-Spiotta, Justine Owen, Tana Wood, Yit Teh, and Wendy Yang. Special thanks goes to Andy Thompson, Marissa Lafler, and Andrew McDowell for being creative, intelligent, and helpful lab managers throughout the years. This research would not have been completed without the help of many hardworking and enthusiastic lab technicians, including Maya Almaraz, John Bister, Wanda Cea, Julia Cosgrove, Zoe Statman-Weil, Jonathan Treffkorn, and Michelle Wong. Several undergraduates also spent long hours in the lab and field. I thank the following undergraduate students for their invaluable contribution to this work and for their positive energy, even during grueling, hot summer days: Katie Clabeaux, Logan Curtis-Whitchurch, Katie Graves, Clayton Ernst, Lisa Kim, Christine Kyauk, Judy Li, Allegra Mayer, Taichi Natake, Edna Nyangau, and Kasturi Sangupta. I had the pleasure of advising Gary Gao, Luke Lintott, and Willis Hon on their undergraduate research projects, which complemented my own research.

The staff and facilities at the UC Sierra Foothill Research and Extension Center (SFREC) were invaluable to this work. Special thanks goes to Dustin Flavell who always greeted me with a smile and responded to all my requests for support with a friendly “you bet!”. Chris Fedderson, Martin Beaton, Abe Mendoza, Johnny Wedgeworth, and Mike Williams also made SFREC a fantastic place to work.

I am grateful for the encouragement and moral support from my friends and family. I am incredibly thankful to Gordon Bennett. He has been my foundation, and his love and partnership made my time at Berkeley the best years of my life. I thank Lindsay Chiono for our study dates and our friendship throughout grad school. I also thank my parents who have taught me to appreciate and pursue life's opportunities, my siblings for their lifelong friendship, and my nephews and niece for reminding me that the future needs a healthy planet.

My research was supported by generous grants from the Marin Community Foundation, Rathmann Family Foundation, the Lia Fund, a USDA Conservation Innovation Grant to Environmental Defense Fund, and the Kearney Foundation for Soil Science.

Chapter 1. Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands

Abstract

Most of the world's grasslands are managed for livestock production. A critical component of the long-term sustainability and profitability of rangelands (e.g. grazed grassland ecosystems) is the maintenance of plant production. Amending grassland soils with organic waste has been proposed as a means to increase net primary productivity (NPP) and ecosystem carbon (C) storage, while mitigating greenhouse gas emissions from waste management. Few studies have evaluated the effects of amendments on the C balance and greenhouse gas dynamics of grasslands. I used field manipulations replicated within and across two rangelands (a valley grassland and a coastal grassland) to determine the effects of a single application of composted green waste amendments on NPP and greenhouse gas emissions over three years. Amendments elevated total soil respiration by 18 ± 4 % at both sites, but had no effect on nitrous oxide or methane emissions. Carbon losses were significantly offset by greater and sustained plant production. Amendments stimulated both above- and belowground NPP by 2.1 ± 0.8 to 4.7 ± 0.7 Mg C ha^{-1} over the three-year study period. Net ecosystem C storage increased by 25 to 70 % without including the direct addition of compost C. The estimated magnitude of net ecosystem C storage was sensitive to estimates of heterotrophic soil respiration, but was greater than controls in five out of six fields that received amendments. The sixth plot was the only one that exhibited lower soil moisture than the control, suggesting an important role of water limitation in these seasonally dry ecosystems. Treatment effects persisted over the course of the study, which were likely derived from increased water holding capacity in most plots, and slow-release fertilization from compost decomposition. I concluded that a single application of composted organic matter can significantly increase grassland C storage and that effects of a single application are likely to carry over in time.

Introduction

Rangelands represent the largest land-use area globally, covering more than one-quarter of the world's land surface and storing approximately one-third of the world's terrestrial C in soils and vegetation (White et al. 2000, Asner et al. 2004). The primary economic output of rangelands is livestock production, supplying meat, dairy products, leather, and wool (Herrero et al. 2009). The sustainability and profitability of rangelands is largely a function of forage quality and quantity (Briske et al. 2011). However, these ecosystems have, to varying degrees, experienced degradation of vegetation and soils due to overgrazing, plant invasions, and climate change (Asner et al. 2004, Schipper et al. 2007, Bai et al. 2008). Thus, management practices aimed at enhancing plant production may have considerable potential to restore or increase grassland C storage and feed back on the global C cycle (Schimel et al. 1990, Conant et al. 2001, Follett et al. 2001, Schuman et al. 2002, Derner et al. 2007).

Management practices can affect grassland C storage or loss by altering soil chemical or physical characteristics (Cambardella and Elliott 1992, Paustian et al. 1997, Janzen et al. 1998), plant morphology or growth, soil moisture, or rates of microbial activity (Stromberg and Griffin 1996, Steenwerth et al. 2002, Jones and Donnelly 2004). Amending soils with organic material generally increases nutrient availability, and thus is a common practice used in cropping systems to enhance NPP (Cassman et al. 2002, Blair et al. 2006) and in some land reclamation sites to facilitate soil amelioration and plant establishment (Larney and Angers 2012). Animal manure, crop residues, composted urban waste, and sewage sludge are common forms of organic matter amendment.

The application of organic matter to rangelands has been proposed as an approach for increasing plant productivity, as a waste management strategy, and for climate change mitigation (Cabrera et al. 2009, Hall and Sullivan 2001). Organic matter additions to rangeland soils increase soil C pools directly and have the potential to indirectly increase ecosystem C storage by stimulating plant growth. Organic matter additions to rangelands can also provide a pathway to divert organic waste from landfills or for manure management from nearby dairies, thereby reducing greenhouse gas emissions from traditional waste management.

Carbon benefits of enhanced NPP due to organic matter amendments may be offset from a global warming perspective by the stimulation of soil greenhouse gas emissions. Organic matter amendments increase soil C and nitrogen (N) pools and may alter soil environmental conditions (e.g., moisture, temperature, and pH), thereby increasing the potential for carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) emissions (Gregorich et al. 2005). The extent of management effects on soil greenhouse gas emissions is a large source of uncertainty in grasslands (Soussana et al. 2004). Manure amendment can increase CO_2 and N_2O fluxes (Chadwick et al. 2000, Dalal et al. 2003, Mosier et al. 2004, Davidson et al. 2009); composted animal waste and plant matter tends to result in lower greenhouse gas emissions relative to green manures or synthetic fertilizers (Vallejo et al. 2006, Alluvione et al. 2010). However, the effects of organic matter additions on greenhouse gas dynamics in rangelands are largely unstudied (Lynch et al. 2005, Cabrera et al. 2009).

The purpose of this study was to examine the immediate and residual effects of amendments of composted green waste on plant production and greenhouse gas emissions in annual grasslands. I hypothesized that the application of composted organic matter to rangeland soils would increase the above- and belowground net primary productivity (NPP) for at least one year and that these increases in ecosystem C inputs would be partially or wholly offset by elevated rates of soil greenhouse gas emissions (CO_2, N_2O, and CH_4). I tested this hypothesis over three years using replicated field experiments in the two dominant annual grassland types in California. Unlike perennial grassland systems, annual grasses germinate, grow and die over an annual cycle allowing us to estimate net ecosystem C storage from the changes in plant and soil C pools and fluxes.

Materials and Methods

Study sites

Rangelands are the dominant cover type in California covering an estimated 17-23 million ha (FRAP 2003, Brown et al. 2004). Of this area, approximately 9 million ha are valley or coast range grasslands (Kuchler 1964). Valley grasslands extend along the central and southern parts of California and are comprised largely of non-native annual grass and forb species such as *Avena barbata*, *Bromus hordeaceus*, *Lolium multiflorum*, *Erodium* spp., and *Trifolium* spp. (Bartolome et al. 2007), as well as invasive species such as *Taeniatherum caput-medusae*. Coast range (hereafter coastal) grasslands experience a mesic coastal climate and are also dominated by a similar mix of non-native annual grasses. Native perennial grass species, such as *Danthonia californica* and *Stipa pulcra* can also occur (Jackson and Bartolome 2002). The Mediterranean climate of both valley and coastal grasslands of California is characterized by cool, wet winters and warm, dry summers. The growing season begins with the rains in September-November and ends with the onset of the dry season in April-June.

The field experiment was conducted on valley grasslands at the Sierra Foothill Research and Extension Center in Browns Valley, CA (39.24 °N, 121.30 °W) and on coastal grasslands in Nicasio, CA (38.06 °N, 122.71 °W). Annual precipitation averages 730 mm y^{-1} (22 y mean) at the valley grassland and 950 mm y^{-1} at the coastal grassland. During the years of this experiment, the valley grasslands received 380, 641, and 843 mm/y of precipitation, primarily as rainfall; mean air temperatures ranged from 2 °C in January to 35 °C in July. Soils are derived from Mesozoic and Franciscan volcanic rock and classified as xeric inceptisols and alfisols in the Auburn-Sobrante complex (http://casoilresource/lawr.ucdavis.edu/soilsurvey, Beaudette and O'Geen 2009). The site has been grazed by cattle for at least 150 years (D. Flavell, pers. comm.). At the coastal grassland, annual precipitation during the study period was 771, 1050, and 1163 mm/y, and mean air temperatures ranged from 6 °C in January to 20 °C in July. Soils are derived from Franciscan mélange and classified as mollisols in the Tocaloma-Saurin-Bonnydoon series (Beaudette and O'Geen 2009, http://casoilresource/lawr.ucdavis.edu/soilsurvey). The site has been grazed by cattle since at least 1900, with a brief period of exclusion from 2000-2005.

Experimental Design

The field experiment was established in October 2008 and continued for three growing seasons to August 2011. Treatments consisted of untreated controls and composted organic matter amendments. Plots were 25 m by 60 m buffered by a ≥ 5 m strip arranged in three randomized complete blocks. Blocks were situated in different microwatersheds in each grassland type to capture some of the landscape-scale heterogeneity associated with vegetation communities, potential residual effects of field-scale land use history, and soil texture. An organic-rich soil amendment was produced by adding a single application of commercially available composted organic green waste (Feather River Organics, Marysville, CA) with a N concentration of 1.87 % and a C:N ratio of 11. A thin surface dressing approximately 1.3 cm thick (equivalent to 1.42 kg C/m^2 and 129 g total N/m^2, or 7.0 kg dry matter m^{-2}) was applied in December 2008. All plots were grazed using a rotational regime typical of the region to achieve a residual dry matter level of approximately 130 g/m^2. Sites were grazed during the spring and fall for up to four weeks, depending on the amount of available forage. During grazing, cattle were not isolated within plots, but instead allowed to graze the entire block.

Climate Data, Soil Moisture, and Soil Temperature

Local daily rainfall and air temperature records were obtained from Browns Valley for the valley grassland site (http://wwwcimis.water.ca.gov/cimis/data.jsp) and from the Marin Municipal Water District (J. Klein, pers. comm.) for the coastal grassland site. Water years were defined as the period that encapsulates one rainy and one dry season, spanning from September 1 through August 31 of the following year (Chou et al. 2008).

In May 2009, automated TDR-based probes were installed at 0 to 30 cm depth to measure volumetric water content (Campbell Scientific CS616, n = 3 per plot). Automated soil temperature probes were installed at 0 to 5 cm depth (Campbell Scientific 108L, n = 1 per plot). Soil moisture and temperature data output was averaged on hourly and daily timescales.

Soil Greenhouse Gas Emissions

Soil respiration was measured weekly during the growing season and every other week during the summer over the first two water years. Soil CO_2 fluxes are extremely low with little temporal variability during the summer in annual grasslands (Nijs et al. 2000, Xu and Baldocchi

2004, Chou et al. 2008). I measured soil respiration monthly during the third water year. Soil CO_2 fluxes were measured during daylight hours using a LI-8100 infrared gas analyzer (approx. 2.5 min flux periods, Li-Cor Biosciences, Lincoln, NE) fitted with an opaque soil efflux chamber. The chamber was used in a survey mode with polyvinyl chloride collars (n = 5 per plot). Collars were driven approximately 3 cm into the soil and left to equilibrate for at least 30 minutes before flux measurements were taken. Soil respiration measurements were started three (coastal) to six (valley) weeks prior to organic matter applications. As a first approximation of annual soil respiration rates I used linear interpolation between sampling time points and summed the resulting data as an estimate the mass of C per unit area respired over the water year (Silver et al. 2005, Chou et al. 2008). Previous work at the valley grassland identified the importance of the fall wet-up event for estimating accurate annual soil respiration fluxes (Chou et al. 2008). In attempt to capture the largest fluxes, I measured soil respiration no more than 48 hours before and after fall wet up events.

Soil CH_4 and N_2O fluxes were measured bi-weekly for the first six months, which was when I expected to see the largest fluxes (Chou et al. 2008), and monthly thereafter until September 2010. Vented static flux chambers (Keller & Reiners 1994) were placed at four random locations within each plot during each sampling period. Thirty mL gas samples were collected from the chamber headspace at 0, 5, 15, 25, and 40-minute time points, immediately transferred to evacuated glass vials, and analyzed within 72 hours (Chou et al. 2008). Methane concentrations were analyzed on a gas chromatograph using a flame ionization detector, and an electron capture detector was used to analyze N_2O concentrations. Fluxes were then calculated using an iterative exponential curve fitting approach (Matthias et al. 1978).

Wet up events can stimulate trace gas emissions, particularly in dry, organic rich soils (Fierer and Schimel 2002, Chou et al. 2008). I conducted a laboratory incubation experiment to determine the potential trace gas emissions from amended and control soils during wet up events. Soils from the valley grassland were collected from control and amended plots (n = 9 per plot) and composited by treatment. Approximately 200 g of soil were placed into quart-sized mason jars and categorized as one of four incubation treatments (n = 5): control, control + wet up, amended, and amended + wet up, where control and amended treatments refer to ambient dry-season field moisture conditions (approximately 4 % gravimetric soil moisture) and wet up refers to the addition of 40 mL of deionized water representing an average first rainfall event in autumn. Jars were incubated in a dark growth chamber with daily air temperatures fluctuating from 20 to 35˚C, a typical range experienced during wet up events in the field. Soil CO_2, N_2O, and CH_4 fluxes were measured daily until trace gas fluxes were not significantly different between treatment and controls (30 days). Cumulative fluxes were calculated as described above. Positive soil greenhouse gas fluxes indicate net source to the atmosphere, whereas negative values indicate a terrestrial sink.

Aboveground and Belowground Net Primary Productivity

The vegetation communities at the study sites are strongly dominated by annual grass and forb species. These plants germinate at the onset of the fall rains and die at the end of the wet season. Aboveground biomass at the end of the wet season plus biomass removed during earlier grazing events is equivalent to aboveground net primary production (ANPP) (Harper et al. 2005). Aboveground biomass was determined by harvesting plants in a 200 cm^2 area (n = 9 per plot) at the time of peak biomass at the end of the growing season. All aboveground vegetation was clipped to the soil surface. Harvested plant material was dried at 65 ^oC, weighed, and analyzed for C concentration on a Carlo Erba Elantech elemental analyzer (Lakewood, NJ). Carbon content of aboveground biomass was determined by multiplying C concentration by mass. Aboveground biomass was also measured using the above approach no more than 24 hours immediately before and after grazing to determine biomass removed by cattle (Marshall et al. 1998). Aboveground NPP was determined by adding the biomass removed by cows to the biomass collected at peak standing biomass. Plots at the coastal grassland were grazed during or just before peak standing biomass due to site-specific management requirements, thus biomass measurements were made prior to grazing at this site and may slightly underestimate ANPP.

The root systems of annual grasses are concentrated in the soil surface layer, with less than 15 % of root biomass occurring below 15 cm and a majority of root growth occurring before April (Jackson et al. 1988). Belowground net primary productivity (BNPP) was determined by measuring root biomass during late spring of water years 2 and 3 and adjusting values by a multiplication factor of 1.5 to account for intra-annual root turnover (Higgins et al. 2002). Eight 0 to10 cm and four 10 to 20 cm quantitative soil cores (6 cm diameter) were sampled per plot. Roots from each core were extracted according to Metcalfe et al. (2007) with the following modifications based on methods testing with soils from these sites. Roots from each core were picked by hand for three ten-minute intervals, sieved at 2 mm to break up large aggregates, and picked by hand for five additional ten-minute intervals for a total of 80 minutes. Roots were then rinsed free of soil and dried at 65 ºC until reaching a constant dry mass. For each core, the pattern of cumulative extraction over time was used to predict total root biomass. Upon drying and weighing, roots were finely ground and analyzed for C concentration on a Carlo Erba Elantech elemental analyzer (Lakewood, NJ) using acetanilide as a standard. Carbon content of belowground biomass was determined by multiplying C concentration by mass. All production data (ANPP, BNPP, and NPP) are expressed in units of mass of C per area using the biomass to C content conversions described above. Expressing production terms using this convention allows us to explicitly compare C inputs and outputs.

Soil carbon content and bulk density

Soil bulk density (0-10 cm depth) was measured by digging pits (one per plot) and carefully excavating 9 cm diameter cores approximately 5 cm back from an undisturbed face of the pit. All bulk density measurements were rock-corrected. Upon extraction from cores, dry rock masses and volumes were measured and subtracted from initial bulk density calculations.

Total soil organic C concentrations were measured prior to the application of organic matter amendment and at the ends of each subsequent growing season (May or June). Soils were collected using a 7 cm diameter corer to 10 cm depth ($n = 9$ per plot). Large roots and identifiable compost fragments were removed by hand. Soils were then air-dried and pulverized with a ball grinder (SPEX Sample Prep Mixer Mill 8000D, Metuchen, NJ). Carbon concentrations were measured using a Carlo Erba Elantech elemental analyzer (Lakewood, NJ) using atropine as a standard and converted to content using bulk density values specific to each plot.

Modeling Net Change in Ecosystem Carbon Storage

I modeled the annual net change in ecosystem C storage with and without compost additions using an approach modified from Hanson et al. (2000) and Chou et al. (2008):

$$\text{Net C Storage} = \text{ANPP} + \text{BNPP} - R_h - R_{oa} - \text{DOC}$$

where R_h is heterotrophic respiration which includes the non-root and non-rhizosphere component of soil respiration during the growing season. R_{oa} is CO_2 evolved from decomposition of the organic matter amendment, and DOC is dissolved organic C losses.

In annual grasslands, the sum of ANPP and BNPP represents total annual litter inputs, which turnover approximately annually (Heady et al. 1992). Partitioning soil respiration into its autotrophic and heterotrophic components is difficult in annual grasslands, and outside the scope of this study. Instead, I modeled net ecosystem C storage with a range of R_h values to represent a low (30 %), medium (50 %), and high (60 %) level measured in similar ecosystems (Craine et al. 1999, Zhou et al. 2007, Chou et al. 2008). Carbon losses through R_h were constrained to the growing season, as no treatment differences in soil respiration were detected during the dry summer months. I assumed that the relative proportion of soil respiration components was equal across treatments as a first approximation. I used 0.05 yr^{-1} as the decomposition rate of the compost (Lynch et al. 2005) to calculate annual losses of compost as CO_2, which is similar to the rate derived from a DayCent model simulation (Ryals et al. in prep). I assumed that the compost mass reduction via decomposition was lost primarily as CO_2 to the atmosphere. Carbon losses through leaching of DOC are negligible in these ecosystems relative to soil CO_2 effluxes (Dahlgren and Singer 1994, Sanderman and Amundson 2009). Therefore, DOC is assumed to be zero for estimates of net C storage. In this experiment, treatment plots were amended only once, at the beginning of water year 1. Therefore, direct C inputs from compost addition are constrained to water year 1, but losses through decomposition carry over all three years. Belowground NPP was estimated for water year 1 using the plot-specific root:shoot ratios averaged for water years 2 and 3.

Statistical analysis

One way analyses of variance (ANOVA) were used to identify statistically significant treatment effects on soil organic C, cumulative soil CO_2 flux, soil moisture, soil temperature, ANPP, BNPP, and net ecosystem C storage. Analyses included a blocking effect and were performed separately for valley and coastal sites. To assess changes over time within sites, I used repeated measures multivariate analysis of variance (MANOVA) with ANPP, BNPP, soil moisture, soil temperature, and soil CO_2, N_2O, and CH_4 fluxes as response variables. Each grassland type was analyzed separately with block, treatment, time, and interactions as MANOVA model effect factors. Correlations between net C storage, ecosystem C pools, soil moisture, and precipitation were explored using multiple linear regressions.

Statistical tests were performed using JMP 7.0.2 (SAS Institute Inc.). Variables that were not normally distributed were log transformed to meet assumptions for ANOVA. Data are reported either as mean values or treatment differences followed by ± 1 standard error. Statistical significance was determined as $p < 0.10$ unless otherwise noted.

Results

Response of Soil Respiration and Greenhouse Gas Emissions to Organic Matter Amendment

Soil respiration (root plus microbial respiration) was significantly higher in amended soils than in control soils during the growing season ($p < 0.0001$ at both sites) (Figure 1). There was approximately 18 ± 2 % and 19 ± 2 % more C respired from the amended soils relative to the controls from the valley ($p = 0.06$) and coastal ($p = 0.10$) sites cumulatively over the three years of the study (Figure 2). Within sites soil CO_2 fluxes varied by as much as 30 to 50 % among years, reflecting patterns in the timing and amount of rainfall. Soil respiration was positively correlated to soil moisture ($R^2 = 0.51$, $p < 0.001$ at valley and $R^2 = 0.65$, $p < 0.001$ at coastal), and treatment effects (amended-control) were positively correlated to annual precipitation ($R^2 = 0.47$, $p < 0.05$ for both sites). The highest cumulative soil respiration losses occurred during water year 2 which was characterized by late warm season rains, and treatment differences in cumulative soil respiration were not statistically significant during this time period.

Approximately 88 ± 0.01 % and 73 ± 0.07 % of the cumulative annual soil CO_2 efflux occurred during the growing season at the valley and coastal grassland, respectively, with no statistically significant treatment effect on the seasonal distribution of soil respiration. Analysis of paired-plot treatment differences of growing season soil respiration revealed that amended soils at the valley grassland lost an additional 614 ± 191 g CO_2-C m^{-2} over the three-year study. Similar trends were observed at the coastal grassland, which experienced increased losses of 646 ± 162 g CO_2-C m^{-2} from the amended plots.

There were no significant treatment effects on CH_4 and N_2O fluxes, and no significant changes over time. At the valley grassland, mean CH_4 fluxes were -2.5 ± 0.6 g CH_4-C ha^{-1} day^{-1}, and mean N_2O fluxes were 0.13 ± 0.13 g N_2O-N ha^{-1} day^{-1}. At the coastal grassland, mean CH_4 fluxes were -1.4 ± 0.7 g CH_4-C ha^{-1} day^{-1}, while mean N_2O fluxes were 1.0 ± 0.4 g N_2O-N ha^{-1} day^{-1}.

Greenhouse gas emissions during wet up events

I conducted a controlled laboratory experiment to estimate the potential greenhouse gas emissions associated with wet-up events. Wet-up led to a pulse of CO_2 from both amended and control soils, and soil respiration in amended soils remained elevated over controls for 22 days ($p < 0.05$). Nitrous oxide fluxes were also elevated from amended soils and lasted four days following wet up ($p < 0.01$; Figure 3). Methane fluxes were negligible and did not respond to the wet up event. Cumulative N_2O emissions accounted for just 0.49 ± 0.05 and 0.83 ± 0.13 % of the total global warming potential during the wet up event from control and amended soils, respectively. The vast majority of the greenhouse gas emissions from these soils was from CO_2.

Response of Net Primary Productivity and Soil C to Organic Matter Amendment

Organic matter amendments significantly enhanced plant growth at both the valley and coastal grassland sites (Appendix 1). During the first water year, amendments increased ANPP by 70 % at the valley site and by 44 % at the coastal site. The effects of the one-time application of organic matter carried over into the two subsequent growing seasons at a similar magnitude. Over all three study years, ANPP increased by a total of 436 ± 68 and 161 ± 78 g C m^{-2} ($p \leq 0.01$) at the valley and coastal grasslands, respectively (Figure 4). Average annual increases in ANPP were 145 ± 16 and 54 ± 3 g C m^{-2} at the valley grassland and coastal grassland, respectively ($p < 0.05$ for all years at both sites). Shoot to root ratios increased significantly with organic matter applications at the valley grassland from 3.75 ± 0.06 to 5.45 ± 0.09 ($p = 0.08$) and did not change significantly at the coastal grassland, where the shoot to root ratio was 3.00 ± 0.14. Root biomass increased at both sites, particularly in the 0-10 cm depth (Figure 4). At the valley grassland, root biomass increased by 33.6 ± 11.7 g C m^{-2} in amended plots at the 0 to 20 cm depth over three years ($p < 0.05$). Belowground NPP showed similar, but more variable trends with amendments at the coastal grassland. Organic matter amendments significantly increased total NPP by 2.0 ± 0.8 Mg C ha^{-1} at the coastal grassland ($p = 0.10$) and 4.7 ± 0.7 Mg C ha^{-1} at the valley grassland ($p < 0.01$) over the three year study period (Figure 5).

By the end of the first water year, organic matter additions significantly increased the soil organic C pool by 24 % in the 0 to 10 cm depth at the valley grassland site ($p = 0.06$). Soil C stocks remained larger in water years 2 and 3 ($p = 0.01$) (Figure 6).

Changes to net ecosystem carbon storage

I assessed the net impact of organic matter amendments at the field scale through a mass balance of inputs and outputs. When including C added in compost, the amended plots showed an increase in net ecosystem C storage of 17.7 ± 1.4 and 13.8 ± 1.8 Mg C ha^{-1} ($p < 0.0001$) at the valley and coastal grassland, respectively. A large portion (65 to 88 %) of the additional ecosystem C was due to the direct addition of C from the compost amendment in water year 1, only 12 % of which decomposed during the three year study period.

Organic matter additions also altered ecosystem C fluxes. Above- and belowground NPP increased, as did C emissions via soil respiration. Modeled C storage from both control and amended plots was consistently negative across years and sites (Appendix 1), indicating that these annual grasslands were losing C, findings that have been observed in many grassland ecosystems (Ojima et al. 1994, Bellamy et al. 2005, and Chou et al. 2008). The net balance of C fluxes (excluding compost amendments) was consistently more positive for all amended plots compared to control plots at the valley grassland site and at two out of three paired plots at the coastal grassland (Figure 7). The magnitude of the response was sensitive to assumptions about the contribution of heterotrophic respiration to total soil respiration. Assuming 50 % R_h, valley grasslands gained 198 ± 50 to 582 ± 43 g C/m^2 over three years following a single application of compost. Amended fields from two sets of paired plots at the coastal grassland showed similar gains of 192 ± 75 and 241 ± 96 g C/m^2. The third showed a decline in C storage by 281 ± 60 g C/m^2. For this pair of plots, soil volumetric water content was lower in the treatment than in the control, whereas the opposite trend was observed for all other fields (Figure 1).

Discussion

Patterns in soil respiration and greenhouse gas emissions

Organic matter amendments to surface soils increased soil respiration by an average of 18 to 19 % over the three-year study. Soil respiration is the combination of heterotrophic and autotrophic respiration, and both could have been stimulated by compost additions. Adding composted organic material to the soil surface likely increased the decomposition rate of the compost, although compost, having already experienced intensive decomposition during formation, is generally considered more stable material than fresh litter (Bernal et al. 1998, Goyal et al. 2005). Approximately 12 % of the added compost decomposed over the three-year study; this resulted in the emission of 171 g CO_2-C/m^2, or approximately 3.5 ± 0.2 % of the total C respired from amended soils over the study. Compost decomposition contributed 24 ± 6 % of the treatment differences in soil respiration. If I assume that autotrophic respiration accounted for 50 % of the total soil respiration measured in both treatment and controls (Hanson et al. 2000), then the remaining heterotrophic respiration (not derived directly from compost) was only approximately 6 ± 3 % greater in the amended treatment than in the control. This indicates that a priming effect by the compost on background soil C losses, if it occurred, was small (Kuzyakov et al. 2000, Kuzyakov 2010). It is important to note that soil respiration rates are temporally dynamic and respond rapidly to changes in soil moisture and temperature. Interpolation of measurements in time may miss important CO_2 fluxes.

Organic matter amendments could have altered the ratio of autotrophic to heterotrophic respiration in these soils. I provide evidence here that both autotrophic and heterotrophic respiration were likely to have increased as a result of the amendments. The tight coupling of root and microbial respiration in annual grassland soils may limit the degree to which their ratio can change. For example, the morphology and phenology of annual grasses results in little carbohydrate storage relative to perennial plants, thus root respiration is likely to be directly proportional to plant activity at short temporal scales (Warembourg and Estelrich 2001, Tang and Baldocchi 2005). Plant activity increased as a result of compost additions, which likely increased autotrophic respiration. Similarly, the majority of heterotrophic respiration in soils comes from recently derived organic matter (Trumbore 2000; Giardina and Ryan 2002, Carbone and Trumbore 2007). During the growing season, most of this input occurs belowground in annual grasslands (Higgins et al. 2002). Therefore, rates of heterotrophic respiration were likely to have increased as a result of increased plant activity (i.e. the production of labile organic matter), and are likely to be relatively tightly coupled with NPP in annual grassland soils.

I predicted that compost additions would increase the production and emissions of N_2O and CH_4 from rangeland soils. Grasslands can be an important source of N_2O (Bouwman et al. 1993), and manure application to rangeland soils can significant increase N_2O emissions (Rees et al. 2004, Jones et al. 2007). Organic matter amendments increase both the amount of mineral N in soils and water holding capacity of soils, creating conditions favorable for N_2O production via nitrification and denitrification (Firestone and Davidson 1989). Similarly, I expected that organic matter amendments would increase CH_4 emissions or decrease the net rate of CH_4 uptake by soils (Le Mer and Roger 2001, Moiser et al. 2004). Although soil moisture was higher in amended fields, I saw no significant changes in N_2O or CH_4 fluxes in these soils. The lack of significant N_2O emissions with compost additions could be due to the relatively slow decomposition of the organic matter, slow rate of N release from decomposition, and greater plant N uptake as evident by increased biomass production. I expected the highest rates of N_2O emissions to occur during wet up events when temperatures were relatively high and plant-microbe competition for mineral N was low (Birch 1958, Franzluebbers et al. 2000, Grover et al. 2012). In my laboratory incubation, I was able to stimulate N_2O production for a short time period following soil wetting, but rates of N_2O emissions as a global warming potential were insignificant, particularly relative to the high CO_2 emissions observed from both treatment and control soils. In contrast, soil N_2O emissions from temperate grasslands amended with a range of chemical fertilizers and manures were up to 4900 g N_2O-N ha^{-1} day^{-1} compared with pre-amendment emissions of 4 g N_2O-N ha^{-1} day^{-1} (Rees et al. 2004, Jones et al. 2007).

Effects on above- and belowground net primary production and soil carbon

I observed large increases in ANPP in both grassland types, and consistent trends over time despite a wide range in precipitation among water years. In annual grasslands, ANPP typically increases linearly with increasing precipitation (McCulley et al. 2005, Chou et al. 2008, Wu et al. 2011). I did not observe a significant linear increase in ANPP with rainfall during the study period. Aboveground NPP was greater in amended plots than in controls, findings that are consistent with studies measuring the response of crops to amendments (Edmeades et al. 2003, Badgley et al. 2007) as well as studies of grassland response to N fertilization (Baer et al. 2003, LeBauer and Treseder 2008, Yahdjian et al. 2011). Aboveground NPP at the valley grassland showed a much stronger response than the coastal grassland, even when considering the slight differences in sampling times. Aboveground NPP at the valley site increased by 78 ± 13 % in the amended plots over the 3 years relative to 42 ± 14 % at the coastal site. This could be due to a greater impact of compost amendments on water and N availability at the former site compared to the more mesic coastal grassland (Harpole et al. 2007).

In rangelands, ANPP provides forage for livestock (Asner et al. 2004, Briske et al. 2011). Land management decisions in rangelands often focus on increasing the amount, quality, and sustainability of forage production. These decisions in turn affect the feasibility and sustainability of associated C sequestration or greenhouse gas offset programs. In annual grasslands, ANPP can often be predicted by the timing and magnitude of rainfall and the previous year's production (Hedrick 1948, Heady 1956, Bartolome et al., 1980, Chou et al., 2008). Organic matter additions and other management practices that increase the size of soil C pools are likely to have a positive impact on NPP through increased water holding capacity and N availability. Compared to chemical fertilizers, which provide a short-term pulse of nutrients (Wight and Godfrey 1985, Fauci and Dick 1994), organic matter amendments act as a slow-release fertilizer during decomposition (Sommers 1977), and thus may provide longer-term or sustained increases in aboveground NPP (e.g. Gerzabek et al. 1997, Blair et al. 2006). These effects may last several years. The long-lasting effects of a single application of compost could serve to buffer impacts of decreasing precipitation predicted for some regions with climate change (Stavast et al. 2005, Kowaljow et al. 2010).

Detecting changes in C storage in the bulk soil pool (i.e. without fractionation or other separation techniques) is difficult due to the large variability of the soil C pool in grasslands. In this study, recognizable compost fragments were removed prior to soil C determination to avoid overestimating C pools from amended soils. Regardless, I were able to detect an increase in the bulk soil C pool at the valley site. The increase in soil C was likely derived from a combination of compost incorporation into soil and additions of newly fixed C from NPP. In a three year study monitoring the effects of a one-time application of either inorganic or organic fertilizers in a degraded semiarid rangeland in Patagonia, Kowaljow et al. (2010) observed a similar pattern of increase in soil organic C and enhanced microbial activity. Soil organic C formation and stabilization is promoted by management practices or technologies that increase the quantity of C inputs (Gentile et al. 2011). In rangelands, soil organic content is strongly dependent on root biomass and turnover due to the high belowground allocation of grasses and forbs and longer residence time of root-derived C (Jobbagy and Jackson 2000, Kätterer et al. 2011). In my experiment, root biomass increased significantly in the valley grassland and followed the same trend in the coastal grassland. Application of composted organic material to the soil surface increased both above- and belowground productivity over the three growing seasons, leading to greater annual C inputs from vegetation. Thus, management techniques like composted organic matter additions that increase production have the potential to increase the size of the soil organic C pool over the long-term.

Changes in net ecosystem carbon storage

The control plots from both sites appeared to be losing C. The ecosystem C balance of grasslands is typically highly variable over time and space (Xu and Baldocchi 2004, Novick et al. 2008, Klumpp et al. 2011), but several recent studies have reported C losses from rangeland soils (Bellamy et al. 2005, Schipper et al. 2007, Chou et al. 2008). Chou et al. (2008) reported significant losses of soil organic C for four consecutive years in a Mediterranean annual grassland with and without experimental rainfall additions. The greatest losses occurred in a year with late-season (late spring-summer) rainfall. In annual grasslands, late, warm-season rainfall can stimulate heterotrophic respiration after plants have stopped growing or have senesced, resulting in ecosystem C losses. Carbon storage in grasslands is also sensitive to shifts in vegetation. Continual declines in ecosystem C storage may also be explained by a non-equilibrium status of California grasslands and ongoing gradual declines in soil C following annual grass invasion (Koteen et al. 2011).

Organic matter amendments decreased the rate of C loss in most plots. This was true even when not considering the C added via the amendments, and was due to the stimulation of NPP. Assuming that heterotrophic respiration was 50 % of total soil respiration (see Figure 8 for a range of $R_h : R_a$ scenarios), rates of net ecosystem C storage increased by 25 to 70 % with organic matter amendments with a rate of C sequestration of 51 ± 77 to 333 ± 52 g C/m^2 over the three year study. This value increases to 1,770 ± 142 to 1,383 ± 188 g C/m^2 when considering the amendment C added at the field scale. A full life cycle assessment would be needed to determine the actual greenhouse costs or savings beyond the field scale, but if organic matter amendments are diverted from fates with high greenhouse gas emissions (i.e. landfills, manure slurry ponds) then considerable C offsets may be achieved through this management approach (DeLonge et al. submitted).

My results show that a single application of composted organic matter led to sustained increases in NPP for at least three years, with no sign of diminishing effects. Moreover, increases in plant production significantly offset elevated soil respiration from microbial activity in five out of six paired plots. The amended plot that had lower net C storage relative to its paired control plot also had lower soil volumetric water content, whereas the opposite trend was observed at the other fields (Figure 2). This finding suggests that water limitation has the potential to alter the source-sink potential of annual grasslands (Harpole et al. 2007).

Conclusions

I found that a single application of composted organic matter shifted the C balance of annual grassland ecosystems and resulted in greater C storage. Increases in above- and belowground NPP were observed over three water years, with no obvious sign of a diminishing trend. Enhanced plant productivity was partially offset by elevated soil respiration, but I detected no statistically significant treatment effects on N_2O or CH_4 fluxes. I were able to detect an increase in soil C at the valley site, which was surprising given the large background pool size.

My results have important implications for rangeland management in the context of climate change mitigation. Urban and agricultural green waste is often an important source of greenhouse gas emissions (IPCC 2001). Here I show that an alternative fate for that material can significantly increase NPP and slow rates of ecosystem C losses at the field scale. This approach provides important co-benefits to land owners, such as the sustained increase in forage production measured here. Multi-year field studies are critically needed to explore the potential of ecosystem management to contribute to climate change mitigation. These data are also needed to construct comprehensive and credible life-cycle analyses that explicitly include ecosystem dynamics in C offset protocols.

References

Alluvione, F., C. Bertora, L. Zavattaro, and C. Grignani. 2010. Nitrous Oxide and Carbon Dioxide Emissions Following Green Manure and Compost Fertilization in Corn. Soil Science Society of America Journal 74:384-395.

Asner, G. P., A. J. Elmore, L. P. Olander, R. E. Martin, and A. T. Harris. 2004. Grazing systems, ecosystem responses, and global change. Annual Review of Environment and Resources 29:261-299.

Badgley, C., J. Moghtader, E. Quintero, E. Zakem, M. J. Chappell, K. Aviles-Vazquez, A. Samulon, and I. Perfecto. 2007. Organic agriculture and the global food supply. Renewable Agriculture and Food Systems 22:86-108.

Baer, S. G., J. M. Blair, S. L. Collins, and A. K. Knapp. 2003. Soil resources regulate productivity and diversity in newly established tallgrass prairie. Ecology 84:724-735.

Bai, Z. G., D. L. Dent, L. Olsson, and M. E. Schaepman. 2008. Global Assessment of Land Degradation and Improvement. 1. Identification by remote sensing. Report 2008/01, ISRIC - World Soil Information, Wageningen.

Bartolome, J. W., W. J. Barry, T. Griggs, and P. Hopkinson. 2007. Valley Grassland. Pages 367-393 *in* M. R. Stromberg, J. D. Corbin, and C. M. D'Antonio, editors. California Grasslands Ecology and Management. University of California Press, Berkeley, CA.

Bartolome, J. W., M. C. Stroud, and H. F. Heady. 1980. Influence of natural mulch on forage production on differing California annual range sites. Journal of Range Management 33:4-8.

Beaudette, D. E., and A. T. O'Geen. 2009. Soil-Web: An online soil survey for California, Arizona, and Nevada. Computers & Geosciences 35:2119-2128.

Bellamy, P. H., P. J. Loveland, R. I. Bradley, R. M. Lark, and G. J. D. Kirk. 2005. Carbon losses from all soils across England and Wales 1978-2003. Nature 437:245-248.

Bernal, M. P., M. A. Sanchez-Monedero, C. Paredes, and A. Roig. 1998. Carbon mineralization from organic wastes at different composting stages during their incubation with soil. Agriculture Ecosystems & Environment 69:175-189.

Birch, H. F. 1964. Mineralisation of plant nitrogen following alternate wet and dry conditions. Plant and Soil 20:43-49.

Blair, N., R. D. Faulkner, A. R. Till, and P. R. Poulton. 2006. Long-term management impacts on soil C, N and physical fertility - Part 1: Broadbalk experiment. Soil & Tillage Research 91:30-38.

Bouwman, A. F., I. Fung, E. Matthews, and J. John. 1993. Global analysis of the potential for N_2O production in natural soils. Global Biogeochemical Cycles 7:557-597.

Briske, D. D., J. D. Derner, D. G. Milchunas, and K. W. Tate. 2011. An Evidence-Based Assessment of Prescribed Grazing Practices. Pages 21-74 *in* D. D. Briske, editor. Conservation Benefits of Rangeland Practices. USDA Natural Resource Conservation Service. Allen Press, Inc., Lawrence, Kansas.

Brown, S., A. Dushku, T. Pearson, D. Shoch, J. Winsten, S. Sweet, and J. Kadyszewski. 2004. Carbon supply from changes in management of forest, range, and agricultural lands of California. Winrock International, for the California Energy Commission, PIER Energy-Related Environmental Research, Sacramento, CA, USA.

Cabrera, V. E., L. J. Stavast, T. T. Baker, M. K. Wood, D. S. Cram, R. P. Flynn, and A. L. Ulery. 2009. Soil and runoff response to dairy manure application on New Mexico rangeland. Agriculture Ecosystems & Environment 131:255-262.

Cambardella, C. A., and E. T. Elliott. 1992. Particulate soil organic-matter changes across a grassland cultivation sequence. Soil Science Society of America Journal 56:777-783.

Carbone, M. S., and S. E. Trumbore. 2007. Contribution of new photosynthetic assimilates to respiration by perennial grasses and shrubs: residence times and allocation patterns. New Phytologist 176:124-135.

Cassman, K. G., A. Dobermann, and D. T. Walters. 2002. Agroecosystems, nitrogen-use efficiency, and nitrogen management. Ambio 31:132-140.

Chadwick, D. R., B. F. Pain, and S. K. E. Brookman. 2000. Nitrous oxide and methane emissions following application of animal manures to grassland. Journal of Environmental Quality 29:277-287.

Chou, W. W., W. L. Silver, R. D. Jackson, A. W. Thompson, and B. Allen-Diaz. 2008. The sensitivity of annual grassland carbon cycling to the quantity and timing of rainfall. Global Change Biology 14:1382-1394.

Conant, R. T., K. Paustian, and E. T. Elliott. 2001. Grassland management and conversion into grassland: Effects on soil carbon. Ecological Applications 11:343-355.

Craine, J. M., D. A. Wedin, and F. S. Chapin. 1999. Predominance of ecophysiological controls on soil CO_2 flux in a Minnesota grassland. Plant and Soil 207:77-86.

Dahlgren, R. A., and M. J. Singer. 1994. Nutrient cycling in managed and non-managed oak woodland-grass ecosystems. Final Report: Integrated Hardwood Range Management Program. Land, Air, and Water Resources Paper 100028, Davis, CA.

Dalal, R. C., W. J. Wang, G. P. Robertson, and W. J. Parton. 2003. Nitrous oxide emission from Australian agricultural lands and mitigation options: a review. Australian Journal of Soil Research 41:165-195.

Davidson, E. A. 2009. The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. Nature Geoscience 2:659-662.

Derner, J. D., and G. E. Schuman. 2007. Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. Journal of Soil and Water Conservation 62:77-85.

Edmeades, D. C. 2003. The long-term effects of manures and fertilisers on soil productivity and quality: a review. Nutrient Cycling in Agroecosystems 66:165-180.

Fauci, M. F., and R. P. Dick. 1994. Soil microbial dynamics – short-term and long-term effects of inorganic and organic nitrogen. Soil Science Society of America Journal 58:801-806.

Fierer, N., and J. P. Schimel. 2002. Effects of drying-rewetting frequency on soil carbon and nitrogen transformations. Soil Biology & Biochemistry 34:777-787.

Firestone, M. K., and E. A. Davidson. 1989. Microbiological basis of NO and N_2O production and consumption in soil. Dahlem Workshop Reports Berlin, Germany.

Follett, R. F. 2001. Soil management concepts and carbon sequestration in cropland soils. Soil & Tillage Research 61:77-92.

Franzluebbers, A. J., R. L. Haney, C. W. Honeycutt, H. H. Schomberg, and F. M. Hons. 2000. Flush of carbon dioxide following rewetting of dried soil relates to active organic pools. Soil Science Society of America Journal 64:613-623.

[FRAP] Fire and Resource Assessment Program. 2003. The Changing California: 2003 Forest and Range Assessment. California Department of Forestry and Fire Protection. State of California. The Resources Agency, Sacramento.

Gentile, R., B. Vanlauwe, and J. Six. 2011. Litter quality impacts short- but not long-term soil carbon dynamics in soil aggregate fractions. Ecological Applications 21:695-703.

Gerzabek, M. H., F. Pichlmayer, H. Kirchmann, and G. Haberhauer. 1997. The response of soil organic matter to manure amendments in a long-term experiment at Ultuna, Sweden. European Journal of Soil Science 48:273-282.

Giardina, C. P., and M. G. Ryan. 2002. Total belowground carbon allocation in a fast-growing Eucalyptus plantation estimated using a carbon balance approach. Ecosystems 5:487-499.

Ginting, D., A. Kessavalou, B. Eghball, and J. W. Doran. 2003. Greenhouse gas emissions and soil indicators four years after manure and compost applications. Journal of Environmental Quality 32:23-32.

Goyal, S., S. K. Dhull, and K. K. Kapoor. 2005. Chemical and biological changes during composting of different organic wastes and assessment of compost maturity. Bioresource Technology 96:1584-1591.

Gregorich, E.G., P. Rochette, A.J. VandenBygaart, D.A. Angers. 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil & Tillage Research 83:53-72.

Grover, S. P. P., S. J. Livesley, L. B. Hutley, H. Jamali, B. Fest, J. Beringer, K. Butterbach-Bahl, and S. K. Arndt. 2012. Land use change and the impact on greenhouse gas exchange in north Australian savanna soils. Biogeosciences 9:423-437.

Hall, B., and P. Sullivan. 2001. Alternative soil amendments.*in* USDA Rural Business Cooperative Service, editor. Horticulture Technical Note. National Center for Appropriate Technology, Washington, D.C.

Hanson, P. J., N. T. Edwards, C. T. Garten, and J. A. Andrews. 2000. Separating root and soil microbial contributions to soil respiration: A review of methods and observations. Biogeochemistry 48:115-146.

Harper, C. W., J. M. Blair, P. A. Fay, A. K. Knapp, and J. D. Carlisle. 2005. Increased rainfall variability and reduced rainfall amount decreases soil CO2 flux in a grassland ecosystem. Global Change Biology 11:322-334.

Harpole, W. S., D. L. Potts, and K. N. Suding. 2007. Ecosystem responses to water and nitrogen amendment in a California grassland. Global Change Biology 13:2341-2348.

Heady, H. F. 1956. Changes in a California annual plant community induced by manipulation of natural mulch. Ecology 37:798-812.

Heady, H. F., J. W. Bartolome, M. D. Pitt, G. D. Savelle, and M. C. Stroud. 1992. California prairie. Pages 313-332 *in* R. T. Coupland, editor. Natural Grasslands. Elsevier, New York, NY.

Hedrick, D. W. 1948. The mulch layer of California annual ranges. Jour Range Management 1:22-25.

Herrero, M., P. K. Thornton, P. Gerber, and R. S. Reid. 2009. Livestock, livelihoods and the environment: understanding the trade-offs. Current Opinion in Environmental Sustainability 1:111-120.

Higgins, P. A. T., R. B. Jackson, J. M. Des Rosiers, and C. B. Field. 2002. Root production and demography in a California annual grassland under elevated atmospheric carbon dioxide. Global Change Biology 8:841-850.

IPCC. 2001. Climate Change 2001: Mitigation - Contribution of Working Group III to the Third Assessment Report of the Intergovernmental Panel on Climate Change.*in* B. Metz, O. Davidson, R. Swart, and J. Pan, editors. Cambridge University Press, Cambridge.

Jackson, L. E., R. B. Strauss, M. K. Firestone, and J. W. Bartolome. 1988. Plant and soil-nitrogen dynamics in California annual grassland. Plant and Soil 110:9-17.

Janzen, H. H., C. A. Campbell, R. C. Izaurralde, B. H. Ellert, N. Juma, W. B. McGill, and R. P. Zentner. 1998. Management effects on soil C storage on the Canadian prairies. Soil & Tillage Research 47:181-195.

Jobbagy, E. G., and R. B. Jackson. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications 10:423-436.

Jones, M. B., and A. Donnelly. 2004. Carbon sequestration in temperate grassland ecosystems and the influence of management, climate and elevated CO_2. New Phytologist 164:423-439.

Jones, S. K., R. M. Rees, U. M. Skiba, and B. C. Ball. 2007. Influence of organic and mineral N fertiliser on N_2O fluxes from a temperate grassland. Agriculture Ecosystems & Environment 121:74-83.

Katterer, T., M. A. Bolinder, O. Andren, H. Kirchmann, and L. Menichetti. 2011. Roots contribute more to refractory soil organic matter than above-ground crop residues, as revealed by a long-term field experiment. Agriculture Ecosystems & Environment 141:184-192.

Keller, M., and W. A. Reiners. 1994. Soil atmosphere exchange of nitrous oxide, nitric oxide, and methane under secondary succession of pasture to forest in the Atlantic lowlands of Costa Rica. Global Biogeochemical Cycles 8:399-409.

Klumpp, K., T. Tallec, N. Guix, and J. F. Soussana. 2011. Long-term impacts of agricultural practices and climatic variability on carbon storage in a permanent pasture. Global Change Biology 17:3534-3545.

Koteen, L. E., D. D. Baldocchi, and J. Harte. 2011. Invasion of non-native grasses causes a drop in soil carbon storage in California grasslands. Environmental Research Letters 6.

Kowaljow, E., M. Julia Mazzarino, P. Satti, and C. Jimenez-Rodriguez. 2010. Organic and inorganic fertilizer effects on a degraded Patagonian rangeland. Plant and Soil 332:135-145.

Kuchler, A. W. 1964. Potential vegetation of the conterminous United States. Special Report, no. 36. American Geographical Society, New York.

Kuzyakov, Y. 2010. Priming effects: Interactions between living and dead organic matter. Soil Biology & Biochemistry 42:1363-1371.

Kuzyakov, Y., J. K. Friedel, and K. Stahr. 2000. Review of mechanisms and quantification of priming effects. Soil Biology & Biochemistry 32:1485-1498.

Larney, F. J., and D. A. Angers. 2012. The role of organic amendments in soil reclamation: A review. Canadian Journal of Soil Science 92:19-38.

Le Mer, J., and P. Roger. 2001. Production, oxidation, emission and consumption of methane by soils: A review. European Journal of Soil Biology 37:25-50.

LeBauer, D. S., and K. K. Treseder. 2008. Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology 89:371-379.

Lynch, D. H., R. P. Voroney, and P. R. Warman. 2005. Soil physical properties and organic matter fractions under forages receiving composts, manure or fertilizer. Compost Science & Utilization 13:252-261.

Marshall, S. A., C. P. Campbell, and J. G. Buchanan-Smith. 1998. Herbage biomass and intake of beef cows with calves grazing a grass-legume pasture in southern Ontario. Canadian Journal of Animal Science 78:211-218.

Matthias, A. D., D. N. Yarger, and R. S. Weinbeck. 1978. Numerical evaluation of chamber methods for determining gas fluxes. Geophysical Research Letters 5:765-768.

McCulley, R. L., I. C. Burke, J. A. Nelson, W. K. Lauenroth, A. K. Knapp, and E. F. Kelly. 2005. Regional patterns in carbon cycling across the Great Plains of North America. Ecosystems 8:106-121.

Metcalfe, D. B., M. Williams, L. Aragao, A. C. L. da Costa, S. S. de Almeida, A. P. Braga, P. H. L. Goncalves, and J. D. Silva. 2007. A method for extracting plant roots from soil which facilitates rapid sample processing without compromising measurement accuracy. New Phytologist 174:697-703.

Mosier, A., R. Wassmann, L. Verchot, J. King, and C. Palm. 2004. Methane and nitrogen oxide fluxes in tropical agricultural soils: Sources, sinks and mechanisms. Environment Development and Sustainability 6:11-49.

Nijs, I., J. Roy, J. L. Salager, and J. Fabreguettes. 2000. Elevated CO_2 alters carbon fluxes in early successional Mediterranean ecosystems. Global Change Biology 6:981-994.

Novick, K. A., P. C. Stoy, G. G. Katul, D. S. Ellsworth, M. B. S. Siqueira, J. Juang, and R. Oren. 2004. Carbon dioxide and water vapor exchange in a warm temperate grassland. Oecologia 138:259-274.

Ojima, D.S., K.A. Galvin, and B.L. Turner. 1994. The global impact of land-use change. Bioscience 44:300-304.

Paustian, K., O. Andren, H. H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Van Noordwijk, and P. L. Woomer. 1997. Agricultural soils as a sink to mitigate CO_2 emissions. Soil Use and Management 13:230-244.

Rees, R., S. Jones, R. E. Thorman, I. McTaggart, B. Ball, and U. Skiba. 2004. The effect of organic and mineral nitrogen fertilisers on emissions of NO, N_2O and CH_4 from cut grassland. Pages 268-276 *in* D. J. Hatch, D. R. Chadwick, S. C. Jarvis, and J. A. Roker, editors. Controlling nitrogen flows and losses. 12th Nitrogen Workshop, University of Exeter, UK, 21-24 September 2003.

Sanderman, J., and R. Amundson. 2009. A comparative study of dissolved organic carbon transport and stabilization in California forest and grassland soils. Biogeochemistry 92:41-59.

Schimel, D. S., W. J. Parton, T. G. F. Kittel, D. S. Ojima, and C. V. Cole. 1990. Grassland biogeochemistry – links to atmospheric processes. Climatic Change 17:13-25.

Schipper, L. A., W. T. Baisden, R. L. Parfitt, C. Ross, J. J. Claydon, and G. Arnold. 2007. Large losses of soil C and N from soil profiles under pasture in New Zealand during the past 20 years. Global Change Biology 13:1138-1144.

Schuman, G. E., H. H. Janzen, and J. E. Herrick. 2002. Soil carbon dynamics and potential carbon sequestration by rangelands. Environmental Pollution 116:391-396.

Silver, W. L., A. W. Thompson, M. E. McGroddy, R. K. Varner, J. D. Dias, H. Silva, P. M. Crill, and M. Keller. 2005. Fine root dynamics and trace gas fluxes in two lowland tropical forest soils. Global Change Biology 11:290-306.

Sommers, L. E. 1977. Chemical composition of sewage sludges and analysis of their potential use as fertilizers. Journal of Environmental Quality 6:225-232.

Soussana, J. F., P. Loiseau, N. Vuichard, E. Ceschia, J. Balesdent, T. Chevallier, and D. Arrouays. 2004. Carbon cycling and sequestration opportunities in temperate grasslands. Soil Use and Management 20:219-230.

Stavast, L. J., T. T. Baker, A. L. Ulery, R. P. Flynn, M. K. Wood, and D. S. Cram. 2005. New Mexico blue grama rangeland response to dairy manure application. Rangeland Ecology & Management 58:423-429.

Steenwerth, K. L., L. E. Jackson, F. J. Calderon, M. R. Stromberg, and K. M. Scow. 2002. Soil microbial community composition and land use history in cultivated and grassland ecosystems of coastal California. Soil Biology & Biochemistry 34:1599-1611.

Stromberg, M. R., and J. R. Griffin. 1996. Long-term patterns in coastal California grasslands in relation to cultivation, gophers, and grazing. Ecological Applications 6:1189-1211.

Tang, J. W., and D. D. Baldocchi. 2005. Spatial-temporal variation in soil respiration in an oak-grass savanna ecosystem in California and its partitioning into autotrophic and heterotrophic components. Biogeochemistry 73:183-207.

Trumbore, S. 2000. Age of soil organic matter and soil respiration: Radiocarbon constraints on belowground C dynamics. Ecological Applications 10:399-411.

Vallejo, A., U. M. Skiba, L. Garcia-Torres, A. Arce, S. Lopez-Fernandez, and L. Sanchez-Martin. 2006. Nitrogen oxides emission from soils bearing a potato crop as influenced by fertilization with treated pig slurries and composts. Soil Biology & Biochemistry 38:2782-2793.

Warembourg, F. R., and H. D. Estelrich. 2001. Plant phenology and soil fertility effects on below-ground carbon allocation for an annual (*Bromus madritensis*) and a perennial (*Bromus erectus*) grass species. Soil Biology & Biochemistry 33:1291-1303.

White, R., S. Murray, and M. Rohweder. 2000. Grassland Ecosystems. World Resources Institute, Washington, D.C.

Wight, J. R., and E. B. Godfrey. 1985. Predicting yield response to nitrogen fertilization on northern great plains rangelands. Journal of Range Management 38:238-241.

Wu, Z. T., P. Dijkstra, G. W. Koch, J. Penuelas, and B. A. Hungate. 2011. Responses of terrestrial ecosystems to temperature and precipitation change: a meta-analysis of experimental manipulation. Global Change Biology 17:927-942.

Xu, L. K., D. D. Baldocchi, and J. W. Tang. 2004. How soil moisture, rain pulses, and growth alter the response of ecosystem respiration to temperature. Global Biogeochemical Cycles 18.

Yandjian, L., L. Gherardi, and O. E. Sala. 2011. Nitrogen limitation in arid-subhumid ecosystems: A meta-analysis of fertilization studies. Journal of Arid Environments 75:675-680.

Zhou, X., S. Q. Wan, and Y. Q. Luo. 2007. Source components and interannual variability of soil CO_2 efflux under experimental warming and clipping in a grassland ecosystem. Global Change Biology 13:761-775.

Figures

Figure 1. Daily precipitation data from September 2008 to August 2011 measured at (a) the valley grassland and at (b) the coastal grassland. Air temperature data from (c) valley and (d) coastal grasslands. (e) Paired-plot treatment difference of volumetric soil water content (0 to 30 cm) at the valley grassland. Values above zero indicate greater soil moisture content in amended soils compared to control soils. Black line is mean of all three blocks and gray area is ± 1 standard error. (f) Paired-plot treatment difference in volumetric soil water content (0 to 30 cm) at the coastal grassland. Black line is mean of all two blocks and gray area is ± 1 standard error. Dark gray line is mean of coastal grassland block 2, indicating greater soil water content in the control plot. Paired-plot treatment difference in soil respiration at (g) valley and (h) coastal grasslands. Error bars are ± 1 standard error. Gray block indicates measurements taken prior to amendment. Values above zero indicate greater total soil respiration from amended soils compared to control soils. There were no treatment differences in soil temperature at either grassland ($p > 0.10$).

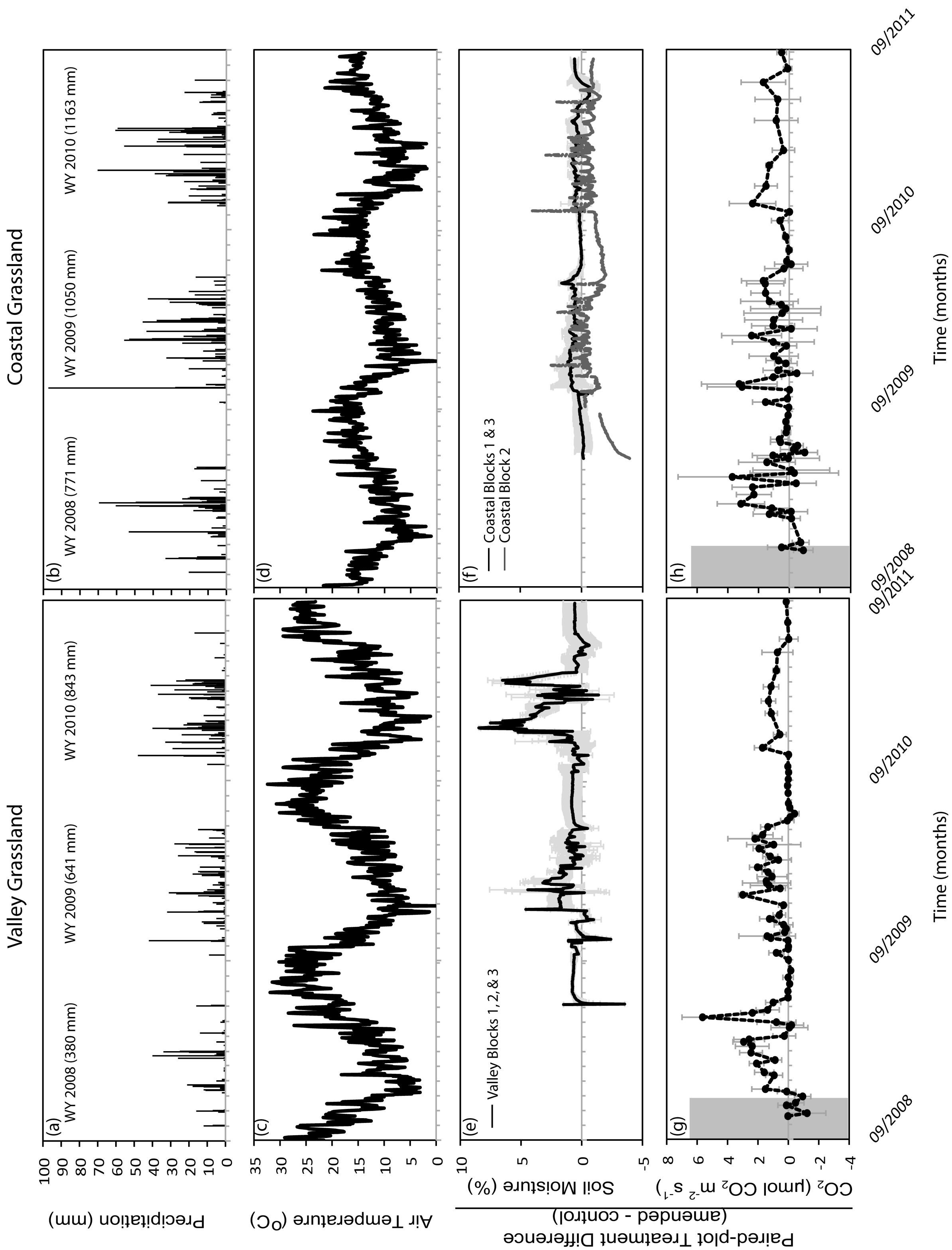

Valley Grassland
Coastal Grassland
(a)
(b)
(c)
(d)
(e)
(f)
(g)
(h)
WY 2008 (380 mm)
WY 2009 (641 mm)
WY 2010 (843 mm)
WY 2008 (771 mm)
WY 2009 (1050 mm)
WY 2010 (1163 mm)
Precipitation (mm)
100
90
80
70
60
50
40
30
20
10
0
Air Temperature (oC)
35
30
25
20
15
10
5
0
Paired-plot Treatment Difference (amended - control)
Soil Moisture (%)
10
5
0
-5
Valley Blocks 1, 2, & 3
Coastal Blocks 1 & 3
Coastal Block 2
CO2 (μmol CO2 m-2 s-1)
8
6
4
2
0
-2
-4
09/2008
09/2009
09/2010
09/2011
Time (months)

Figure 2. Annual total soil respiration for three water years (WY) from the valley and coastal grassland experimental field sites. Bars are means of paired-plot treatment differences with ± 1 standard error bars. Instantaneous fluxes (measured approximately weekly) were linearly interpolated and integrated to obtain mass of C per unit area per year. Values above zero indicate greater total soil respiration from amended soils compared to control soils. * indicates $p < 0.05$.

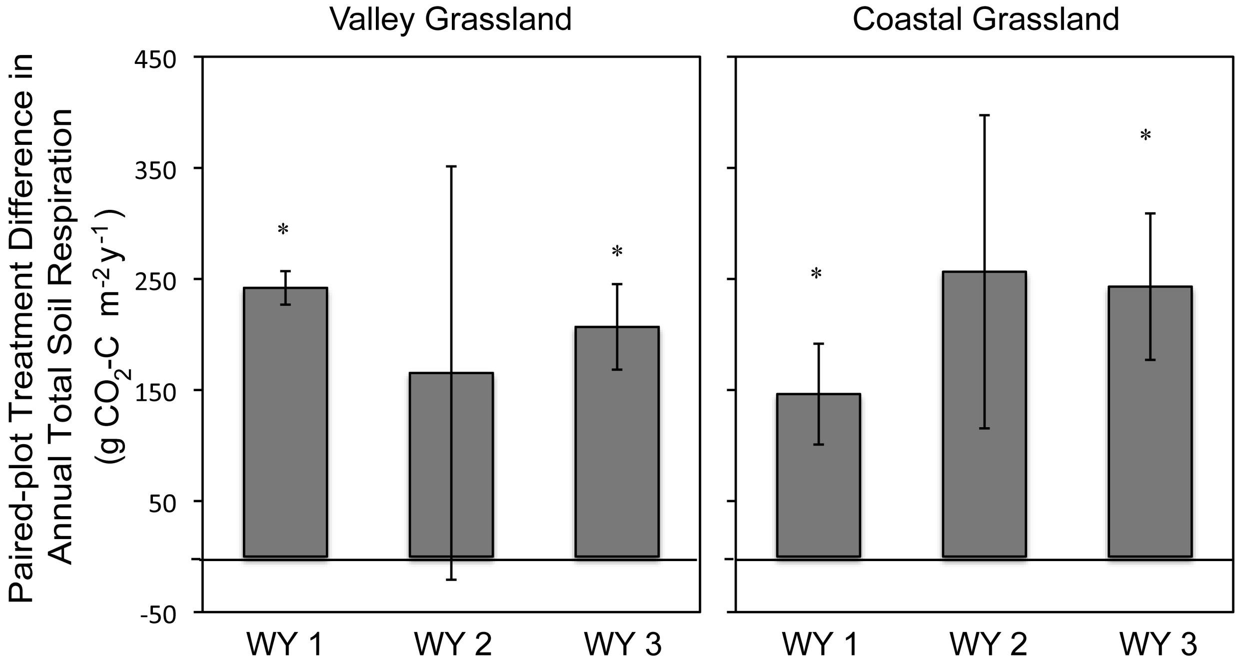


Figure 3. Greenhouse gas emissions during a 30-day laboratory incubation of valley grassland control and amended soils. Soils were incubated in quart-sized mason jars (approximately 200 g) in a growth chamber under typical late summer climate. A fall wet up event was simulated by the addition of 40 mL of distilled water. (a-b) Fluxes of CO_2 and N_2O were significantly great from amended soils compared to control soils for 22 (p = 0.018) and 4 (p = 0.009) days, respectively. Methane fluxes were not significant from zero from both treatments. Symbols are means ± 1 standard error (n = 5). (c) Cumulative N_2O emissions accounted for 0.5 % of total emissions from control soils and 0.8% of total soil greenhouse gas emissions evolved during the 30-day experiment.

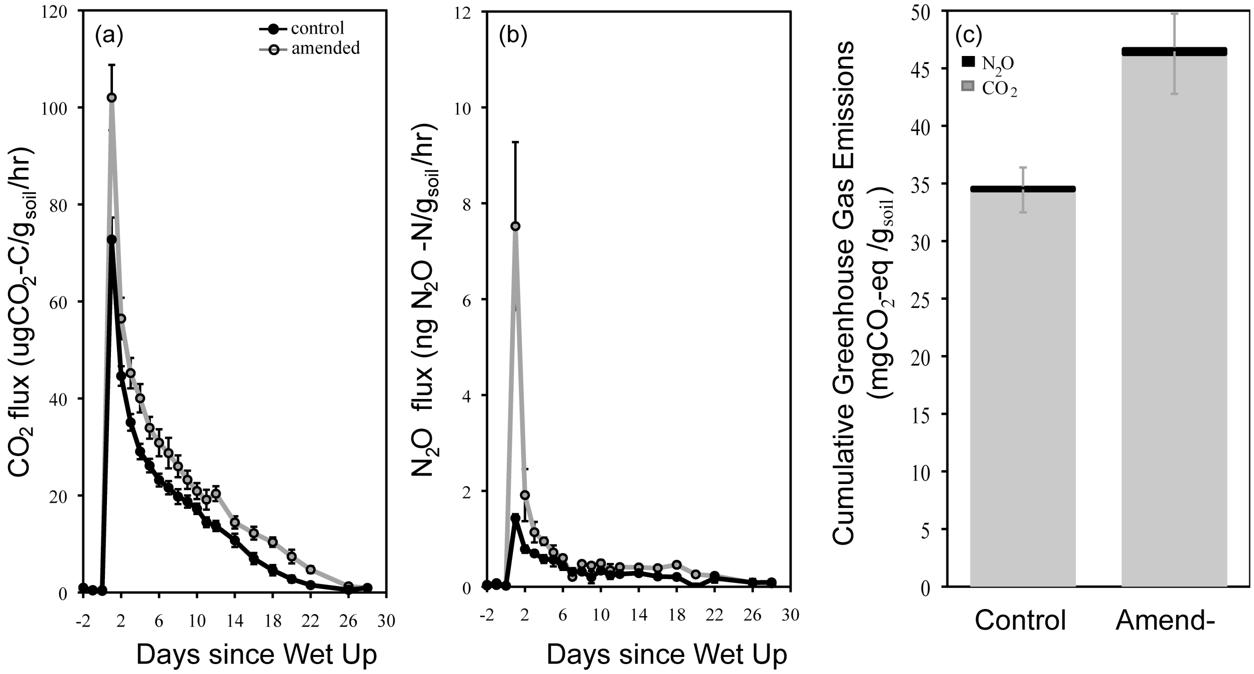


Figure 4. (a) Aboveground net primary production from three water years (WYs) at the valley and coastal grassland experimental field sites. (b) Belowground net primary production (0-20 cm) from three water years (WYs) at the valley and coastal grassland experimental field sites. Root biomass was sampled at 0-10 and 10-20 cm depths. Values are provided in Appendix 1. Bars represent paired-plot treatment differences (mean ± 1 standard error). * indicates $p < 0.05$. ** indicates $p < 0.10$.

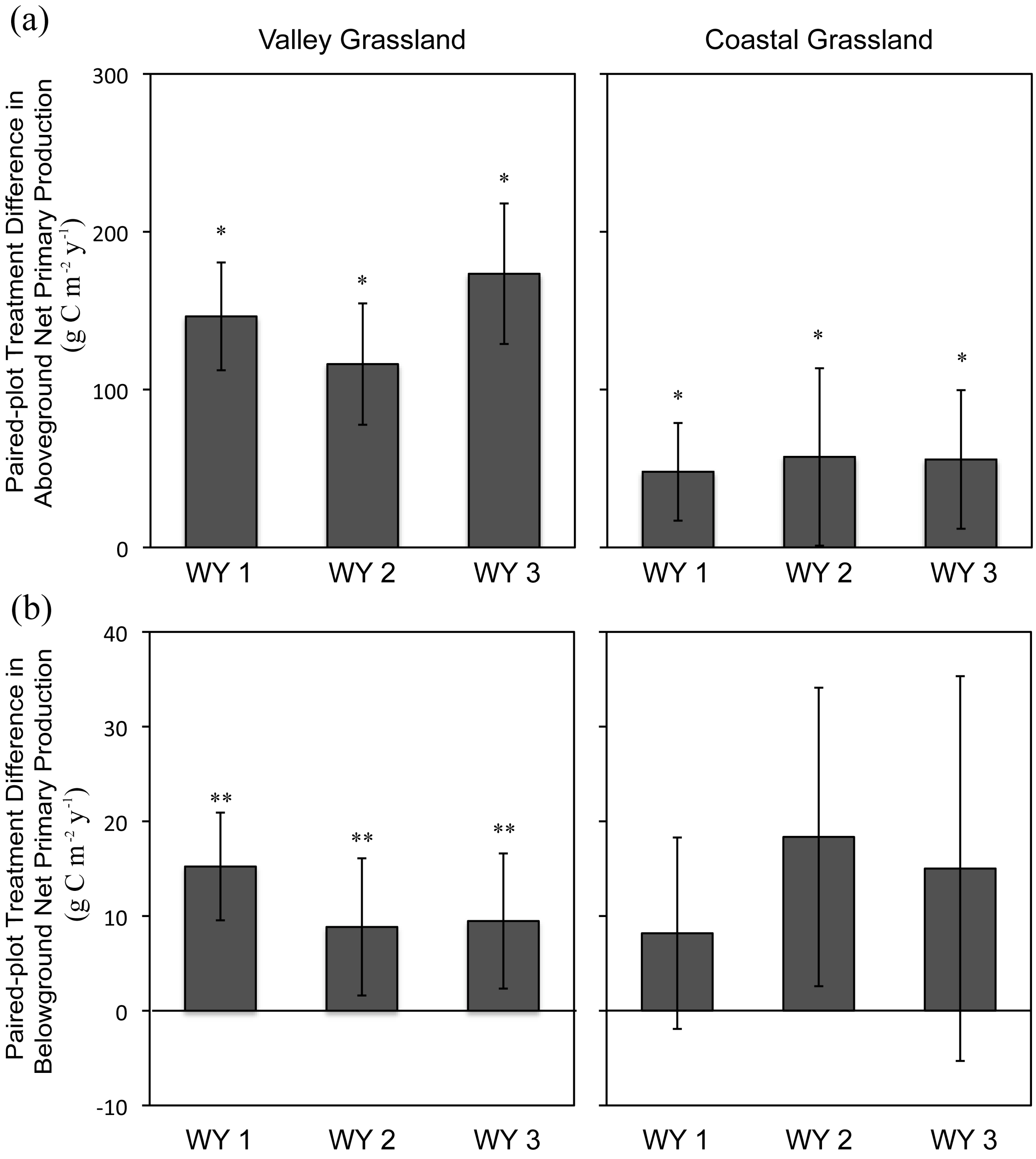


Figure 5. Net primary production from three water years (WYs) at the valley and coastal grassland experimental field sites. The solid portion of the bars shows aboveground biomass and the striped portion shows belowground biomass in control (white) and amended (black) plots. Data shown as mean ± 1 standard error.

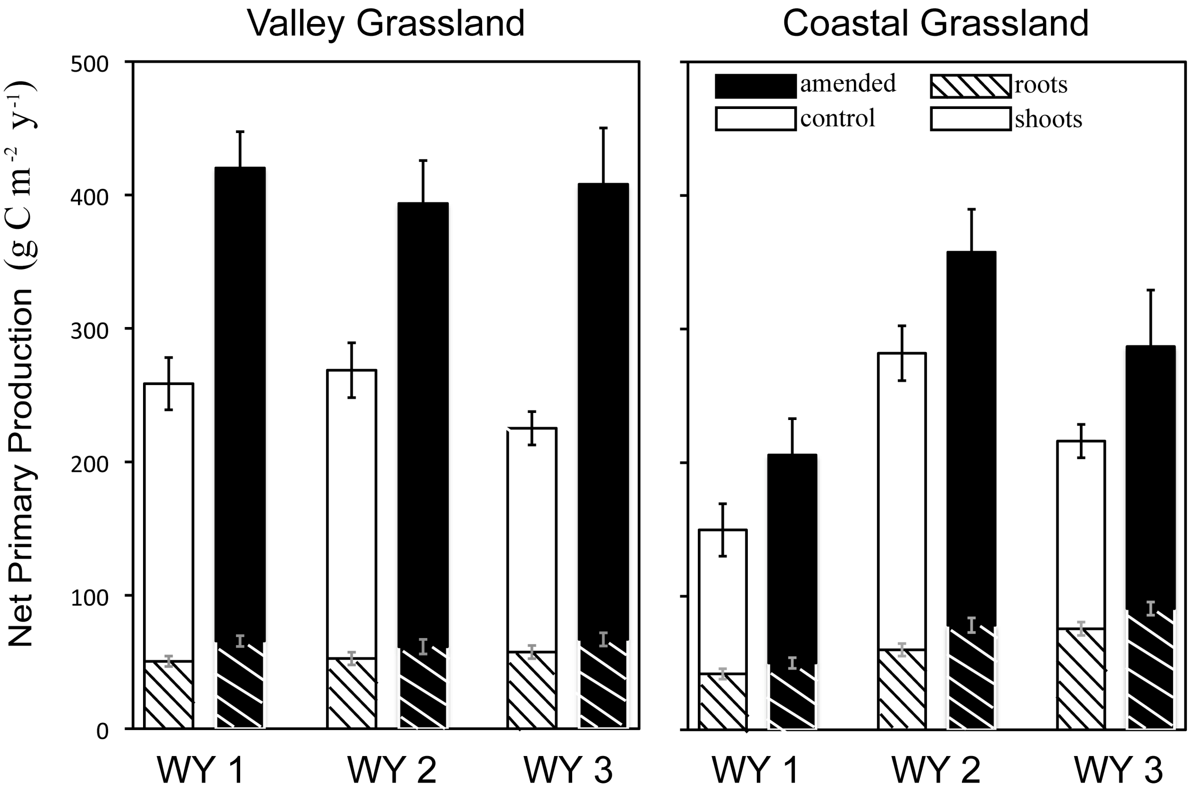


Figure 6. Soil organic C content at 0 to 10 cm data at the valley and grassland sites measured prior to the application (Pre) of composted organic matter and at the end of each water year (WY). Bars are means of paired-plot treatment differences with ± 1 standard error bars.

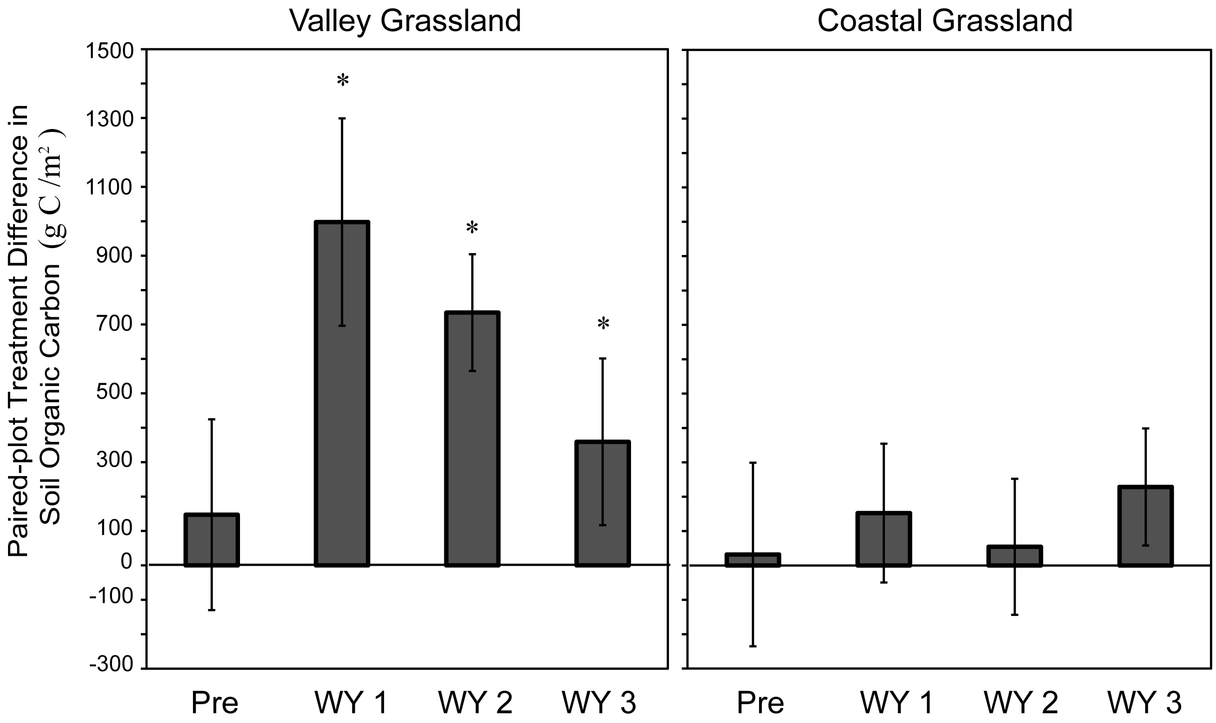
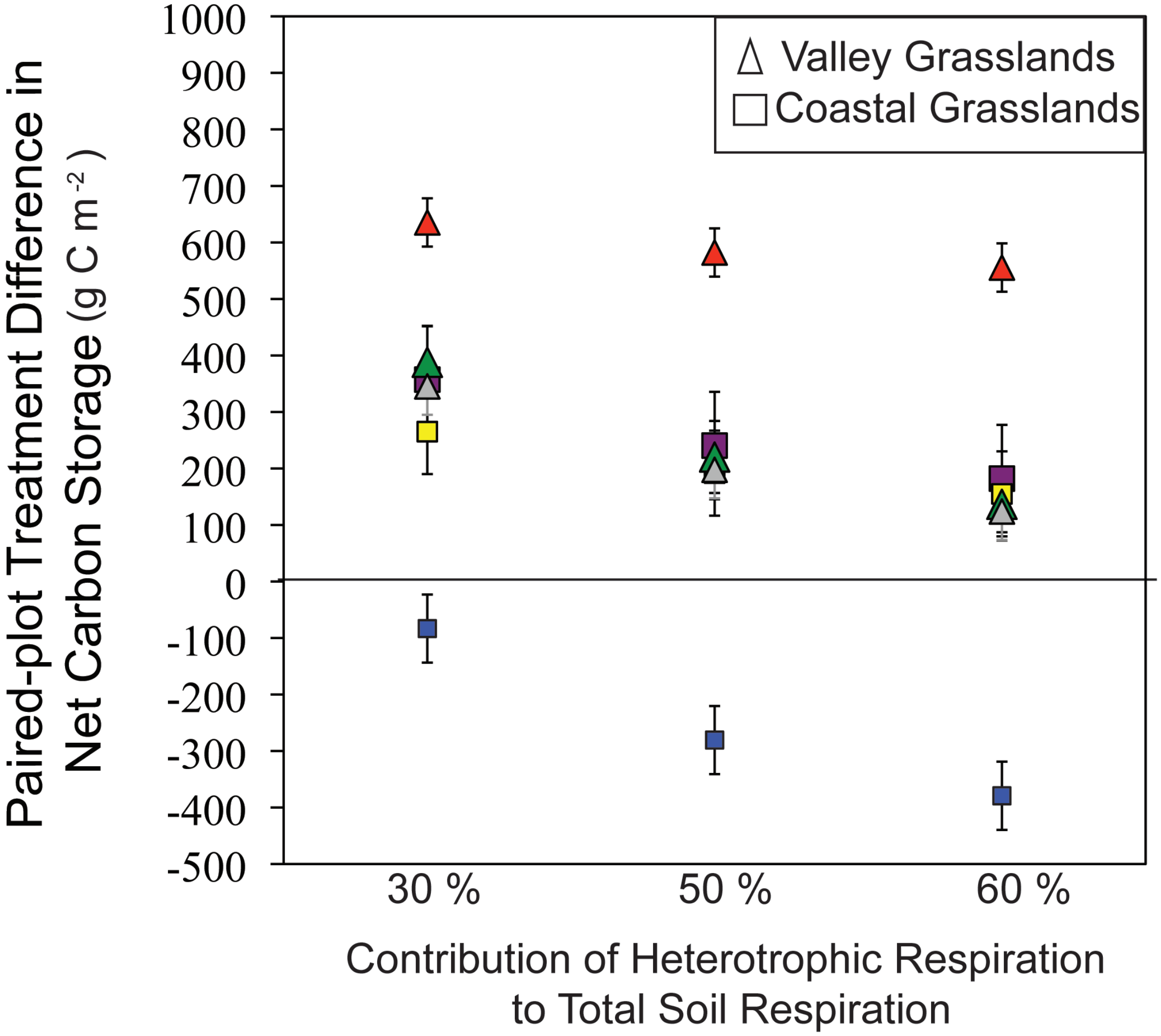


Figure 7. Paired-plot treatment difference in net C storage summed over three water years at the valley (triangles) and coastal (squares) grasslands sites calculated using low (30 %), medium (50 %), and high (60 %) scenarios of the fraction of heterotrophic respiration to total soil respiration. Each paired plot (n = 3 per grassland type) is shown in a unique color. Errors are ± 1 standard error. Values above zero indicate greater C storage in amended plots compared to control plot. Only one paired plot at the coastal grassland showed consistent C loss in the amended plot.



Chapter 2. Impacts of organic amendments on carbon and nitrogen stocks in rangeland soils

Abstract

Rangelands are the dominant global land use type and contain a large global reservoir of soil organic carbon (C). Overgrazing and other intensive management practices have, however, led to significant C losses from rangeland soils. Organic matter amendments have been proposed as a means to enhance rangeland soil C stocks. I measured changes in the amount and chemical quality of C pools in soils following a single application of composted organic material to two annual grasslands. There was a significant and sustained increase in bulk soil C at the valley grassland, and a similar but non-significant trend at the coastal grassland. Physical fractionation of soil in year three revealed a significant increase in C and N in the free and occluded light fractions by 3.31 ± 1.64 and 3.11 ± 1.08 Mg C/ha in the valley and coastal grassland, respectively. I used diffuse reflectance infrared Fourier transform (DRIFT) analysis to explore the chemical composition of the soil fractions. This showed an increase in the ratio of carboxyl and carbonyl functional groups to aliphatic methylene groups suggesting the incorporation of new C and stability of compost C in free- and occluded light fractions. These results suggest that a single application of compost to rangeland soils can increase soil C storage in physically and chemically protected pools and contribute to climate change mitigation.

Introduction

Soil organic matter contributes to soil biological, physical and chemical properties by retaining and supplying plant nutrients, increasing cation exchange capacity, improving soil aggregation, reducing soil erosion, and enhancing water holding capacity (Tisdall and Oades, 1982; Brady and Weil, 2002). Many of the world's rangelands have experienced loss of soil organic matter and C stocks due to poor management, over use, vegetation shifts, or changes in climate (Asner *et al.*, 2004; Bai *et al.*, 2008; USDA, 2012). The large global land area of rangelands with depleted C stocks has focused interest in these ecosystems for C sequestration efforts to help mitigate climate change (Asner *et al.*, 2004; Bai *et al.*, 2008; Lipper *et al.*, 2010; Morgan *et al.*, 2010).

There are considerable opportunities for C sequestration via management in degraded rangelands (Conant *et al.*, 2001; Follett, 2001). To sequester C in soils, management practices must increase C inputs and/or slow the rate of decomposition (Jastrow *et al.*, 2007). Management-induced soil C sequestration has been well studied in agricultural systems in the context of tillage, cropping, organic and synthetic fertilization, and residue incorporation (Paustian *et al.*, 1997; Lal, 2002; Kaiser *et al.*, 2007; Kong *et al.*, 2009; Lugato *et al.*, 2010). In contrast, information regarding the impact of rangeland management on soil C sequestration is much more limited (Conant *et al.*, 2001; Follett, 2001; Lal, 2002; Derner and Schuman, 2007). Compared to cultivated lands, rangelands occupy a larger geographic extent and are generally thought to sequester C at lower rates due to nutrient and water limitations on productivity. Temporal variability in rainfall, low nutrient soils, and periodic fire make managing the balance between plant production and decomposition challenging in rangelands (Conant *et al.*, 2011).

Organic matter amendments have been proposed as a means to increase soil C storage in agricultural and rangelands (Cabrera *et al.*, 2009; Powlson *et al.*, 2012). Organic matter amendments are widely used in agriculture as a means to increase plant production (Eghball and Power, 1999), enhance soil water holding capacity (Gagnon *et al.*, 1998; Zebarth *et al.*, 1999;

Pandey and Shukla, 2006), decrease bulk density (Lynch *et al.*, 2005), and improve soil fertility (Mader *et al.*, 2002). Organic matter amendments can also enhance soil C stocks (Karami *et al.*, 2012; Powlson *et al.*, 2012), and may provide opportunities for greenhouse gas offsets if materials are diverted from high emissions sources (Powlson *et al.*, 2012; DeLonge *et al.*, in review).

It is unclear, however, what the fate of organic matter amendments is in soils. Most organic matter amendments are applied to the soil surface, effectively increasing the size of the fine litter layer. The added organic material can contribute C to labile pool that is easily decomposable and set free again, or can become incorporated into more stable pools. Several mechanisms can act, to varying degrees, to stabilize C, including physical protection in soil aggregation or microbial production of binding agents (Golchin *et al.*, 1994; Six *et al.*, 1998; Gulde *et al.*, 2008). Detecting and interpreting changes to bulk soil and fraction C can be difficult. If surface applied organic matter is not sorted or removed when soils are sampled and analyzed, the contribution of additional plant C inputs to soil C may be greatly exaggerated.

In this study, I investigated the effects of an organic matter amendment (compost) on bulk soil C and N in two rangeland soils. I used physical fractionation procedures and chemical analyses to determine the potential fate of the amendment after a three-year period. I hypothesized that a single application of compost would promote soil C sequestration through a combination of physical and chemical protection mechanisms, and that management-induced changes could be detected over a short (3 year) timeframe. I predicted that detecting changes to bulk soil C would be limited over the three-year experiment, but that physical fractionation procedures and chemical analyses would reveal the location and chemical quality of the incorporated new C inputs into the soil. I further predicted that amended soil fractions would begin to resemble the chemistry of the compost amendment.

Methods

Study sites and experimental design

The study was conducted in valley grasslands at the Sierra Foothill Research and Extension Center (SFREC) in Browns Valley, CA (39.24 °N, 121.30 °W) and coast range grasslands (hereafter referred to as "coastal") in Nicasio, CA (38.06 °N, 122.71 °W). Valley and coastal grasslands represent the two dominant grassland types in California (Kuchler, 1964; Brown *et al.*, 2004): Valley grasslands occupy the hot, dry interior regions of the Central Valley and Sierra Nevada foothills. Coastal grasslands occur adjacent to the coast and experience milder summertime temperatures and greater winter rainfall (Jackson and Bartolome, 2002). At the valley grassland site, annual precipitation averages 730 mm/y (22 y mean), and mean air temperatures range from 2 °C in January to 35 °C in July. Soils are derived from Mesozoic and Franciscan volcanic rock and classified as xeric Inceptisols and Alfisols in the Auburn-Sobrante complex (Beaudette and O'Geen, 2009). At the coastal grassland site, annual precipitation averages 950 mm/y (37 y mean), and mean air temperatures range from 6 °C in January to 20 °C in July. Soils are derived from Franciscan mélange and classified as Mollisols in the Tocaloma-Saurin-Bonnydoon series (Beaudette and O'Geen, 2009). Both grassland types are dominated by non-native annual grasses (e.g. *Avena barbata*, *Festuca perennis*), with native perennial grass species (e.g. *Stipa pulcra*, *Danthonia californica*) also present at the coastal grassland.

Paired plots (25 m x 60 m) were established in three microwatersheds at each site in October 2008. Treatments consisted of one-time organic matter amendments and non-amended controls. An organic-rich soil amendment was added as a single application of commercially available composted organic green waste (Feather River Organics, Marysville, CA) with a N concentration of 1.87 % and a C:N ratio of 11. A thin surface dressing approximately 1.3 cm thick (equivalent to 7 kg/ha, 1.42 kg C/m^2, and 129 g total N/m^2) was applied in December 2008. Previous work at these sites showed that the amendment stimulated C losses through elevated soil respiration, but that losses were offset by larger C gains through increased above- and belowground net primary productivity (Ryals and Silver, *in press*).

Organic matter amendment characteristics

Prior to application, subsamples of the amendment were collected from multiple locations in the compost pile, dried at 65 ºC, finely ground, and analyzed for C and N concentration on a Carlo Erba Elemental Analyzer (Lakewood, NJ). Three 200 g subsamples of material made with the same feedstock were collected from the same compost facility in June 2010 to determine particle size class components. Three 250 g subsamples were separated into the size classes < 1, 1 – 2, 2 – 4.75, and > 4.75 mm through sequential sieving. Each size class was weighed, finely ground, and analyzed for C and N concentration as described above.

Soil texture, pH, and bulk density

Soil texture was determined using the hydrometer method adapted from Gee and Bauder (Gee and Bauder, 1986). Three 0-10 cm soil samples were collected using a 6 cm diameter corer and composited from three transects within each plot (n = 3 per plot). Soils were air dried, and rocks were removed with a 2 mm sieve. Forty g of soil was soaked in and blended with 100 mL sodium hexametaphosphate at a density of 50 g/L. The soil solution was transferred to a 1 L graduated cylinder with distilled water and mixed thoroughly. Hydrometer readings of samples and blanks were recorded at 0.5, 1, and 90 min, and at 24 h. The sand fraction was determined using data collected at 0.5 and 1 min. The clay fraction was determined using 90 min and 24 h time points. The silt fraction was calculated from the remainder after subtracting sand and clay fractions. Soils pH was measured in a ratio of 1:2 soil to water (McLean, 1982).

Soil bulk density was measured to 100 cm depth in 10 cm depth increments in one pit per plot. I carefully excavated 10 cm diameter cores approximately 5 cm back from the face of the pit. Soils at the valley grassland contain a large proportion of rocks, thus all bulk density measurements were corrected for rock volume and mass for all soils. Upon extraction from cores, dry rock masses and volumes were measured and subtracted from initial bulk density calculations (Robertson *et al.*, 1999).

Carbon and nitrogen

Soils were collected prior to compost application and at end of each water year for three years using a 7 cm diameter corer. Sample depths were 0-10, 10-30, 30-50, and 50-100 cm at the coastal grassland and 0-10, 10-30, and 30-50 cm at the valley grassland (n = 9 per depth per plot) where soils were shallower. Roots and identifiable compost fragments were removed by hand. Soils were then air-dried and pulverized with a ball grinder (SPEX Sample Prep Mixer Mill 8000D, Metuchen, NJ). No carbonates were present based on a lack of effervesce upon addition of 4 N HCl to soil. Carbon concentrations were measured in duplicate on a Carlo Erba Elantech elemental analyzer (Lakewood, NJ) using atropine as a standard. Carbon concentrations were converted to content using bulk density values specific to each plot.

Density fractionation

Bulk soil C was partitioned into three fractions using a physical density fractionation procedure (Swanston *et al.*, 2005; Castanha *et al.*, 2008; Marin-Spiotta *et al.*, 2008; Cusack *et al.*, 2011; Appendix 3). This method defines fractions as the low density free light fraction (FLF), the physically protected occluded light fraction (OLF), and mineral-associated heavy fraction (HF; Appendix 4). Physically separated fractions provide a relative index of turnover times and stability of soil C pools (although see (Schmidt *et al.*, 2011). The FLF closely resembles plant material and is particulate organic matter free-floating in the soil matrix. It is generally the most easily accessible by decomposer microorganisms and their extracellular enzymes. The OLF consists of a pool of C that is physically protected in soil aggregates and therefore less accessible to decomposers. The heavy fraction consists of C that is mineral-bound and considered to be the most well protected form of C.

Soils for density fractionation (0-10 cm depth) were collected at the end of WY 3 in 2011 from all treatment plots at both sites using a 7 cm corer. Three cores were composited along three slope transects within each of the twelve plots, for a total of 36 samples (n = 3 per treatment per site). Methods for soil fractionation were adapted from Swanston et al. (Swanston *et al.*, 2005). Rocks were removed by passing soil through a 2 mm sieve, and live roots were extracted by hand. Approximately 20 g was weighed into centrifuge tubes, and 75 mL of sodium polytungstate (NaPT, Na_6 [$H_2W_{12}O_{40}$] TC-Tungsten Compounds, Bavaria, Germany) dissolved to a density of 1.85 g cm^{-3} was added. Samples were gently shaken and centrifuged before aspirating off the light density, floating FLF. The remaining portion of sample was then agitated to disrupt microaggrates with a benchtop mixer (G3U05R, Lightning, New York, NY) at 1700 rpm for 1 min and sonicated in an ice bath for 3 min at 70 % pulse (Branson 450 Sonifier, Danbury, CT). The liberated OLF was then collected via aspiration after centrifugation. The remaining dense HF was rinsed with deionized water and centrifuged until all NaPT was removed. The NaPT was removed from each FLF and OLF by rinsing with deionized water through a 0.8μm polycarbonate membrane filter (Whatman Nuclepore Track Etch Membrane). The light fractions and HF were dried in tins at 65˚C and 105˚C, respectively, until weight stabilized. All fractions were ground using a ball mill and analyzed for C and N as described above. Average mass recovery was 98.3 %.

DRIFT FT-IR spectroscopic analysis

Soil fractions were analyzed for chemical characteristics via diffuse reflectance infrared Fourier transform spectroscopy (DRIFT) (Nguyen *et al.*, 1991). Spectra were recorded with a Fourier transform infrared (FTIR) spectrophotometer (Bruker IFS 66v/S, Ettlingen, Germany) equipped with a Praying Mantis DRIFT apparatus (Harrick Scientific Corporation, Ossining, NY). Samples were prepared by homogenizing a 4.00 mg sample and 40.00 mg spectroscopy-grade KBr (Aldrich Chemical Co., Milwakee, WI) with a mortar and pestle. Finely ground KBr powder was used as a background reference. Absorption spectra were obtained in the mid-IR range of wave numbers between 4000 and 400 cm^{-1} averaged over 32 scans at 4 cm^{-1} resolution and converted to Kubelka-Munk units. Peaks of DRIFT spectra represent overlapping bands of compounds (Nguyen *et al.*, 1991). The peak heights of four prominent peaks (Table 1) were recorded. Peaks at wavenumbers 2910-2930 and 2853 cm^{-1} consisted of aliphatic methylene compounds and were grouped into Band A. Peaks at wavenumbers 1653 and 1400 cm^{-1} consisted of carboxyl compounds with the presence of oxygen and were grouped into Band B, indicative of a greater degree of decomposition compared to Band A. Tangential baseline corrections were made when applicable (Smidt *et al.*, 2002). Peak areas were further normalized to the organic C content of the sample (Djukic *et al.*, 2010).

Statistical analyses

Treatment differences in bulk soil C and N at each grassland site were determined using analysis of variance (ANOVA). Analyses included a blocking effect and were performed separately for valley and coastal sites. Repeated measures ANOVAs were employed to explore treatment effects through time with block, treatment, time, and interactions as model effect factors. ANOVAs were used to identify statistically significant treatment effects on the density fraction C and N mass, concentration and content. DRIFT peak heights and the ratio of band B to band A peak heights were analyzed using ANOVA to detect potential differences in grassland type, fraction, and treatment. Correlations between soil fractions and chemical composition and total soil C content, net primary productivity, and texture were explored using multiple linear regressions.

Statistical tests were performed using JMP 7.0.2 (SAS Institute Inc.). Variables that were not normally distributed were log transformed to meet assumptions for ANOVA. Data are reported either as mean values or treatment differences followed by ± 1 standard error. Error propagation was used when applicable. Statistical significance was determined as $p < 0.05$ unless otherwise noted.

Results

Soil characteristics

Initial soil pH was 6.44 ± 0.05 and 5.92 ± 0.13 at the valley and coastal grassland, respectively ($p < 0.005$). There were no statistically significant differences in initial soil pH across blocks or plots. Organic matter amendments increased pH by 0.2 to 0.3 pH units ($p < 0.05$ at the valley grassland and $p < 0.10$ at the coastal grassland).

The two grassland types had similar soil texture, with no significant difference between treatment plots. Mean percentages of clay, silt, and sand were 16 ± 2, 42 ± 1, and 42 ± 2 % at the valley grassland and 16 ± 2, 40 ± 2, and 43 ± 4 % at the coastal grassland (Fig. 1). At the coastal grassland, one block ("WR 6") contained significantly more sand (54 ± 1 versus 38 ± 2 %) and 54 % less clay (12 ± 1 versus 18 ± 1 %) content than the other two blocks ($p < 0.0001$). Organic matter amendments did not significantly alter soil texture.

Soil bulk density increased with depth in both grassland sites ($p < 0.001$ at both sites; Appendix 2). Soils were shallower at the valley grassland, with bedrock around 50 cm depth versus > 100 cm at the coastal grassland. Soil bulk density was significantly greater at the coastal grassland versus the valley grassland at all depths ($p < 0.10$ for the 0-10 and 10-30 cm depth increments and $p < 0.05$ for 30-50 cm). Organic matter amendments did not significantly alter the bulk density.

Changes in soil carbon and nitrogen

Soil C and N concentrations were similar across sites and declined significantly with depth ($p < 0.0001$). There were no significant differences in C and N concentrations at any depth increment prior to amendment application. By the end of the first water year, there were significant increases in soil C and N concentrations at 0 to 10 cm at the valley grassland ($p < 0.05$ for % C and % N) and a similar, albeit not statistically significant, trend at the coastal grassland. These patterns persisted such that I was able to detect a general positive effect of the treatment on bulk soil C and N concentrations 3 years following a single application of composted organic material, particularly in surface soils (Appendix 2). Soil C and N concentrations exhibited interannual variability typical of annual grasslands (Chou *et al.*, 2008), with stronger interannual variability occurring at the valley grassland.

Soil C and N content of surface (0-10 cm) soils were greater in amended versus control plots by the end of year one and remained elevated by the end of year three at the valley grassland (Figure 2). At the end of year three, the total soil C content of the valley site was 5.3 ± 0.2 kg C m^{-2} and 6.3 ± 0.5 kg C m^{-2} to 50 cm depth for control and amended plots, respectively. Total N at this site was 317 ± 14 and 373 ± 23 g N m^{-2} for the control and amended plots at the valley site. The positive effect on C content was strongest in the first year and declined during the following two years. The positive effect on N content was sustained through all three experimental years following amendment. Similar trends occurred at the coastal site, although differences were not statistically significant. Changes in soil C and N contents were not attributed to the impact of amendment on soil bulk density as there were no significant treatment effects to bulk density at either grassland type (Appendix 2).

Soil density fractions

Density fractions showed different responses to organic matter amendment. At the valley grassland, amended soils had significantly greater FLF mass ($p < 0.05$) but no significant difference in the masses of OLF or HF compared to control soils. At the coastal grassland, amended soils had significantly greater FLF and OLF masses and less HF mass ($p < 0.05$). Carbon concentrations of all fractions showed an increasing, but non-significant trend with amendment at both grassland sites. The HF had lower C concentrations than the FLF and OLF (approximately 1.5 % versus 22 % and 31 %, respectively), but contained most of the soil C because it comprised > 93 % (valley) and > 96 % (coastal) of total soil mass (Table 3).

Organic matter amendments significantly increased the C content of the FLF by 2.56 ± 1.13 and 1.82 ± 0.99 Mg C/ha in the valley and coastal grasslands, respectively (Fig. 3 a and b). The C content of the OLF at the coastal grassland increased by 1.29 ± 1.75 Mg C/ha ($p < 0.05$) with a significant block effect ($p < 0.001$), and the OLF C content at the valley grassland showed a similar but non-significant trend. The response of the HF C content was marginally significant at both grassland sites ($p < 0.1$) but showed opposite effects, with C gains at the valley grassland site. At the coastal grassland, the lower mean C content of amended HF was driven by one paired plot, WR6 ($p = 0.06$). Greater mass and C concentrations of the amended light and heavy fractions were responsible for the observed increase in total soil C at the valley grassland.

Organic matter amendments also increased total soil N (Figure 3 c and d). Nitrogen concentration and content increased significantly in all soil fractions at the valley grassland and in the FLF and OLF at the coastal grassland. There were decreases in the C to N ratios of all three fractions at both grassland types with organic matter amendment (Table 3). Carbon to N ratios significantly decreased from 19.2 ± 0.7 to 14.2 ± 0.2 in valley grassland FLF with amendment, and from 18.8 ± 1.2 to 15.8 ± 0.9 in coastal grassland FLF. Valley grassland OLF C to N ratios significantly decreased from 18.0 ± 0.8 to 15.4 ± 0.6, and from 16.5 ± 0.5 to 14.6 ± 0.1 in coastal grassland OLF. There were significant correlations between C and N concentrations among all soil fractions at both grassland sites ($R^2 = 0.98$, $p < 0.0001$ at both sites). This relationship was most pronounced in the HF ($R^2 = 0.66$ and 0.99 in the valley and coastal grassland sites, respectively; $p < 0.05$, $n = 6$ for each site) and OLF ($R^2 = 0.80$ and 0.61 in the valley and coastal grassland sites, respectively; $p < 0.05$, $n = 6$ for each site). The correlations between C and N concentrations in the FLF were not significant at either grassland. Carbon to N ratios significantly decreased from 19.2 ± 0.7 to 14.2 ± 0.2 in valley grassland FLF with amendment, and from 18.8 ± 1.2 to 15.8 ± 0.9 in coastal grassland FLF. Valley grassland OLF C to N ratios significantly decreased from 18.0 ± 0.8 to 15.4 ± 0.6, and from 16.5 ± 0.5 to 14.6 ± 0.1 in coastal grassland OLF.

Nearly two-thirds of all soil organic C was stored in the HF at the coastal grassland. In contrast, more than half of the soil organic C was stored in the light fractions at the valley grassland. The stock of C and N in the FLF was significantly greater at in the valley versus coastal grassland soils ($p < 0.001$), whereas significantly more C and N were stored in the HF at the coastal grassland ($p < 0.001$). Texture was correlated with C and N distribution across fractions at the coastal grassland. Carbon and N content of the OLF and HF were positively correlated to clay content ($R^2 = 0.40$ and 0.37 for OLF C and N content; $R^2 = 0.57$ and 0.58 for HF C and N content; $p < 0.01$ in all cases). All three fractions were negatively correlated with sand content ($R^2 = 0.30$, 0.54, and 0.60 for FLF, OLF, and HF C content; $R^2 = 0.30$, 0.52, and 0.64 for FLF, OLF, and HF N content; $p < 0.05$ in all cases).

DRIFT spectra of soil density fractions

The DRIFT spectra revealed differences in C chemistry between grassland sites and soil fractions. In all soil fractions, there were two bands at wavenumbers 2920 and 2853 cm^{-1} representing aliphatic methylene (CH_2 and CH_3) groups. The sum of these two band intensities is referred to here as band A. Carbonyl and carboxyl were represented by bands at wavenumbers 1653 and 1400 cm^{-1}, and the combined intensities referred to here as band B. There were qualitative similarities in peak intensities between the FLF and OLF and a noticeable diminution of absorbance intensities of the aliphatic methylene groups in the HF (Figure 4). Analyzing the ratio of bands B:A provides an indication of decomposition, where a higher B:A ratio suggests greater degradation of organic matter (Kaiser *et al.*, 2007). The B:A ratio of the control soil FLF and OLF from the valley grassland was significantly greater than that of the coastal grassland ($p < 0.02$, $p = 0.07$ for the fractions, respectively). The B:A ratio of control soil HF was significantly greater than the B:A ratio of the control soil FLF and OLF ($p < 0.05$). This suggests that the HG has less aliphatic compounds and more of carbonyl and carboxyl compounds than the FLF and OLF (Ouatmane *et al.*, 2000). The amended soils has a diminished intensity of band A groups and more intense peaks of band B groups, although the variability was high (Fig. 5). As a result, the mean B:A ratio of the FLF and OLF in amended soil was greater than that of control (Fig. 6). The B:A ratio of these fractions showed similarities to the compost amendment, suggesting some incorporation of the amendment into both the FLF and physically-protected OLF in the 0-10 cm soil depth.

Discussion

Changes to soil C and N

Soil C pools in grasslands tend to large and spatially variable, making it difficult to detect changes over the short-term (< 10 years). Organic matter amendments add C to the soil surface, but the fate that C is poorly understood. The material can be lost via decomposition, surface runoff, or wind erosion prior to integration into soil. In this study, I carefully removed recognizable compost litter fragments, and thus focused on the material that was incorporated into soil organic matter. I hypothesized that I would not be able to detect changes in bulk soil C and N using this approach. However, I saw significant increases in both C and N in the bulk soil at the valley site with amendments and a similar trend at the coastal grassland.

Increased soil C storage has been consistently reported in the few existing short-term amendment experiments conducted on pasturelands and rangelands. In a four-year study, soil C increased by up to 9.7 Mg C/ha in a perennial grassland after receiving compost amendments for two consecutive years in combination with seeding of legumes (Lynch *et al.*, 2005). Similarly, manure amendments to rangeland in New Mexico increased soil organic matter by 34 % after only three years (Cabrera *et al.*, 2009). The application of composted biosolids to a semi-arid rangeland in Colorado resulted in significant increases in soil C, as well as increases in plant productivity, microbial biomass, and net N mineralization rates (Pascual *et al.*, 1999; Sullivan *et al.*, 2006; Fernandez *et al.*, 2007b; Ippolito *et al.*, 2010; Kowaljow *et al.*, 2010), the effects of which were evident for at least 14 years following a single application. (Ippolito *et al.*, 2010).

Soil density fractions

Soil density fractions can provide an early indication of management-induced changes to soil C storage (He *et al.*, 2008). Using a physical fractionation procedure, I detected greater soil C storage in the amended plots, particularly in the FLF and OLF. After three years, compost amendment increased soil C by 3.31 ± 1.64 and 3.11 ± 1.08 Mg C/ha, or by 26 and 37 %, in the FLF and OLF at the valley and coastal grassland, respectively. The C sequestration rates estimated from the density fractionation procedure are similar to estimates based on the balance of annual C gains (*i.e.* above- and belowground net primary production) and losses (*i.e.* heterotrophic soil respiration) at these sites (Ryals and Silver, *in press*). Changes to the HF were marginally significant, with no change or slight gains in HF C content in all but one paired-plot. This plot (WR 6) had lower clay content and lower soil moisture (Ryals and Silver, in press) than all the other coastal plots. It is possible that the differences in the HF C content between treatment and control plots in this block represent pre-existing differences in soil physical properties (Burke *et al.*, 1989).

Stabilization of soil C in amended agricultural soils has been attributed to the promotion of physical protection of soil organic C in soil aggregates (Six *et al.*, 2000; Chivenge *et al.*, 2007; Majumder *et al.*, 2010; Majumder and Kuzyakov, 2010), and greater physico-chemical association of organic C with clay and silt (Razafimbelo *et al.*, 2008). My finding that C content was greater in OLF of amended soils is consistent with reports of improved soil aggregation across a range of soil conditions (Pagliai *et al.*, 1981; Aggelides and Londra, 2000; Pagliai *et al.*, 2004). When occluded in soil aggregates, soil organic C is generally less accessible to decomposer organisms and thus more protected from loss to the atmosphere (Dungait *et al.*, 2012).

The additional C found in the FLF and OLF likely originated from two possible sources. Some of the additional soil C in amended plots was due to incorporation of the C from the amendment material. A second source for the additional C found in amended soil may be above- and belowground plant litter. Many grasslands are limited by N and water availability (Dukes *et al.*, 2005; Harpole *et al.*, 2007; Chou *et al.*, 2008). Studies of inorganic N fertilization to rangelands have consistently shown a positive response of plant production to fertilization, which is often accompanied by increased soil C storage (Derner and Schuman, 2007). Compost amendments in this study added 129 g total N/m^2. Compost N is complexed in organic matter and less immediately available for plant uptake compared to inorganic N fertilizers or manure. However, net primary productivity generally increases with compost additions (Lynch *et al.*, 2004; Cabrera *et al.*, 2009; McFarland *et al.*, 2010). Ryals and Silver (*in press*) measured cumulative increases of 2.0 ± 0.8 Mg C/ha and 4.7 ± 0.7 Mg C/ha in net primary production (sum of above- and belowground production) at the coastal and valley grassland, respectively, for three years following a single compost amendment. These increases in C inputs to the soil, combined with presence of soil C in the physically-protected OLF, suggest potential for long-term C storage (Carrington *et al.*, 2012). During the three-year study period, the positive plant production response was consistent and showed no signs of diminishing over time.

The increase in N concentration was greater relative to the increase in C concentration of the soil fractions of the amended plots, the leading to a decrease in the C:N ratio of the light fractions with amendment. Narrower C:N ratios suggest a greater degree of microbial transformation (Rodionov *et al.*, 2000; Six *et al.*, 2001). Therefore, the stability of C stored in the light fractions may differ between treatments. Composted organic materials tend to be highly recalcitrant, and may exhibit prolonged C storage in non-mineral fractions relative to fresh plant litter (Lynch *et al.*, 2006).

The notion of inherent recalcitrance of organic material is currently being debated (Koegel-Knabner *et al.*, 2008; Marschner *et al.*, 2008; Schmidt *et al.*, 2011; Dungait *et al.*, 2012). However, by combining chemical characterization with physical fractionation, I can assess the degree of physical and chemical protection of organic matter in soils. Qualitative and quantitative FTIR analyses have been used to infer the potential difference in stability among soil fraction or across treatment effects (Gerzabek *et al.*, 1997; Kaiser *et al.*, 2007; Simonetti *et al.*, 2011). In this study, DRIFT FTIR on soil fractions revealed marked differences in the chemical composition of the two light fractions and the mineral-bound HF in control soils at both grassland sites. Light fractions consisted of greater absorbance intensities of aliphatic methylene compounds, lower absorbance intensities of carboxyl and carbonyl compounds, and lower B:A ratios. These patterns are consistent with the HF having undergone a greater degree of decomposition compared to the FLF and OLF (Poirier *et al.*, 2005). Organic matter amendments altered the chemical composition of the FLF and OLF. B:A ratios of the light fractions were greater in amended versus control soils. At both grassland types, the ratio of BA of the FLF and OLF began to reflect the ratio of the compost, suggesting incorporation of the amendment into the light fractions.

Implications for long-term C storage

Studies of organic amendment to rangeland soils are scarce compared to those in croplands. My results show that a single application of compost to two grassland sites results in soil C sequestration rates of over 3 Mg C/ha over three years with concomitant increases in soil N. At the Rothamsted Broadbalk Experiment, one of the longest running agricultural experiments, farmyard manure was added to winter-wheat cropping fields annually for over 150 years (Blair *et al.*, 2006). They found that the most rapid rates of soil C gain occurred during the first 20 years of application (1.0 Mg C ha^{-1} yr^{-1}), and then slowed as a new equilibrium approached (0.2 Mg C ha^{-1} yr^{-1} during the last 20 years) (Powlson *et al.*, 2012).

It has been hypothesized that soils have a maximum C storage potential (*i.e.* C-saturation capacity) determined by physicochemical properties of the soil (Six *et al.*, 2002). This has not been determined for the valley or coastal grasslands studied here. However, several lines of evidence suggest that these soils are considerably below their C-saturation level. The shift from perennial-dominated grasslands to annual grasslands in California since the 1800s may have caused a decline in soil C stocks by approximately 40 Mg C/ha in the top 50 cm of soil (Koteen *et al.*, 2011), which appears to be declining still (Chou *et al.*, 2008). Thus, there is theoretical potential for these systems to at least regain the soil C that has been lost to the atmosphere. In a broad survey of 34 coastal rangelands, Silver et al. (*in prep*) measured a mean difference of 24 Mg C/ha in surface soils (0 – 10 cm depth) that had received organic matter amendments versus soils that had received no organic matter amendments, and approximately half of the additional C was stored in the OLF. Soil organic C stocks vary widely across grasslands in California, ranging from 28 to 137 Mg C/ha in the top 50 cm of soil (Silver *et al.*, 2010). The background soil C content of the valley and coastal grasslands fall well below the regional average at approximately 53 and 57 Mg C/ha to 50 cm depth, once again supporting the hypothesis that these soils have considerable capacity for additional C storage and are not near saturation. Incorporation of new C into the physically-protected fraction in combination with previous research showing sustained boost in production suggests that compost amendments have potential for long-term C sequestration.

Conclusions

I found that a single application of composted green waste induced increases in soil C and N over three years. The addition C and N was stored in the FLF and physically-protected OLF, and was likely derived from a combination of direct inputs of compost C and increases in plant production. The C:N ratio and the proportion of intensities of C=O functional groups to C-H functional group increased in the light fractions of the amended soils, suggesting that greater C stabilization with amendment. My results indicate that grassland soils may be highly responsive to C sequestration management strategies. Long-term studies and monitoring in grasslands are critically needed to explore the potential for management-induced soil C sequestration, stabilization, and potential saturation.

References

Aggelides, S.M., Londra, P.A., 2000. Effects of compost produced from town wastes and sewage sludge on the physical properties of a loamy and a clay soil. Bioresource Technology 71, 253-259.

Asner, G.P., Elmore, A.J., Olander, L.P., Martin, R.E., Harris, A.T., 2004. Grazing systems, ecosystem responses, and global change. Annual Review of Environment and Resources 29, 261-299.

Bai, Z.G., Dent, D.L., Olsson, L., Schaepman, M.E., 2008. Global Assessment of Land Degradation and Improvement. 1. Identification by remote sensing. World Soil Information, Wageningen.

Beaudette, D.E., O'Geen, A.T., 2009. Soil-Web: An online soil survey for California, Arizona, and Nevada. Computers & Geosciences 35, 2119-2128.

Blair, N., Faulkner, R.D., Till, A.R., Poulton, P.R., 2006. Long-term management impacts on soil C, N and physical fertility - Part 1: Broadbalk experiment. Soil & Tillage Research 91, 30-38.

Brady, N.C., Weil, R.R., 2002. The Nature and Properties of Soils. Pearson Education, Inc., Upper Saddle River, New Jersey.

Brown, S., Dushku, A., Pearson, T., Shoch, D., Winsten, J., Sweet, S., Kadyszewski, J., 2004. Carbon supply from changes in management of forest, range, and agricultural lands of California. Winrock International, for the California Energy Commission, PIER Energy-Related Environmental Research, Sacramento, CA, USA, p. 114.

Burke, I.C., Yonker, C.M., Parton, W.J., Cole, C.V., Flach, K., Schimel, D.S., 1989. Texture, climate, and cultivation effects on soil organic-matter content in US grassland soils. Soil Science Society of America Journal 53, 800-805.

Cabrera, V.E., Stavast, L.J., Baker, T.T., Wood, M.K., Cram, D.S., Flynn, R.P., Ulery, A.L., 2009. Soil and runoff response to dairy manure application on New Mexico rangeland. Agriculture Ecosystems & Environment 131, 255-262.

Carrington, E.M., Hernes, P.J., Dyda, R.Y., Plante, A.F., Six, J., 2012. Biochemical changes across a carbon saturation gradient: Lignin, cutin, and suberin decomposition and stabilization in fractionated carbon pools. Soil Biology & Biochemistry 47, 179-190.

Castanha, C., Trumbore, S., Amundson, R., 2008. Methods of separating soil carbon pools affect the chemistry and turnover time of isolated fractions. Radiocarbon 50, 83-97.

Chivenge, P.P., Murwira, H.K., Giller, K.E., Mapfumo, P., Six, J., 2007. Long-term impact of reduced tillage and residue management on soil carbon stabilization: Implications for conservation agriculture on contrasting soils. Soil & Tillage Research 94, 328-337.

Chou, W.W., Silver, W.L., Jackson, R.D., Thompson, A.W., Allen-Diaz, B., 2008. The sensitivity of annual grassland carbon cycling to the quantity and timing of rainfall. Global Change Biology 14, 1382-1394.

Conant, R.T., Ogle, S.M., Paul, E.A., Paustian, K., 2011. Measuring and monitoring soil organic carbon stocks in agricultural lands for climate mitigation. Frontiers in Ecology and the Environment 9, 169-173.

Conant, R.T., Paustian, K., Elliott, E.T., 2001. Grassland management and conversion into grassland: Effects on soil carbon. Ecological Applications 11, 343-355.

Cusack, D.F., Silver, W.L., Torn, M.S., McDowell, W.H., 2011. Effects of nitrogen additions on above- and belowground carbon dynamics in two tropical forests. Biogeochemistry 104, 203-225.

DeLonge, M., Ryals, R., Silver, W.L. in review. Carbon sequestration potential and greenhouse gas dynamics of managed grasslands. Ecosystems.

Derner, J.D., Schuman, G.E., 2007. Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. Journal of Soil and Water Conservation 62, 77-85.

Djukic, I., Zehetner, F., Tatzber, M., Gerzabek, M.H., 2010. Soil organic-matter stocks and characteristics along an Alpine elevation gradient. Journal of Plant Nutrition and Soil Science 173, 30-38.

Dukes, J.S., Chiariello, N.R., Cleland, E.E., Moore, L.A., Shaw, M.R., Thayer, S., Tobeck, T., Mooney, H.A., Field, C.B., 2005. Responses of grassland production to single and multiple global environmental changes. Plos Biology 3, 1829-1837.

Dungait, J.A.J., Hopkins, D.W., Gregory, A.S., Whitmore, A.P., 2012. Soil organic matter turnover is governed by accessibility not recalcitrance. Global Change Biology 18, 1781-1796.

Eghball, B., Power, J.F., 1999. Composted and noncomposted manure application to conventional and no-tillage systems: Corn yield and nitrogen uptake. Agronomy Journal 91, 819-825.

Fernandez, J.M., Plaza, C., Hernandez, D., Polo, A., 2007. Carbon mineralization in an and soil amended with thermally-dried and composted sewage sludges. Geoderma 137, 497-503.

Follett, R.F., 2001. Soil management concepts and carbon sequestration zin cropland soils. Soil & Tillage Research 61, 77-92.

Gagnon, B., Simard, R.R., Goulet, M., Robitaille, R., Rioux, R., 1998. Soil nitrogen and moisture as influenced by composts and inorganic fertilizer rate. Canadian Journal of Soil Science 78, 207-215.

Gee, G.W., Bauder, J.W., 1986. Particle-size analysis *in* Methods of soil analysis. Part 1. Physical and mineralogical methods. .

Gerzabek, M.H., Pichlmayer, F., Kirchmann, H., Haberhauer, G., 1997. The response of soil organic matter to manure amendments in a long-term experiment at Ultuna, Sweden. European Journal of Soil Science 48, 273-282.

Golchin, A., Oades, J.M., Skjemstad, J.O., Clarke, P., 1994. Soil-structure and carbon cycling. Australian Journal of Soil Research 32, 1043-1068.

Gulde, S., Chung, H., Amelung, W., Chang, C., Six, J., 2008. Soil carbon saturation controls labile and stable carbon pool dynamics. Soil Science Society of America Journal 72, 605-612.

Harpole, W.S., Potts, D.L., Suding, K.N., 2007. Ecosystem responses to water and nitrogen amendment in a California grassland. Global Change Biology 13, 2341-2348.

He, Y., Xu, Z., Chen, C., Burton, J., Ma, Q., Ge, Y., Xu, J., 2008. Using light fraction and macroaggregate associated organic matters as early indicators for management-induced changes in soil chemical and biological properties in adjacent native and plantation forests of subtropical Australia. Geoderma 147, 116-125.

Ippolito, J.A., Barbarick, K.A., Paschke, M.W., Brobst, R.B., 2010. Infrequent composted biosolids applications affect semi-arid grassland soils and vegetation. Journal of Environmental Management 91, 1123-1130.

Jackson, R.D., Bartolome, J.W., 2002. A state-transition approach to understanding nonequilibrium plant community dynamics in Californian grasslands. Plant Ecology 162, 49-65.

Jastrow, J.D., Amonette, J.E., Bailey, V.L., 2007. Mechanisms controlling soil carbon turnover and their potential application for enhancing carbon sequestration. Climatic Change 80, 5-23.

Kaiser, M., Ellerbrock, R.H., Gerke, H.H., 2007. Long-term effects of crop rotation and fertilization on soil organic matter composition. European Journal of Soil Science 58, 1460-1470.

Karami, A., Homaee, M., Afzalinia, S., Ruhipour, H., Basirat, S., 2012. Organic resource management: Impacts on soil aggregate stability and other soil physico-chemical properties. Agriculture Ecosystems & Environment 148, 22-28.

Koegel-Knabner, I., Guggenberger, G., Kleber, M., Kandeler, E., Kalbitz, K., Scheu, S., Eusterhues, K., Leinweber, P., 2008. Organo-mineral associations in temperate soils: Integrating biology, mineralogy, and organic matter chemistry. Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 171, 61-82.

Kong, A.Y.Y., Fonte, S.J., van Kessel, C., Six, J., 2009. Transitioning from standard to minimum tillage: Trade-offs between soil organic matter stabilization, nitrous oxide emissions, and N availability in irrigated cropping systems. Soil & Tillage Research 104, 256-262.

Koteen, L.E., Baldocchi, D.D., Harte, J., 2011. Invasion of non-native grasses causes a drop in soil carbon storage in California grasslands. Environmental Research Letters 6.

Kowaljow, E., Julia Mazzarino, M., Satti, P., Jimenez-Rodriguez, C., 2010. Organic and inorganic fertilizer effects on a degraded Patagonian rangeland. Plant and Soil 332, 135-145.

Kuchler, A.W., 1964. Potential vegetation of the conterminous United States. Special Report, no. 36. American Geographical Society, New York.

Lal, R., 2002. Soil carbon dynamics in cropland and rangeland. Environmental Pollution 116, 353-362.

Lipper, L., Dutilly-Diane, C., McCarthy, N., 2010. Supplying Carbon Sequestration From West African Rangelands: Opportunities and Barriers. Rangeland Ecology & Management 63, 155-166.

Lugato, E., Simonetti, G., Morari, F., Nardi, S., Berti, A., Giardini, L., 2010. Distribution of organic and humic carbon in wet-sieved aggregates of different soils under long-term fertilization experiment. Geoderma 157, 80-85.

Lynch, D.H., Voroney, R.P., Warman, P.R., 2004. Nitrogen availability from composts for humid region perennial grass and leemne-grass forage production. Journal of Environmental Quality 33, 1509-1520.

Lynch, D.H., Voroney, R.P., Warman, P.R., 2005. Soil physical properties and organic matter fractions under forages receiving composts, manure or fertilizer. Compost Science & Utilization 13, 252-261.

Lynch, D.H., Voroney, R.P., Warman, P.R., 2006. Use of 13C and 15N natural abundance techniques to characterize carbon and nitrogen dynamics in composting and in compost-amended soils. Soil Biology & Biochemistry 38, 103-114.

Mader, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P., Niggli, U., 2002. Soil fertility and biodiversity in organic farming. Science 296, 1694-1697.

Majumder, B., Joerg, R., Yakov, K., 2010. Effects of aggregation processes on distribution of aggregate size fractions and organic C content of a long-term fertilized soil. European Journal of Soil Biology 46, 365-370.

Majumder, B., Kuzyakov, Y., 2010. Effect of fertilization on decomposition of C-14 labelled plant residues and their incorporation into soil aggregates. Soil & Tillage Research 109, 94-102.

Marin-Spiotta, E., Swanston, C.W., Torn, M.S., Silver, W.L., Burton, S.D., 2008. Chemical and mineral control of soil carbon turnover in abandoned tropical pastures. Geoderma 143, 49-62.

Marschner, B., Brodowski, S., Dreves, A., Gleixner, G., Gude, A., Grootes, P.M., Hamer, U., Heim, A., Jandl, G., Ji, R., Kaiser, K., Kalbitz, K., Kramer, C., Leinweber, P., Rethemeyer, J., Schaeffer, A., Schmidt, M.W.I., Schwark, L., Wiesenberg, G.L.B., 2008. How relevant is recalcitrance for the stabilization of organic matter in soils? Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 171, 91-110.

McFarland, M.J., Vasquez, I.R., Vutran, M., Schmitz, M., Brobst, R.B., 2010. Use of Biosolids to Enhance Rangeland Forage Quality. Water Environment Research 82, 455-461.

McLean, E.O., 1982. Soil pH and lime requirement. In: Page, A.L. (Ed.), Methods of Soil Analysis, Part 2. American Society of Agronomy, Madison, Wisconsin, pp. 199-209.

Morgan, J.A., Follett, R.F., Allen, L.H., Jr., Del Grosso, S., Derner, J.D., Dijkstra, F., Franzluebbers, A., Fry, R., Paustian, K., Schoeneberger, M.M., 2010. Carbon sequestration in agricultural lands of the United States. Journal of Soil and Water Conservation 65, 6A-13A.

Nguyen, T.T., Janik, L.J., Raupach, M., 1991. Diffuse reflectance infrared fourier-transform (DRIFT) spectroscopy in soil studies. Australian Journal of Soil Research 29, 49-67.

Ouatmane, A., Provenzano, M.R., Hafidi, M., Senesi, N., 2000. Compost maturity assessment using calorimetry, spectroscopy and chemical analysis. Compost Science & Utilization 8, 124-134.

Pagliai, M., Guidi, G., Lamarca, M., Giachetti, M., Lucamante, G., 1981. Effects of sewage sludges and composts on soil porosity and aggregation. Journal of Environmental Quality 10, 556-561.

Pagliai, M., Vignozzi, N., Pellegrini, S., 2004. Soil structure and the effect of management practices. Soil & Tillage Research 79, 131-143.

Pandey, C., Shukla, S., 2006. Effects of composted yard waste on water movement in sandy soil. Compost Science & Utilization 14, 252-259.

Pascual, J.A., Garcia, C., Hernandez, T., 1999. Lasting microbiological and biochemical effects of the addition of municipal solid waste to an arid soil. Biology and Fertility of Soils 30, 1-6.

Paustian, K., Andren, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tiessen, H., Van Noordwijk, M., Woomer, P.L., 1997. Agricultural soils as a sink to mitigate CO2 emissions. Soil Use and Management 13, 230-244.

Poirier, N., Sohi, S.P., Gaunt, J.L., Mahieu, N., Randall, E.W., Powlson, D.S., Evershed, R.P., 2005. The chemical composition of measurable soil organic matter pools. Organic Geochemistry 36, 1174-1189.

Powlson, D.S., Bhogal, A., Chambers, B.J., Coleman, K., Macdonald, A.J., Goulding, K.W.T., Whitmore, A.P., 2012. The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study. Agriculture Ecosystems & Environment 146, 23-33.

Razafimbelo, T.M., Albrecht, A., Oliver, R., Chevallier, T., Chapuis-Lardy, L., Feller, C., 2008. Aggregate associated-C and physical protection in a tropical clayey soil under Malagasy conventional and no-tillage systems. Soil & Tillage Research 98, 140-149.

Robertson, G.P., Coleman, D.C., Bledsoe, C.S., Sollins, P., 1999. Standard Soil Methods for Long-term Ecological Research. Oxford University Press, New York.

Rodionov, A., Amelung, W., Urusevskaja, I., Zech, W., 2000. Carbon and nitrogen in the enriched labile fraction along a climosequence of zonal steppe soils in Russia. Soil Science Society of America Journal 64, 1467-1473.

Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Koegel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S., Trumbore, S.E., 2011. Persistence of soil organic matter as an ecosystem property. Nature 478, 49-56.

Silver, W.L., Ryals, R., Eviner, V., 2010. Soil Carbon Pools in California's Annual Grassland Ecosystems. Rangeland Ecology & Management 63, 128-136.

Simonetti, G., Francioso, O., Nardi, S., Berti, A., Brugnoli, E., Lugato, E., Morari, F., 2011. Characterization of humic carbon in soil aggregates in a long-term experiment with manure and mineral fertilization. Soil Science Society of America Journal.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. Plant and Soil 241, 155-176.

Six, J., Elliott, E.T., Paustian, K., 2000. Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. Soil Biology & Biochemistry 32, 2099-2103.

Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Science Society of America Journal 62, 1367-1377.

Six, J., Guggenberger, G., Paustian, K., Haumaier, L., Elliott, E.T., Zech, W., 2001. Sources and composition of soil organic matter fractions between and within soil aggregates. European Journal of Soil Science 52, 607-618.

Smidt, E., Lechner, P., Schwanninger, M., Haberhauer, G., Gerzabek, M.H., 2002. Characterization of waste organic matter by FT-IR spectroscopy: Application in waste science. Applied Spectroscopy 56, 1170-1175.

Sullivan, T.S., Stromberger, M.E., Paschke, M.W., Ippolito, J.A., 2006. Long-term impacts of infrequent biosolids applications on chemical and microbial properties of a semi-arid rangeland soil. Biology and Fertility of Soils 42, 258-266.

Swanston, C.W., Torn, M.S., Hanson, P.J., Southon, J.R., Garten, C.T., Hanlon, E.M., Ganio, L., 2005. Initial characterization of processes of soil carbon stabilization using forest stand-level radiocarbon enrichment. Geoderma 128, 52-62.

Tisdall, J.M., Oades, J.M., 1982. ORGANIC-MATTER AND WATER-STABLE AGGREGATES IN SOILS. Journal of Soil Science 33, 141-163.

Zebarth, B.J., Neilsen, G.H., Hogue, E., Neilsen, D., 1999. Influence of organic waste amendments on selected soil physical and chemical properties. Canadian Journal of Soil Science 79, 501-504.

Tables

Table 1. Selected infrared bands for soil fraction DRIFT spectra.

Wavenumber (cm^{-1})	Vibration	Functional group or compound
2910 – 2930	Asymmetric C-H stretching	aliphatic methylene groups[a,b,c,d]
2853	Symmetric C-H stretching	aliphatic methylene groups[a,b,c]
1653	C=O stretching	amid I, carboxylates[e]
	C=C stretching	aromatic ring modes, alkenes[b,d,f]
1420	COO- stretching	carboxylic acids[b]

[a] Tatzber et al. 2007; [b] Smith 1999; [c] Smidt et al. 2002; [d] Haberhauer et al. 1998; [f] Outamane et al. 2000

Table 2. Size class characteristics of compost applied at both grassland experimental sites. Values are means with standard errors of replicate samples in parentheses.

Compost Size Class	Fraction of Total	% C	% N	C : N
< 1 mm	43 (1)	20.9 (0.05)	1.85 (0.01)	11.3 (0.1)
1 - 2 mm	35 (1)	23.7 (0.49)	1.84 (0.03)	12.9 (0.2)
2 - 4.75 mm	16 (3)	22.6 (0.81)	1.53 (0.11)	14.3 (0.4)
> 4.75 mm	6.0 (0.5)	28.5 (1.7)	1.52 (0.01)	18.7 (1.2)
rocks	0.8 (0.4)	----	----	----
synthetics	1.3 (0.6)	----	----	-----
bulk	100	22.7 (0.5)	1.78 (0.05)	12.7 (0.2)

Table 3. Mass, C, and N characteristics of soil density fractions

Fraction	**Grassland**	**Treatment**	**Percent of bulk soil mass**	**C (%)**	**C (Mg C/ha)**	**N (%)**	**N (Mg C/ha)**	**C : N**
Free light fraction	Valley	control	3.2 (0.1)*	21.8 (1.0)	5.57 (0.54)*	1.14 (0.01)*	0.30 (0.02)*	19.2 (0.7)*
		amended	4.0 (0.2)*	22.2 (0.4)	8.13 (1.00)*	1.56 (0.04)*	0.57 (0.07)*	14.2 (0.2)*
	Coastal	control	0.9 (0.2)*	21.3 (1.1)	1.88 (0.10)*	1.13 (0.03)*	0.10 (0.01)*	18.8 (1.2)*
		amended	1.8 (0.7)*	22.9 (1.4)	3.70 (1.22)*	1.44 (0.03)*	0.24 (0.10)*	15.8 (0.9)*
Occluded light fraction	Valley	control	3.3 (0.4)	28.5 (3.3)	7.15 (1.00)	1.58 (0.22)*	0.40 (0.06)**	18.0 (0.8)*
		amended	2.9 (0.4)	31.1 (1.3)	7.91 (0.62)	2.00 (0.17)*	0.51 (0.02)**	15.4 (0.6)*
	Coastal	control	2.2 (0.7)*	31.7 (1.6)	6.58 (1.64)*	1.90 (0.09)*	0.41 (0.11)*	16.5 (0.5)*
		amended	2.5 (0.6)*	33.0 (1.1)	7.87 (2.00)*	2.25 (0.09)*	0.54 (0.14)*	14.6 (0.1)*
Heavy fraction	Valley	control	93.5 (0.4)	1.45 (0.10)	10.8 (0.4)**	0.15 (0.01)*	1.10 (0.06)*	9.9 (0.4)**
		amended	93.1 (0.2)	1.48 (0.11)	12.7 (1.9)**	0.16 (0.01)*	1.34 (0.15)*	9.4 (0.4)**
	Coastal	control	96.9 (0.8)*	1.64 (0.40)	15.2 (2.0)**	0.17 (0.04)	1.60 (0.18)	9.5 (0.2)
		amended	95.7 (1.1)*	1.58 (0.24)	14.4 (2.4)**	0.17 (0.02)	1.53 (0.24)	9.4 (0.2)

* indicates significant within-site treatment differences of each fraction at $p < 0.05$
** indicates significant within-site treatment differences of each fraction at $p < 0.10$

Table 4. DRIFT peak heights of aliphatic methylene groups present at wavenumbers 2920 and 2852 cm^{-1} (Band A), carbonyl and carboxyl groups present at wavenumbers 1653 and 1400 cm^{-1} (Band B), and the ratio of Band B : Band A absorbance in soil density fractions and in bulk compost.

Fraction	**Grassland**	**Treatment**	**Absorbance (cm^{-1} (mg OC)$^{-1}$) at**				**Ratio of Band B : Band A**
			Wavenumber 2920 cm$^{-1, a}$	**Wavenumber 2853 cm$^{-1, a}$**	**Wavenumber 1653 cm$^{-1, b}$**	**Wavenumber 1400 cm$^{-1, b}$**	
Free light fraction	Valley	control	1.21 (0.71)	0.25 (0.20)	1.97 (0.81)	0.68 (0.23)	2.82 (0.94)
		amended	0.64 (0.19)	0.10 (0.06)	1.59 (0.17)	0.57 (0.05)	4.08 (1.33)
	Coastal	control	1.02 (0.43)	0.07 (0.04)	1.28 (0.10)	0.42 (0.04)	2.53 (1.19)
		amended	0.72 (0.33)	0.07 (0.04)	1.78 (0.21)	0.68 (0.10)	5.48 (2.72)
Occluded light fraction	Valley	control	0.34 (0.16)	0.28 (0.16)	1.07 (0.13)	0.46 (0.08)	2.88 (0.55)
		amended	0.67 (0.35)	0.17 (0.14)	1.68 (0.13)	0.65 (0.08)	6.05 (3.61)
	Coastal	control	0.58 (0.45)	0.48 (0.34)	1.41 (0.17)	0.56 (0.07)	2.78 (0.97)
		amended	0.32 (0.15)	0.27 (0.16)	1.42 (0.13)	0.66 (0.06)	4.15 (1.20)
Heavy fraction	Valley	control	0.64 (0.24)	0.65 (0.32)	2.23 (0.42)	0.20 (0.07)	4.65 (1.89)
		amended	0.52 (0.21)	0.60 (0.25)	1.80 (0.35)	0.14 (0.04)	2.69 (1.10)
	Coastal	control	0.56 (0.20)	0.32 (0.08)	1.74 (0.25)	0.22 (0.06)	4.10 (1.43)
		amended	0.94 (0.43)	0.48 (0.19)	1.72 (0.30)	0.11 (0.05)	2.22 (0.93)
Bulk Compost	---	---	0.29 (0.07)	0.21 (0.07)	2.14 (0.14)	0.72 (0.09)	7.00 (2.31)

a indicates groups that are included in Band A
b indicates groups that are included in Band B
OC = organic C content

Figures

Figure 1. Soil texture at two grassland types. Stacked bars represent mean values of clay, silt, and sand components from control and amended soils (n = 3 pre treatment) from all paired plots (Block 1, 2, 5 at the valley grassland and WR 5, 6, and 7 at the coastal grassland). Error bars are ± 1 standard error.

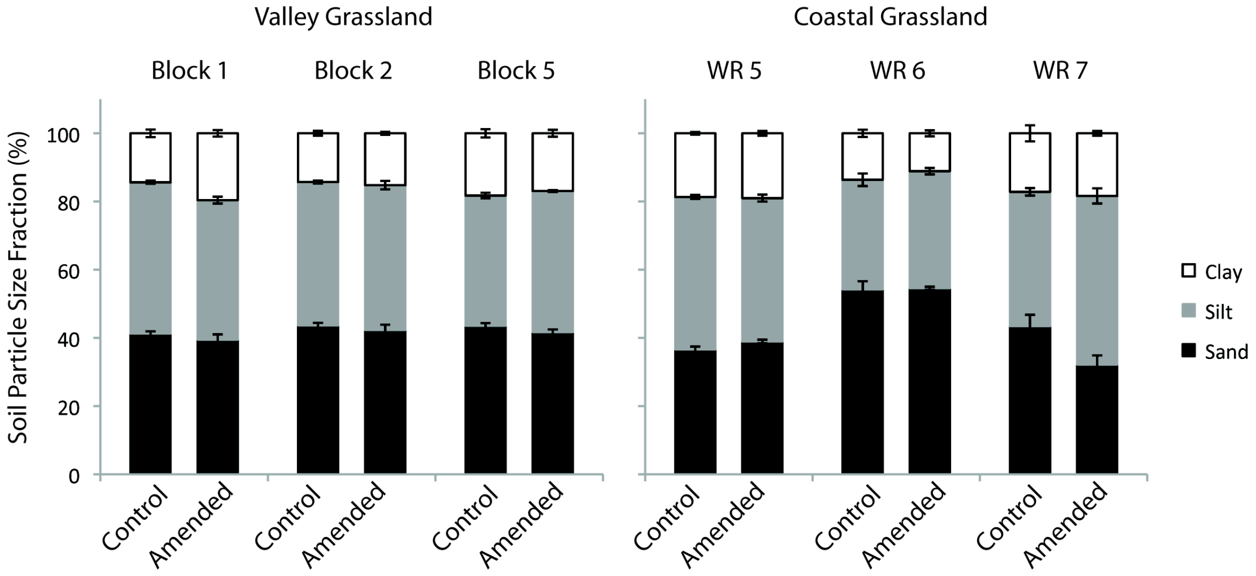


Figure 2. (a) Total soil organic C and total N content to 10 cm depth prior to and three years following organic matter amendment at two grassland types. Circles with solid lines represent C and N content of controls soils, and circles with dashed lines represent those of amended soils. (b) Treatment effects are shown as paired plot differences between amended and control soils. Positive values indicate greater C or N in amended soils versus control soils. Soils were sampled annually, at the end of each water year (WY). Pre-treatment (PreT) data are shaded. * indicates $p < 0.05$ for differences between paired plots.

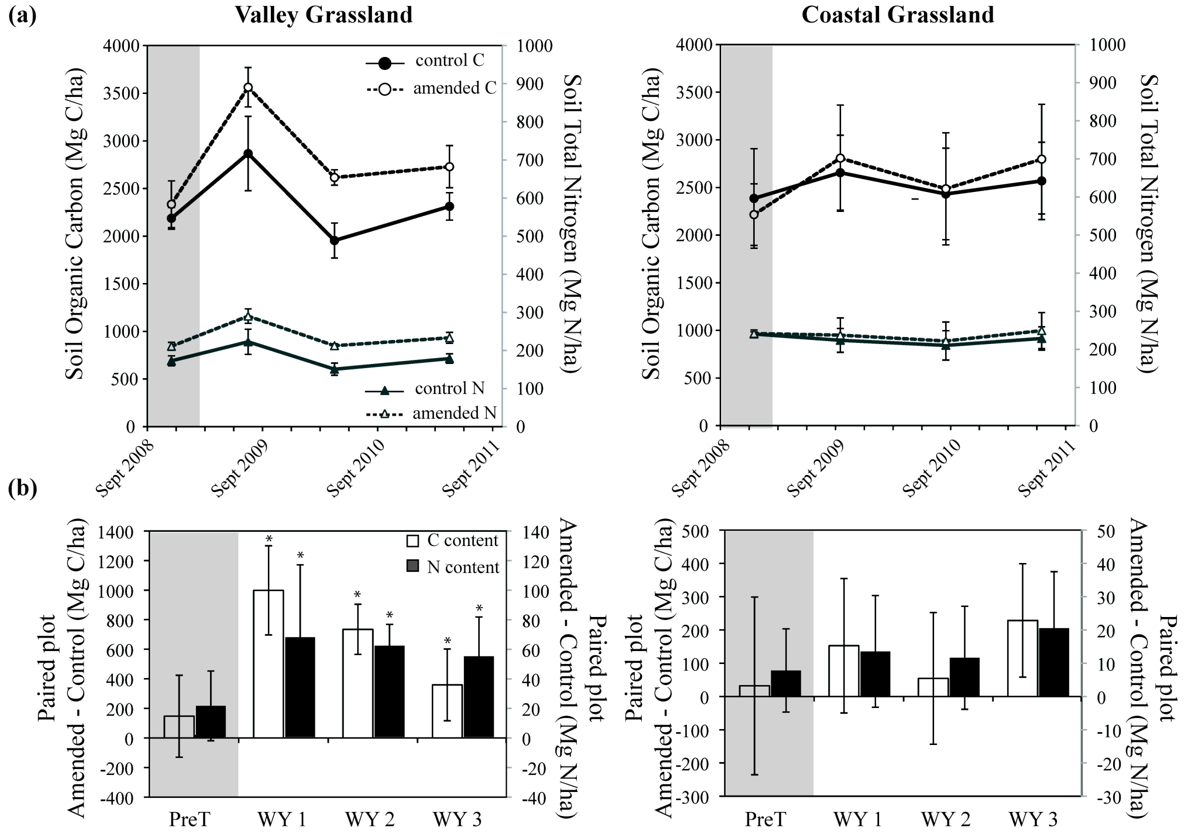


Figure 3. (a) The contribution of three soil fractions to total soil organic C content from 0 to 10 cm depth for control and amended soils in two grassland ecosystems (mean ± 1 standard error, n = 3). (b) The treatment effect on C storage for each soil fraction shown as differences between each pair of amended and control plots (n = 3). (c) The contribution of three soil fractions to total soil N content from 0 to 10 cm depth. (d) The treatment effect on soil N content for each soil fraction shown as paired plot treatment differences. * Indicates $p < 0.1$; ** Indicates $p < 0.05$ for differences between paired plots.

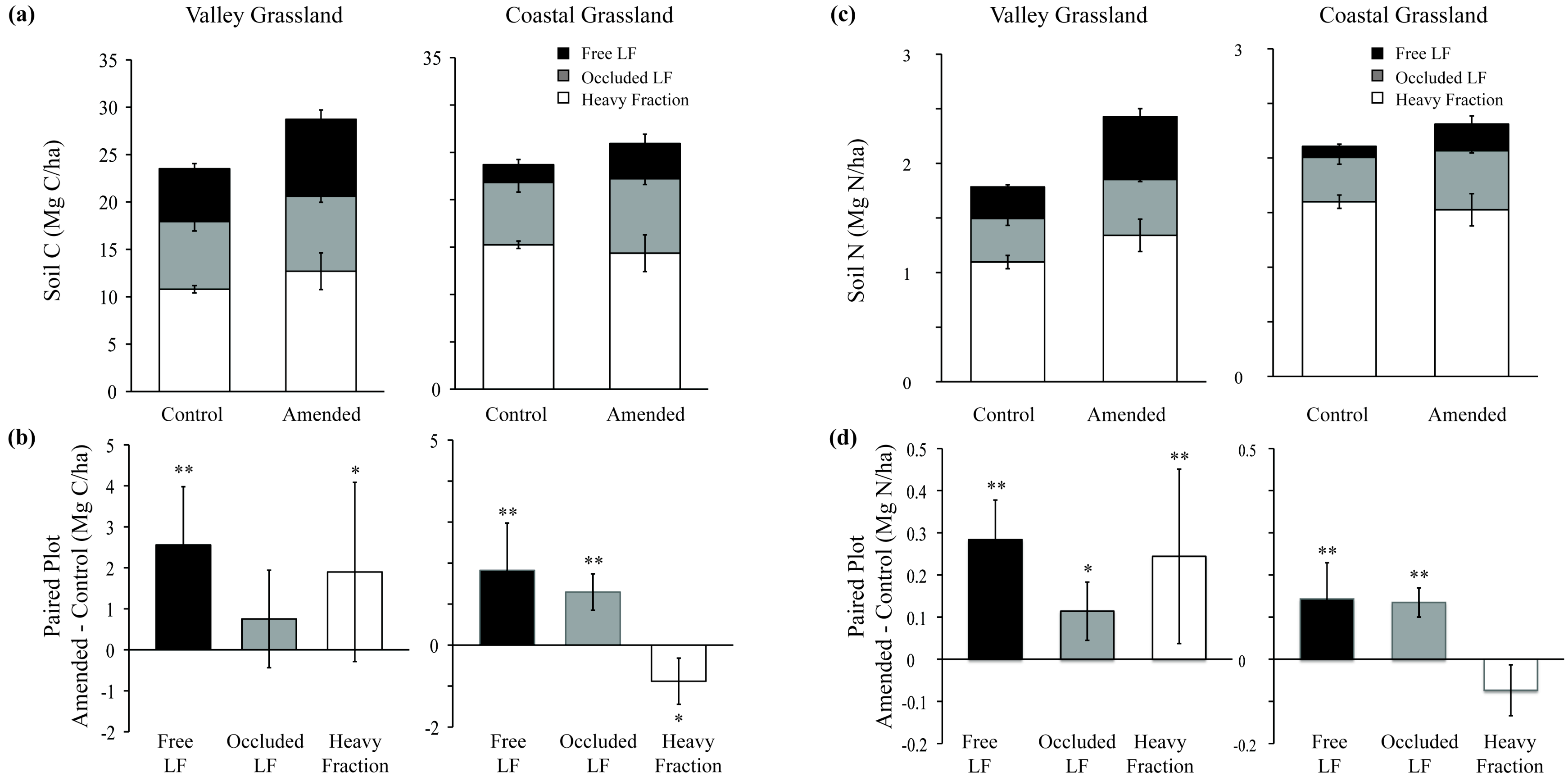


Figure 4. Representative DRIFT spectra of control and amended soil density fractions and of bulk compost. Spectra are means of field replicates (n =3). Assignments for evaluated bands are shown in Table 2.

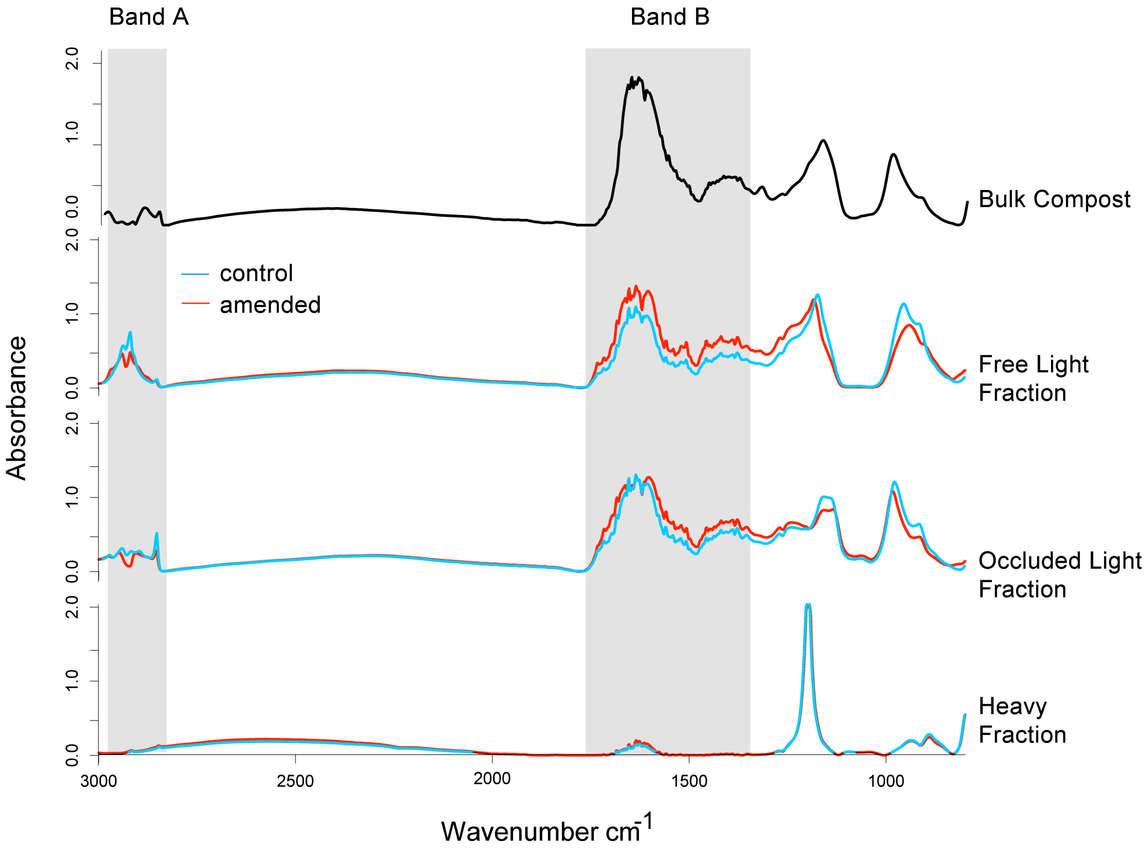


Figure 5. DRIFT spectra peak heights of compounds identified in Table 2 from FLF, OLF, and HF in amended and control soils from valley and coastal grasslands. Bars represent mean peak height absorbance. Standard errors are included in Table 4.

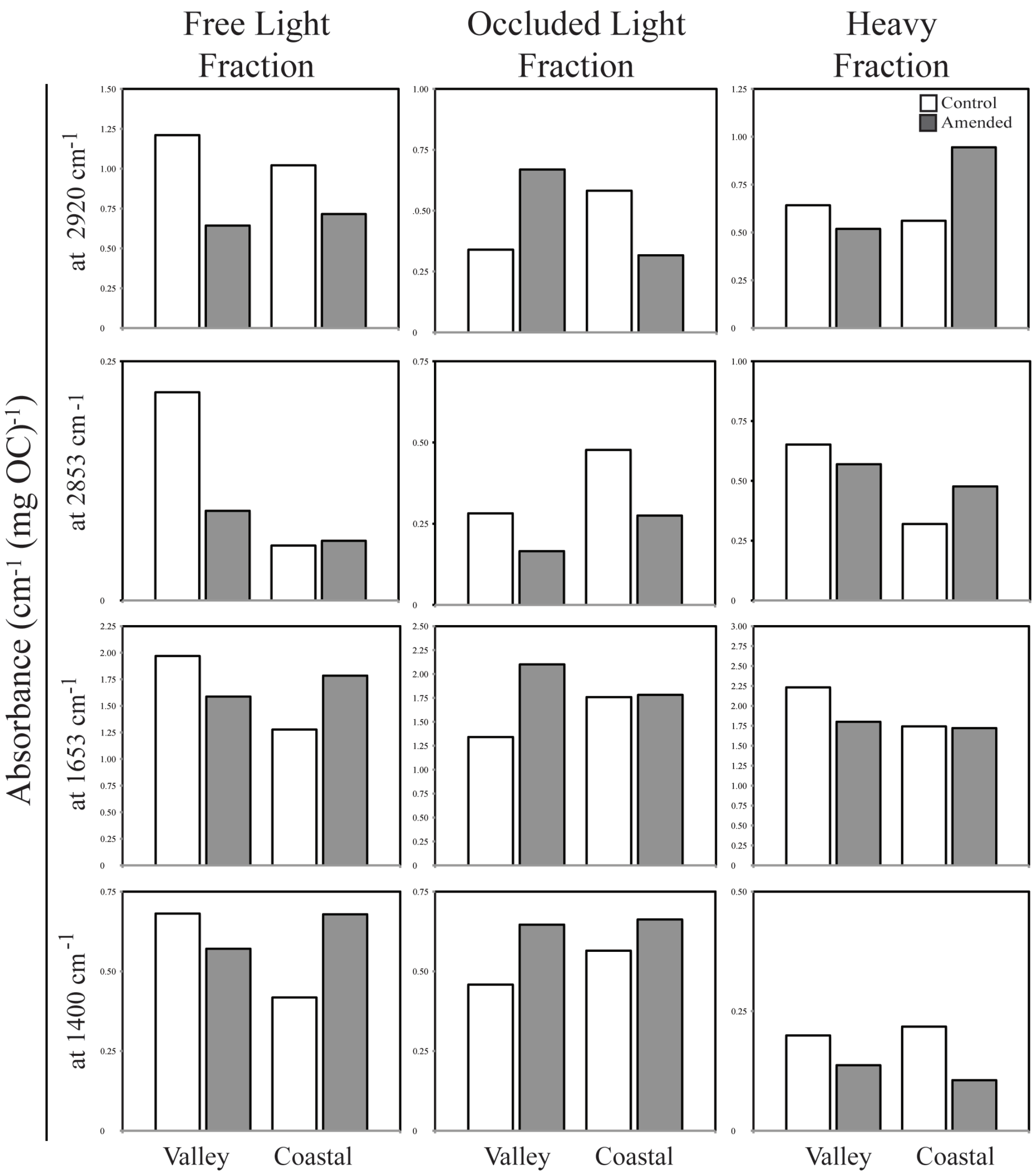
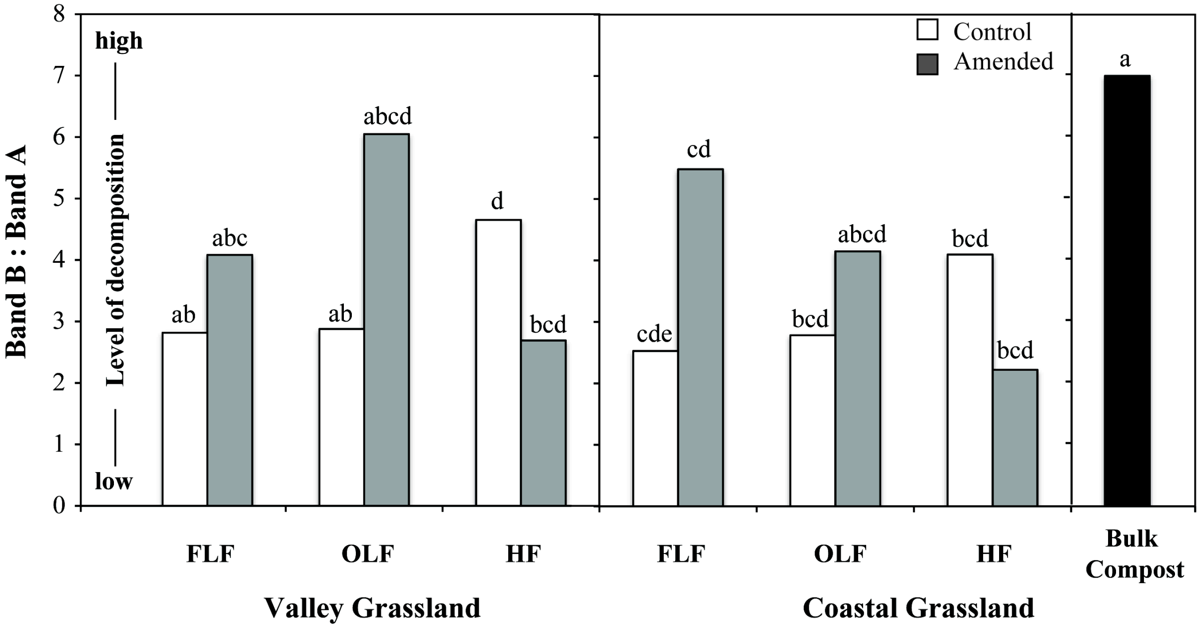


Figure 6. Ratio of Band B absorbance to Band A absorbance from FT-IR spectra of free-light (FLF), occluded-light (OLF), and heavy fractions (HF) from valley grassland and coastal grassland experimental plots. White bars represent mean values from control soils. Gray bars represent mean values from amended soils. The black bar represents the mean value of bulk compost. Standard errors are provided in Table 4. Letters above bars represent statistical significance at $p < 0.05$.



Chapter 3. Simulating soil carbon and greenhouse gas dynamics in grasslands amended with compost

Abstract

Compost amendment to grasslands has been proposed as a way to mitigate climate change through carbon (C) sequestration, yet little research has been done exploring the source-sink potential of this management strategy. I used the ecosystem biogeochemical model, DAYCENT, to investigate the climate change mitigation potential of compost amendments to three grassland sites in California, including a valley grassland in the Sierra foothills and two coastal grasslands that differed in soil texture (e.g. sandy loam and loam texture) in northern California. The model was parameterized using site-specific characteristics, including long-term weather records and edaphic characteristics. Model validation was conducted by comparing simulated above- and belowground net primary production (NPP) and soil C with that from a three-year field experiment at each site and iteratively adjusting crop parameters. I then used the model to test ecosystem responses and source-sink potential of a variety of compost qualities and application rates. I found that ecosystem C and N responded rapidly to amendments, but the effects tended to be down-regulated by higher compost C:N ratios. Carbon sequestration rates were greater with low C:N compost, but soils amended with low C:N ratio compost experienced greater N_2O fluxes relative to composts with higher C:N ratios. These results suggest a trade-off between maximizing plant production and minimizing N_2O losses. I also found that the source-sink potential varied greatly when considered over short (10 year), medium (30 year), or long (100 year) time periods. I conclude that compost amendments to rangeland soils can result in significant C sinks, but that the full suite of soil greenhouse gas emissions and timeframes for C sequestration must be explicitly considered.

Introduction

Carbon sequestration in rangeland soils has been proposed as a means to remove carbon dioxide (CO_2) from the atmosphere (Lal, 2004). Rangeland management practices that increase plant C inputs to the soil via irrigation, fertilization, or improved grazing practices are often associated with greater soil C stocks (Conant *et al.*, 2001; Follett, 2001; Dean *et al.*, 2012). Soil C stocks have been shown to increase in short-term field experiments with organic matter additions in agricultural and rangeland soils (Lynch *et al.*, 2005; Cabrera *et al.*, 2009; Ippolito *et al.*, 2010). Organic matter amendments represent a direct C addition to the soil and can contribute to offsets such as avoided methane (CH_4) emissions when composted material is diverted from landfills or slurry ponds (DeLonge *et al.*, in review). Organic matter amendments also enhance C inputs indirectly through increased plant production. Productivity of most rangeland ecosystems is limited by nitrogen (N) and water (Derner and Schuman, 2007; Harpole *et al.*, 2007). Rangeland management practices that increase N and/or water availability generally boost plant production (Derner and Schuman, 2007).

Organic matter amendments can also lead to greater greenhouse gas emissions from soils (Dalal *et al.*, 2003; Gregorich *et al.*, 2005). Soil carbon gains are typically associated with increases in soil N stocks, as C and N cycling are tightly coupled under most conditions (Gagnon *et al.*, 1998; Conant *et al.*, 2005). Mechanisms limiting ecosystem N losses tend to be less effective as ecosystems become more N-rich. As N availability increases nitrous oxide (N_2O) and N leaching are also likely to increase (Chang *et al.*, 1998; Bouwman *et al.*, 2002). This can decrease the net value of the practice from a global warming perspective.

Ecosystem C sequestration protocols operate on the timescale of management (typically 1-10 years), but require protection of sequestered C in perpetuity. Long-term studies (> 20 years) of management effects on soil C storage and greenhouse gas emissions are lacking in rangelands, and few exist to test the long-term impacts of organic matter additions to agricultural soils (Paustian *et al.*, 1997; Blair *et al.*, 2006; Powlson *et al.*, 2012). It is unclear if the C added to the ecosystem via amendments (both direct and indirect) are retained and if so for how long. Thus the ultimate fate of amendment-C once applied to soil and the long-term ecosystem response to amendments in rangelands remains unclear.

The purpose of this research was to assess the climate change mitigation potential of compost additions to rangeland soils and to compare the relative costs of greenhouse gas emissions from soil with the benefits of C sequestration in these ecosystems. I used the DAYCENT ecosystem model to simulate changes to soil greenhouse gas fluxes (N_2O, CH_4, and heterotrophic respiration) and C stocks following a single application of composted green waste. I evaluated model simulations compared to field observations of above- and belowground net primary production (NPP), soil C, and soil greenhouse gas fluxes. Finally I use the model to test the climate change mitigation potential of a range of compost amendment scenarios differing in application intensity, frequency, and C:N ratio. I explored the short (10 year), medium (30 year), and long-term (100 year) impacts of each scenario.

Methods

DAYCENT model description

The DAYCENT model is the daily time step version of CENTURY, a generalized model used to simulate ecosystem C, N, S, and P dynamics (Parton *et al.*, 2001). CENTURY and DAYCENT are widely-used biogeochemical models first calibrated to grasslands and since used for a variety of ecosystem types, climate simulations, and management scenarios (Kelly *et al.*, 2000; Del Grosso *et al.*, 2002; Li *et al.*, 2006; Parton *et al.*, 2007). These models were developed to simulate changes to ecosystem C, nutrient dynamics, and plant productivity in response to changes in land management and climate (Parton *et al.*, 1994). DAYCENT was developed to accurately simulate soil fluxes of CO_2, CH_4, and N_2O at a finer timescales, and is a state of the art assessment tool for regional to global greenhouse gas inventories (e.g. (EPA, 2012). The model contains interacting soil organic matter, waterflow, denitrification, nitrification, and P submodels. I used DAYCENT_UV, a version of the model derived from DAYCENT version 4.5, which explicitly simulates photosynthesis as well as photodegradation of surface litter and is publicly available at http://www.nrel.colostate.edu/projects/daycent/.

Site descriptions

Data from replicated field experiments within valley grasslands and coastal grasslands in California were used to parameterize and validate the DAYCENT model. The valley grassland site was located in the Sierra foothills, with mean annual precipitation of 730 mm/yr (22 y mean), and seasonal temperatures ranging from 2 ºC in January to 35 ºC in July (Figure 1). The coastal grassland site was located in the north-central coast of California, with mean annual precipitation of 950 mm/yr (38 y mean), and seasonal temperatures ranging from 6 ºC in January to 20 ºC in July (Figure 1). Both grassland types are dominated by non-native annual grasses, with native perennial bunchgrasses also present at the coastal grassland. Within each grassland type, three replicate blocks were established with treatments consisting of a single organic matter amendment and non-amended controls. Detailed site descriptions are available in Ryals and Silver (Ryals and Silver, in press).

Ecosystem data were collected for three years following the one-time application of composted green waste. Field data collection included aboveground net primary production (ANPP), belowground net primary production (BNPP), plant C and N content, weekly to monthly soil CO_2 flux, biweekly to monthly soil CH_4 and N_2O fluxes, annual soil C and N content, and soil texture. For modeling purposes, three site characterizations were used, including one representing valley grasslands and two representing coastal grasslands. One site characterization was used to represent all three valley grassland blocks as differences in ecosystem characteristics and management responses between blocks at the valley grassland site were minimal. These model simulations are referred to as “valley.” Two site characterizations were generated at the coastal grassland site to capture significant differences in soil texture between blocks. Simulations referred to as “coastal sandy loam” represent site conditions from the coastal grassland block with sandy-loam textured soils. Simulations referred to as “coastal loam” represent average site conditions from the other two coastal blocks which did not differ significantly in edaphic characteristics.

Model parameterization

There are more than 1000 interacting parameters in DAYCENT, but only a small subset of these are needed to calibrate the generalized model to site-specific conditions (Parton *et al.*, 1998; Parton *et al.*, 2001). Required abiotic drivers for the DAYCENT model consist of daily precipitation and daily minimum and maximum air temperatures. For valley grassland simulations, weather data from 1990 - 2012 were acquired from a local station (Browns Valley, #84), which is part of the California Irrigation Management Information System (CIMIS, http://wwwcimis.water.ca.gov/cimis/data.jsp) located less than two kilometers northwest of the study site. For coastal grassland simulations, local precipitation data from 1980 - 2012 were acquired < 2 km from the site at the Nicasio Town Station from Marin Municipal Water District (J. Klein pers. comm.). I simulated air temperature values from Daymet retrieved from 1 km x 1 km pixel using latitude and longitude of the plots (Thornton *et al.*, 2012). Some soil input variables (texture, bulk density, and pH) were measured at each field site, while others (field capacity, wilting point, and saturated hydraulic conductivity) were estimated using algorithms based on soil texture (Saxton *et al.*, 1986). User-defined plant characteristics were established for a C3 perennial grass and a C3 annual grass growing in a Mediterranean climate (Appendix 5).

Model equilibration consisted of a period with perennial grass and light grazing (between year 0 and 1750; run until equilibrium), a period with annual grass and moderate grazing representing the widespread shift in vegetation due to introduction of European grasses and cattle grazing (between years 1750 and 1900), and a period of modern cattle grazing with annual grasses (between years 1900 and 2007).

Once weather, soils, crop, and site parameters were determined for each site characterization, the model was further calibrated by evaluating model outputs under model equilibrium conditions. Since long-term NPP and soil C datasets are not available for these sites, I assessed whether the model outputs were within the range of published values from similar ecosystems (*e.g.* (Chou *et al.*, 2008; Silver *et al.*, 2010) and within the range of observed values from control plots from 2008 – 2012. When modeled ANPP and BNPP were unrealistic, it was necessary to adjust parameters controlling the potential growth rate (PRDX), and the fraction of C allocated to roots at germination (FRTC(1)), during moisture stress (FRTC(4)), and under nutrient stress (FRTC(5)) in vegetation input files. Once the vegetation input parameter values were optimized, they remained unchanged in later simulations. When absolute values of simulated soil C differed from measured values, I checked that the magnitude and direction of soil C change with organic matter amendment was similar between measured and simulated values (Del Grosso *et al.*, 2005).

Simulated management scenarios

I used the model to test the impact of a range of application intensities and compost qualities on ecosystem C storage, soil greenhouse gas emissions, and net climate change mitigation potential. Two types of compost application intensities were simulated: a one-time surface application of 14.27 Mg C ha^{-1} (equivalent to 70 Mg-dry weight ha^{-1}; the amount and intensity applied in the field experiment) and annual applications of 1.427 Mg C ha^{-1} y^{-1} for ten consecutive years. This approach allowed for the total amount of nutrient inputs to be held constant while exploring the difference between the lasting effects of acute versus chronic nutrient additions. Compost application was initiated in the model using the OMAD function, which includes information about the C content, C to nutrient ratios, lignin content, and OMAD type, which indicates which pools in the model the amendment is added to initially. In these simulations, composted material was added to the model-defined surface slow pool (*i.e.* som2c(1)) to mimic partially decomposed organic material. To test a range of compost qualities, I adjusted the C:N ratio from 11.1 (S11, as in the field experiment), to 20 (S20), and to 30 (S30). These conditions represent a range of suggested C:N ratios for finished compost depending on feedstock and level of maturity (CCQC, 2001). Finally, the variations in application intensities and compost qualities were run in combination. A list and description of model scenarios is provided in Table 1.

In an experimental model run of the S11 scenario, I added the compost as ^{14}C-labeled material into the surface SOM pool. This allowed me to trace the decomposition of compost through time following application by tracing the extinction of the added label through time.

Model validation, scenario analyses, and statistical analyses

Model simulations considered during validation include the control and S11 scenarios, which correspond to the management treatments included in the field experiment described above and in Ryals and Silver (in press). I tested the validity of the model using correlation analyses between measured versus modeled values of soil C, ANPP, and BNPP for all three site characterizations combined. Measured plot-level values (mean ± 1 standard error) prior to treatment for three years following treatment were included in correlation analyses for all sites and for control and amended plots.

Most model output variables were plotted at an annual time scale from years 2008 through 2108. Treatments were compared as the differences relative to the control scenario. Soil fluxes of N_2O, CH_4, and heterotrophic respiration were plotted on a daily timescale for three years following compost addition, as these variables are temporally dynamic and sensitive to short-term changes in microclimate. Annual climate change mitigation potential was assessed over a 10, 30 and 100-year timeframe, and calculated as ecosystem C sequestration minus greenhouse gas emissions in units of g CO_2-eq m^{-2} y^{-1} using global warming potentials of 1, 25, and 298 for CO_2, CH_4, and N_2O (IPCC, 2007), as follows:

Climate change mitigation potential (g CO_2-eq m^{-2} y^{-1}) = $\frac{\Delta C_{seq} - \Delta GHG_{soil}}{t - 2008}$

[Eq.1]

where t refers to the last year considered in the timeframe of mitigation potential. I calculated rates over a 10 (year 2018), 30 (year 2038), and 100-year (year 2108) timeframe. The term, ΔC_{seq}, refers to the difference in total system C between amended and control simulations. Total system C included aboveground biomass, belowground biomass, and soil C to 20 cm depth. Additional C losses via heterotrophic soil respiration are accounted for in changes to the soil C pool. The term ΔGHG_{soil} refers to the difference in soil N_2O and CH_4 emissions from year 2008 to year t between amended and control simulations. Nitrous oxide emissions included contributions from gaseous fluxes and nitrate (NO_3^-) leaching. Leaching N_2O losses were calculated using an emissions factor of 0.0075 g N_2O-N / g NO_3^- leached (Nevison, 2000). I use the sign convention where a net source to the atmosphere is a negative sum and a net sink to the ecosystem is a positive sum, compared to control simulations.

To assess the effect of management on model simulation results, I used analysis of variance (ANOVA) with treatment as a fixed-effect and site as a blocking effect. When significant at $p < 0.05$, ANOVA tests were followed by student's t-test. For response variables, I used ANPP, BNPP, NO_3^- leaching, N_2O flux, CH_4 flux, and heterotrophic respiration summed over 10, 30, or 100 years for each site (n = 3). Differences in total system C, soil C and active, slow, and passive soil C pools at year 10, 30, and 100 were also assessed. To assess changes over time, I used repeated measures ANOVA with annual measures of ANPP, BNPP, NO_3^- leaching, N_2O flux, CH_4 flux, heterotrophic respiration, total system C, soil C, and active, slow, and passive soil C pools as response variables. For repeated measures analyses, site, treatment, time, and treatment x time interactions were included as factors. Analyses were performed using 7.0.2 JMP software (SAS Institute Inc. 2007). Statistical significance was determined as $p < 0.05$ unless otherwise noted. Data are reported as mean ± 1 standard error and plotted separately for each site.

Results

Effects of initial soil conditions

During model equilibration, soil C increased slightly until the year 1750, when I induced a shift from perennial grasses to annual grasses (Figure 2). The vegetation shift caused an exponential decline in soil C, the rate of which reduced but did not stabilize by the year 2120. The average decline in soil C over the period of 1750 to 1900 was approximately -2.7, -2.9, and -4.3 g C m^{-2} y^{-1} at the coastal sandy loam, coastal loam, and valley grassland sites. Rates of soil C loss declined to -0.5 to -0.9 g C m^{-2} y^{-1} at the coastal and valley grassland sites from the period of 1990 to 2000, and continued at a similar rate when projected for 100 years. Changes to soil C induced by management were considered in the context of the underlying decline observed with the baseline control model.

Model validation

The model was able to approximate soil C at the coastal sandy loam and valley grassland sites, despite considerable interannual variability in soil C measurements of both control and amended plots, particularly at the valley grassland. Simulated soil C at the coastal loam site was consistently lower than measured values. However, the model did a good job of simulating the relative response of soil C to compost amendment (Figure 3a). The model tended to slightly overestimate control ANPP and underestimate amended ANPP compared to measured values (Figure 3b). Belowground NPP in the model also correlated reasonably well with measured values (Figure 3c). Modeled daily N_2O and CH_4 fluxes were typically low, with short-lived N_2O pulses associated with rainfall events, and fluxes were within the range of values measured in a range field conditions at these sites. Annual total soil CO_2 fluxes, including heterotrophic and autotrophic components, were also within the range of observed fluxes over the first three years of the field experiment.

The extinction of the ^{14}C-labelled amendment material was traced through time. The mass of amendment remaining was plotted against time (Figure 4). Both exponential ($R^2 = 1.00$) and linear ($R^2 = 0.98$) curves fit the simulated data well, reflecting the partially decomposed nature of compost amendments compared to fresh plant litter. The decomposition k value predicted in the model was 0.057 y^{-1}. Approximately 60 % and 30 % of the compost-C remained in the ecosystem 10 and 30 years following application to soil, respectively.

Temporal extrapolation of field observations: effects of compost addition on soil C

The DAYCENT model identifies three soil C pools that are explicitly defined by their turnover times: active (1-5 years), slow (20-40 years), and passive (200-1500 years). Simulated changes to these three soil C pools were similar in trend and magnitude across all three sites (Figure 5). In control model simulations, the active and slow pools remain relatively constant through time, with slight interannual variability. In contrast, the passive soil C decreased through time in the control model, reflecting ongoing losses of soil C observed during model equilibrium following vegetation shift.

In the S11 scenario, all three soil pools experienced gains in C with amendments that persisted through time (Fig 5). The largest increase was observed in the slow soil C pool, and no pool experienced C loss after compost amendment relative to control soil C pools. The active and slow soil C pools responded quickly to amendment. Treatment differences were greatest at approximately 15 -25 years following the compost addition then declined through time. Acitve and slow C pools remained significantly larger relative to controls at the end of the 100 y simulation ($p < 0.0001$). Compost additions increased the active soil C pool by 14.3 ± 0.2, 11.4 ± 0.5, and 3.6 ± 0.3 g C m^{-2} y^{-1} after 10, 30, and 100 years, respectively, when averaged across sites. For the slow C pool, increases were 501 ± 9.0, 371.5 ± 16.4, and 115.3 ± 2.8 g C m^{-2} y^{-1} after 10, 30, and 100 years, respectively. The passive soil C pool declined through time following amendment, but loss rates were roughly half that of the control. As a result, the treatment differences in passive soil C continued to increase through time and appeared to start saturated after 100 years ($p < 0.0001$). The average increase in the passive soil C in the compost model relative to control across sites was 2.4 ± 0.3, 9.7 ± 1.1, and 20.1 ± 2.2 g C m^{-2} y^{-1} after 10, 30, and 100 years. Thus, the response of total soil C was greatest within the first 30 years following a single addition of compost, but C continues to build up in the most stable, passive soil C pool even after 100 years (Figure 5d).

Application intensity and compost quality

I used the model to test the effect of a range of compost application intensities (a single large addition versus several small additions) and qualities (as determined by C:N ratio). Considerable interannual variability in ANPP was observed in both control and amendment model scenarios. However, ANPP was always greater in amendment scenarios, a result that lasted throughout the duration of the model projection (ANOVA $p = 0.0002$ for 10 year time frame, $p < 0.0001$ for 30 and 100 year time frames) (Figure 6). Aboveground NPP responded rapidly following a single large compost addition (scenarios S11, S20, and S30). Similar net C gains in aboveground plant growth was observed with multiple smaller additions of compost, but the effects were more gradual for the first twenty years. Scenario S11 had the strongest response and accumulated an additional 853, 2758, and 7230 g C m^{-2} compared to control over 10, 30, and 100 years, respectively ($p < 0.05$ for all time frames). Scenario M30 had the weakest response and accumulated an additional 178, 1208, and 3737 g C m^{-2} compared to control over 10, 30, and 100 years, respectively (not significant at 10 years; $p < 0.05$ for 30 and 100 years). The response of ANPP to amendments was almost two times greater at the coastal loam and coastal sandy loam sites than at the valley grassland. Aboveground NPP was greatly affected by compost quality. At the highest C:N ratio tested (30), the increase in ANPP was reduced by about half compared to the low C:N ratio scenario.

The response of cumulative BNPP to amendments was five to seven times greater at the valley grassland than either of the coastal sites ($p < 0.0001$). Belowground NPP increased at the valley and coastal sandy loam sites under all amendment scenarios, but decreased slightly at the coastal loam site (Figure 7). The BNPP response at the coastal sites alternated between positive and negative, whereas the response was positive at the valley grassland site during most years. In the model, allocation to roots changes with N and water availability via the FRTC parameters in the crop file. Thus, BNPP was affected by interannual variability in rainfall as well as the contribution of available N and moisture from compost amendments. The positive response of BNPP to amendments was an order of magnitude less than the aboveground response, which corresponded to the changes observed in the three-year field experiment. The effects of application intensity or compost quality on BNPP were less clear than for ANPP responses due to differences between sites (ANOVA treatment effect $p = 0.50$, site effect $p < 0.0001$) . Generally, the application intensity did not alter the cumulative response at 10, 30 or 100 years. However, the positive BNPP response was reduced with higher compost C:N ratios, an effect that was evident over longer time periods.

Total soil C, the sum of active, slow, and passive pools, increased under all compost scenarios, with greatest differences under scenarios M11 and S11 ($p < 0.05$ for all scenarios considered at 10, 30, or 100 year timeframes) (Figure 8). Smaller, frequent applications of compost resulted in a more gradual build up of soil C and in a shift of the peak of treatment response by about 10 years compared to a single large application. Thus, soil C increased more rapidly and declined earlier in the single application scenarios. Over 100 years, there were no significant differences among the responses of soil C pools to the different application intensity scenarios. However, the effect of compost quality persisted through time. Model scenarios with lower C:N ratios showed significantly greater increases to the total soil C pools compared to higher C:N ratios (Figure 8).

Treatment effects on soil N_2O and CO_2 fluxes were primarily restricted to the wet season (Oct – May), as low moisture conditions limited microbial activity during the dry season (June – Sept) (Figure 9). Amendments tended to increase soil N_2O and heterotrophic CO_2 emissions when moisture was not limiting, and the largest responses corresponded to rain events followed by warm, dry conditions. Increases in daily soil N_2O fluxes were greatest for the single, large compost addition with low C:N ratios. Fluxes were considerably lower for scenarios with multiple, small additions or higher C:N ratios. Cumulative N_2O emissions were significantly greater compared to control from scenarios S11, S20, and M11 over the 10 and 30 year timeframes and from scenarios S11, M11, S20, and M20 over the 100 year time frame ($p < 0.05$). Cumulative differences ranged from 1.41 to 2.57 g N_2O-N m^{-2} at the 100 year timeframe.

Nitrous oxide emissions from NO_3^- leaching were also restricted to the wet season, and treatment differences diminished over time (Figure 10). At the 10-year timeframe, scenarios S11, S20, and M11 were significantly greater than the control scenario. Analyzed at the 100-year timeframe, all scenarios were significantly greater than the control scenario. Scenarios with different compost qualities were significantly different, with the greatest treatment differences resulting from low C:N ratios ($p < 0.05$). Nitrous oxide from NO_3^- leaching were an order of magnitude less than those from gaseous N_2O fluxes.

For heterotrophic soil respiration, treatment differences were greatest for scenarios with a single, large application of compost. C:N mattered less than application intensity, and responses for S11 and S30 were similar through time. Cumulative C losses through heterotrophic respiration were significantly greater compared to control from scenarios when analyzed over the 10 and 30 year timeframes ($p < 0.05$). Cumulative differences ranged from 2832 to 5405 g CO_2-C m^{-2} at the 100 year timeframe. These ecosystems are slight CH_4 sinks and CH_4 fluxes have been shown to be unresponsive to organic matter amendments (Ryals and Silver, *in press*). I did not detect differences in CH_4 with amendment.

Potential for climate change mitigation

The potential for climate change mitigation for each model scenario was evaluated by calculating the difference between net C sequestration and net soil greenhouse gas emissions over 10, 30, and 100-year timeframes (Figure 11). Trends in net climate change mitigation potential were similar across scenarios and sites, but differed in magnitude depending on the site specific responses to C and N dynamics described above. Climate benefits of compost additions were offset partially by soil greenhouse gas emissions, particularly N_2O. While C sequestration rates were similar across sites (despite differences in the magnitude of ANPP and BNPP responses), net soil greenhouse gas emissions were greater at the valley grassland sites compared to either coastal grassland site. Therefore, estimated net mitigation potential was greater at the coastal grassland sites and highest at the coastal loam site in particular.

As expected based on examining the responses to individual variables to amendment scenarios, compost quality and application intensity played a role in determining the net climate change mitigation potential. The C:N ratio of compost amendments played an important role in regulating soil greenhouse gas losses. Greater losses were projected with lower C:N ratios, leading to lower overall rates of climate change mitigation in most cases. Application intensity played a smaller role. Typically, slightly greater mitigation rates were achieved using multiple smaller applications of the same quality and cumulative amount of composted material.

The time frame considered for climate change mitigation of the management scenarios substantially changed the determination of the source or sink potential of a given management treatment. Because the largest soil C and plant responses are observed within the first 10 – 30 years, the potential for these amended ecosystems to act as sinks is greatest when evaluated over shorter timescales. Analyzed over a 10-year time period, significant net sinks (relative to control) were observed ranging from +130 ± 3 to +158 ± 8 g CO_2-eq m^{-2} y^{-1} across sites and amendment scenarios ($p < 0.0001$ for all scenarios). Sinks were greater relative to control in this order: M11 > M20 > M30 > S20 > S11 > S30. Over a 30-year time period, all scenarios remained net sinks ($p < 0.0001$ for all scenarios). However, the net climate change mitigation potential decreased relative to the10-year timeframe, with values range from +63 ± 2 to +84 ± 10 g CO_2-eq m^{-2} y^{-1}.

Sinks were significantly greater relative to control in this order: M11 > M20 > M30 > S11 > S20 > S30. Extended further to a 100-year timeframe, no amendment scenarios were significantly different from control. Values ranging from -2.4± 13.1 to +1.4 ± 2.8 g CO_2-eq m^{-2} y^{-1} compared to control scenarios.

Discussion

Effects of model equilibrium and validation

Model predictions of soil C changes with management or climate are sensitive to initialization conditions (Foereid *et al.*, 2012). I found that soil C baseline conditions are sensitive to the model equilibrium phase due to the non-steady state conditions of most California grasslands resulting from a widespread shift from perennial to annual-dominated grasslands (Biswell, 1956; D'Antonio, 2007; Koteen *et al.*, 2011). The DAYCENT model operates with one vegetation type at a time, thus vegetation changed abruptly in model simulations. In reality, there was likely a transition phase of several years to decades. Baseline conditions of soil C decline, therefore, may be sensitive to the timing and intensity of the vegetation shift, which is not well documented at the field scale. Discrepancies between measured versus modeled values of soil C, BNPP, and ANPP can also be attributed to the occurrence of perennial grasses at the coastal grassland site and sampling variability across years.

Response of soil C and plant productivity to organic amendments

My results suggest compost decomposes very slowly (k = 0.057 y^{-1}) once applied to soil. This modeled decomposition rate is similar to values generated from field observations of a range of composted green waste, manure, and peat (k = 0.0048 to 0.21 y^{-1}) and greater than soil humus (Gerzabek *et al.*, 1997; Lynch *et al.*, 2005). The DAYCENT model adds the composted plant waste first into the slow soil organic matter pool, reflecting the partially decomposed nature of the material. From there, the added C can be humified further and incorporated into more stable soil organic matter pools or decomposed, mineralized to CO_2, or leached through the soil profile.

The DAYCENT model partitions soil C into two surface microbial pools (active and slow) and three soil organic matter pools (active, slow, and passive) defined by turnover time. It is difficult to correlate these model-defined soil C pools to laboratory fractionation data. However, the rapid positive response of the active and slow C pools to simulated amendments generally agree with laboratory fractionation methods which detected increases in the free- and occluded light fractions from soils collected three years following a single application of compost (Ryals and Silver *in press*). The peak increase in total soil C storage occurred 15 – 25 years after compost amendment in amendment scenarios. This likely reflects both the contributions from compost-C undergoing humification as well as contributions from increases plant C inputs. Surprisingly, as the short-term impacts of compost amendments to the active and slow soil C pools decreased over time, the passive soil C pool continued to increase over time. California's Mediterranean grassland ecosystems have experienced considerable loss in soil C due to a widespread shift from the dominance of perennial bunchgrass to annual grasses (Norton *et al.*, 2007; Koteen *et al.*, 2011). My modeling approach suggests that soil C at these sites still has a declining soil C baseline, which has been observed in multi-year studies conducted in similar grasslands (Chou *et al.*, 2008).

Although long-term observations of amendment response to rangelands are lacking, several short-term studies ranging from one to 14 years measured increases in total soil C as well as several indicators that suggest increased C stabilization, such as improved soil aggregation, increases in microbial biomass, and shifts in chemical quality of soil C (Pascual *et al.*, 1999; Lynch *et al.*, 2005; Sullivan *et al.*, 2006; Fernandez *et al.*, 2007a; Cabrera *et al.*, 2009; Ippolito *et al.*, 2010; Kowaljow *et al.*, 2010). Rangelands are generally N-limited. As a result, plant production typically responds positively to synthetic or organic N additions (Derner and Schuman, 2007). My modeling results suggest that both single, large and multiple, small applications of compost have an immediate and surprisingly long-lasting impact on plant productivity. Plant production responses were sensitive to compost quality with lower compost C:N ratios resulting in greater plant production.

Response of soil greenhouse gas emissions to organic matter amendments

Soil greenhouse gas emissions are sensitive to short-term changes in soil moisture and temperature conditions. In Mediterranean ecosystems, even small rain events following a dry period can result in large soil CO_2 and N_2O fluxes (Fierer and Schimel, 2002). The dynamic patterns in soil greenhouse gas emissions make it difficult to assess annual fluxes or to adequately compare treatment effects using commonly available field methods. However, changes in N_2O or heterotrophic respiration have the potential to offset gains by C sequestration (Dalal *et al.*, 2003; Conant *et al.*, 2005; Derner and Schuman, 2007). Significant increases in total soil respiration during the wet season were detected for three years following a single application of compost to valley and coastal grasslands (Ryals and Silver *submitted*). Similar results were observed in the S11 scenario. The treatment responses of heterotrophic respiration were not affected by C:N ratio, but were considerably reduced when considering multiple, small additions compared to a single, large application of compost. While autotrophic and heterotrophic components of total soil CO_2 fluxes were not explicitly measured for the field experiment, the model includes outputs for each component. According to the model, heterotrophic respiration accounted for no more than 40 % of the annual cumulative soil CO_2 flux.

No change to soil N_2O emissions were detected from the field experiment when measured once every 2 to 4 weeks. However, laboratory wet up events revealed short-lived but significant increases in the N_2O flux from amended soils (Ryals and Silver *submitted*, Chapter 1). My modeling results suggested that N_2O fluxes from all amendment scenarios were greater than those from control scenarios, and that smaller increases were generated from amendments with higher C:N ratio. This may be due to N immobilization or a reduction in the amount of available mineral N contributed by the compost amendment. I detected interesting site differences in the response of soil N_2O fluxes to amendments. Valley grasslands had greater gaseous N_2O losses compared to coastal sites, which may result from differences in soil texture. Emissions of N_2O tend to be lowest in well-drained, sandy soils where denitrification is slower, and texture controls on these characteristics are adequately modeled by DAYCENT (Del Grosso *et al.*, 2002; Del Grosso *et al.*, 2006). Conversely, NO_3^- leaching losses were larger from the coastal sandy loam site than the coastal loam site. Nitrate leaching has been shown to increase with N fertilization (Ledgard *et al.*, 1999) and urine patches from intensively grazed pastures (Di and Cameron, 2000, 2005).

Net climate change mitigation potential

My modeling results suggest that amendments, regardless of the quality or application intensity, generate net sinks consistently across grassland sites over timescales relevant to management. While net sink potential of amendments remained positive over short (10 and 30 year) timescales, the amendment scenarios reveal interesting trade-offs between C sequestration and soil greenhouse gas emissions through time. Specifically, the amendment scenarios that resulted in the greatest rates of C gains through ANPP, BNPP, and soil C storage (scenarios S11 and M11) also resulted in the greatest rates of losses of N_2O via gaseous fluxes and NO_3^- leaching. Thus, a land manager may select compost with lower C:N ratio to maximize forage production, which conflicts with climate goals of minimizing potential N_2O losses from soil.

The assessment of sink-source potential of amendments relative to control based on short (10 year), medium (30 year), and long-term (100 year) timeframes revealed the impact of amendment management through time. Positive impacts to ecosystem C storage occured immediately and last several decades, revealing the capacity of short-term management activities to have multi-decadal carry-over effects on ecosystem processes. When climate change mitigation potential was assessed over a 10-year timeframe, greenhouse gas emissions offset 5 ± 1.5 % of the benefits from C sequestration. This offset grows over time as the rate of C sequestration benefits decline. When assessed over 30 and 100-year timeframes, 13.9 ± 2.7 and 93.2 ± 7.9 % of C sequestration benefits are offset by greenhouse gas losses. These results reveal that single or short-term management events can have significant and long-lasting impacts on ecosystem C storage.

Conclusion

Sequestration of atmospheric C in rangeland soils through organic matter amendments is proposed as a way to mitigate climate change by reducing atmospheric CO_2, yet little research has been done exploring net sink potential taking other biogenic greenhouse gases into account. I used a modeling approach to test ecosystem responses to compost varying in quality and intensity of application at three grassland sites. Aboveground NPP, BNPP, and soil C pools responded rapidly and positively to all amendments scenarios. Changes to these pools were persistent for several decades, reflecting the ability of compost to act as a slow release fertilizer. Scenarios of a single versus multiple applications of compost resulted in similar changes in C and N cycling, but responses were delayed in multiple application scenarios. Compost quality significantly influenced rates of C sequestration and greenhouse gas emissions. Ecosystem C storage responses tended to be down regulated by higher C:N ratios. As a result, increases to ANPP, BNPP, and soil C pools were greatest in the S11 and M11 amendment scenarios and lowest in the S30 and M30 scenarios. Soil greenhouse gas emissions, particularly direct soil N_2O fluxes and indirect losses through NO_3^- leaching, partially offset C sequestration benefits. Highest rates of soil greenhouse gas emissions were observed in the S11 scenario, where compost with a low C:N ratio was added as a single event. Relative to the S11 scenario, N_2O emissions were reduced if using compost with higher C:N ratio or if compost was applied in multiple, small applications. These results suggest there is a trade-off between maximizing production increases and minimizing soil N_2O emissions. All amendment scenarios indicate that short-term compost amendments have considerable potential for mitigating climate change over many decades.

References

Biswell, H.H., 1956. Ecology of California grasslands. Jour Range Management 9, 19-24.

Blair, N., Faulkner, R.D., Till, A.R., Poulton, P.R., 2006. Long-term management impacts on soil C, N and physical fertility - Part 1: Broadbalk experiment. Soil & Tillage Research 91, 30-38.

Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002. Emissions of N2O and NO from fertilized fields: Summary of available measurement data. Global Biogeochemical Cycles 16.

Cabrera, V.E., Stavast, L.J., Baker, T.T., Wood, M.K., Cram, D.S., Flynn, R.P., Ulery, A.L., 2009. Soil and runoff response to dairy manure application on New Mexico rangeland. Agriculture Ecosystems & Environment 131, 255-262.

CCQC, 2001. Compost Maturity Index. California Compost Quality Council., Nevada City, CA, p. 26.

Chang, C., Cho, C.M., Janzen, H.H., 1998. Nitrous oxide emission from long-term manured soils. Soil Science Society of America Journal 62, 677-682.

Chou, W.W., Silver, W.L., Jackson, R.D., Thompson, A.W., Allen-Diaz, B., 2008. The sensitivity of annual grassland carbon cycling to the quantity and timing of rainfall. Global Change Biology 14, 1382-1394.

Conant, R.T., Paustian, K., Del Grosso, S.J., Parton, W.J., 2005. Nitrogen pools and fluxes in grassland soils sequestering carbon. Nutrient Cycling in Agroecosystems 71, 239-248.

Conant, R.T., Paustian, K., Elliott, E.T., 2001. Grassland management and conversion into grassland: Effects on soil carbon. Ecological Applications 11, 343-355.

D'Antonio, C.M., 2007. Ecology of invasive non-native species in California grasslands. In: Stromberg, M.R., Corbin, J.D., D'Antonio, C.M. (Eds.), California Grasslands: Ecology and Management. UC Press, Berkeley, CA.

Dalal, R.C., Wang, W.J., Robertson, G.P., Parton, W.J., 2003. Nitrous oxide emission from Australian agricultural lands and mitigation options: a review. Australian Journal of Soil Research 41, 165-195.

Dean, C., Wardell-Johnson, G.W., Harper, R.J., 2012. Carbon management of commercial rangelands in Australia: Major pools and fluxes. Agriculture Ecosystems & Environment 148, 44-64.

Del Grosso, S., Ojima, D., Parton, W., Mosier, A., Peterson, G., Schimel, D., 2002. Simulated effects of dryland cropping intensification on soil organic matter and greenhouse gas exchanges using the DAYCENT ecosystem model. Environmental Pollution 116, S75-S83.

Del Grosso, S.J., Mosier, A.R., Parton, W.J., Ojima, D.S., 2005. DAYCENT model analysis of past and contemporary soil N(2)O and net greenhouse gas flux for major crops in the USA. Soil & Tillage Research 83, 9-24.

Del Grosso, S.J., Parton, W.J., Mosier, A.R., Walsh, M.K., Ojima, D.S., Thornton, P.E., 2006. DAYCENT national-scale simulations of nitrous oxide emissions from cropped soils in the United States. Journal of Environmental Quality 35, 1451-1460.

DeLonge, M., Ryals, R., Silver, W.L. in review. Carbon sequestration potential and greenhouse gas dynamics of managed graasslands. Ecosystems.

Derner, J.D., Schuman, G.E., 2007. Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. Journal of Soil and Water Conservation 62, 77-85.

Di, H.J., Cameron, K.C., 2000. Calculating nitrogen leaching losses and critical nitrogen application rates in dairy pasture systems using a semi-empirical model. New Zealand Journal of Agricultural Research 43, 139-147.

Di, H.J., Cameron, K.C., 2005. Reducing environmental impacts of agriculture by using a fine particle suspension nitrification inhibitor to decrease nitrate leaching from grazed pastures. Agriculture Ecosystems & Environment 109, 202-212.

EPA, 2012. Inventory of US Greenhouse Gas Emissions and Sinks: 1990 - 2010. US Environmental Protection Agency, Washington, DC, p. 481.

Fernandez, J.M., Hernandez, D., Plaza, C., Polo, A., 2007. Organic matter in degraded agricultural soils amended with composted and thermally-dried sewage sludges. Science of the Total Environment 378, 75-80.

Fierer, N., Schimel, J.P., 2002. Effects of drying-rewetting frequency on soil carbon and nitrogen transformations. Soil Biology & Biochemistry 34, 777-787.

Foereid, B., Bellamy, P.H., Holden, A., Kirk, G.J.D., 2012. On the initialization of soil carbon models and its effects on model predictions for England and Wales. European Journal of Soil Science 63, 32-41.

Follett, R.F., 2001. Soil management concepts and carbon sequestration zin cropland soils. Soil & Tillage Research 61, 77-92.

Gagnon, B., Simard, R.R., Goulet, M., Robitaille, R., Rioux, R., 1998. Soil nitrogen and moisture as influenced by composts and inorganic fertilizer rate. Canadian Journal of Soil Science 78, 207-215.

Gerzabek, M.H., Pichlmayer, F., Kirchmann, H., Haberhauer, G., 1997. The response of soil organic matter to manure amendments in a long-term experiment at Ultuna, Sweden. European Journal of Soil Science 48, 273-282.

Gregorich, E.G., Rochette, P., VandenBygaart, A.J., Angers, D.A., 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil & Tillage Research 83, 53-72.

Harpole, W.S., Potts, D.L., Suding, K.N., 2007. Ecosystem responses to water and nitrogen amendment in a California grassland. Global Change Biology 13, 2341-2348. IPCC, 2007. Climate Change 2007: The Physical Science Basis. Contributions of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., M., T., Miller, H.L. (Eds.), Cambridge, United Kingdom, p. 996.

Ippolito, J.A., Barbarick, K.A., Paschke, M.W., Brobst, R.B., 2010. Infrequent composted biosolids applications affect semi-arid grassland soils and vegetation. Journal of Environmental Management 91, 1123-1130.

Kelly, R.H., Parton, W.J., Hartman, M.D., Stretch, L.K., Ojima, D.S., Schimel, D.S., 2000. Intra-annual and interannual variability of ecosystem processes in shortgrass steppe. Journal of Geophysical Research-Atmospheres 105, 20093-20100.

Koteen, L.E., Baldocchi, D.D., Harte, J., 2011. Invasion of non-native grasses causes a drop in soil carbon storage in California grasslands. Environmental Research Letters 6.

Kowaljow, E., Julia Mazzarino, M., Satti, P., Jimenez-Rodriguez, C., 2010. Organic and inorganic fertilizer effects on a degraded Patagonian rangeland. Plant and Soil 332, 135-145.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma 123, 1-22.

Ledgard, S.F., Penno, J.W., Sprosen, M.S., 1999. Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. Journal of Agricultural Science 132, 215-225.

Li, X.Y., Meixner, T., Sickman, J.O., Miller, A.E., Schimel, J.P., Melack, J.M., 2006. Decadal-scale dynamics of water, carbon and nitrogen in a California chaparral ecosystem: DAYCENT modeling results. Biogeochemistry 77, 217-245.

Lynch, D.H., Voroney, R.P., Warman, P.R., 2005. Soil physical properties and organic matter fractions under forages receiving composts, manure or fertilizer. Compost Science & Utilization 13, 252-261.

Norton, J.B., Monaco, T.A., Norton, U., 2007. Mediterranean annual grasses in western North America: kids in a candy store. Plant and Soil 298, 1-5.

Parton, W.J., Hartman, M., Ojima, D., Schimel, D., 1998. DAYCENT and its land surface submodel: description and testing. Global and Planetary Change 19, 35-48.

Parton, W.J., Holland, E.A., Del Grosso, S.J., Hartman, M.D., Martin, R.E., Mosier, A.R., Ojima, D.S., Schimel, D.S., 2001. Generalized model for NOx and N2O emissions from soils. Journal of Geophysical Research-Atmospheres 106, 17403-17419.

Parton, W.J., Morgan, J.A., Wang, G.M., Del Grosso, S., 2007. Projected ecosystem impact of the Prairie Heating and CO2 Enrichment experiment. New Phytologist 174, 823-834.

Parton, W.J., Ojima, D.S., Schimel, D.S., 1994. Environmental-change in grasslands – assessment using models. Climatic Change 28, 111-141.

Pascual, J.A., Garcia, C., Hernandez, T., 1999. Lasting microbiological and biochemical effects of the addition of municipal solid waste to an arid soil. Biology and Fertility of Soils 30, 1-6.

Paustian, K., Andren, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tiessen, H., Van Noordwijk, M., Woomer, P.L., 1997. Agricultural soils as a sink to mitigate CO2 emissions. Soil Use and Management 13, 230-244.

Powlson, D.S., Bhogal, A., Chambers, B.J., Coleman, K., Macdonald, A.J., Goulding, K.W.T., Whitmore, A.P., 2012. The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study. Agriculture Ecosystems & Environment 146, 23-33.

Ryals, R., Silver, W.L., in press. Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grassland ecosystems. Ecological Applications.

Saxton, K.E., Rawls, W.J., Romberger, J.S., Papendick, R.I., 1986. Estimating generalized soil-water characteristics from texture. Soil Science Society of America Journal 50, 1031-1036.

Silver, W.L., Ryals, R., Eviner, V., 2010. Soil Carbon Pools in California's Annual Grassland Ecosystems. Rangeland Ecology & Management 63, 128-136.

Sullivan, T.S., Stromberger, M.E., Paschke, M.W., Ippolito, J.A., 2006. Long-term impacts of infrequent biosolids applications on chemical and microbial properties of a semi-arid rangeland soil. Biology and Fertility of Soils 42, 258-266.

Thornton, P.E., Thornton, M.M., Mayer, B.W., Wilhelmi, N., Wei, Y., Cook, R.B., 2012. Daymet: Daily surface weather on a 1 km grid for North America, 1980 - 2008. Oak Ridge National Laboratory Distributed Active Archive Center, Oak Ridge, Tennessee.

Tables

Table 1. Descriptions of organic matter amendment scenarios simulated with the DAYCENT model parameterized to three rangeland sites. * indicates simulation of treatments included in a three-year field experiment replicated within a valley and coastal grassland.

Scenario Name	Description
Control *	No organic matter amendments Moderate spring grazing
S11 *	Single large application of compost (14.27 Mg C ha^{-1}) with C : N ratio of 11.1. Moderate spring grazing
S20	Single large application of compost (14.27 Mg C ha^{-1}) with C : N ratio of 20. Moderate spring grazing
S30	Single large application of compost (14.27 Mg C ha^{-1}) with C : N ratio of 30. Moderate spring grazing
M11	Multiple small applications of compost (1.427 Mg C ha^{-1} y^{-1} for 10 consecutive years) with C : N ratio of 11. Moderate spring grazing
M20	Multiple small applications of compost (1.427 Mg C ha^{-1} y^{-1} for 10 consecutive years) with C : N ratio of 20. Moderate spring grazing
M30	Multiple small applications of compost (1.427 Mg C ha^{-1} y^{-1} for 10 consecutive years) with C : N ratio of 30. Moderate spring grazing

Figures

Figure 1. Historical weather records from (a) valley grassland and (b) coastal grassland. Bars show historical mean monthly precipitation; Solids circles show mean maximum air temperature, and open circles show mean minimum air temperature. Error bars are ± 1 standard deviation. Data from the valley grassland were collected at Browns Valley climate station since 1990. Precipitation data at the coastal grassland were collected from the Marin County Water District in Nicasio, CA (< 1 km from site) since 1975. Air temperature data at the coastal grassland were modeled using DayMet since 1980.

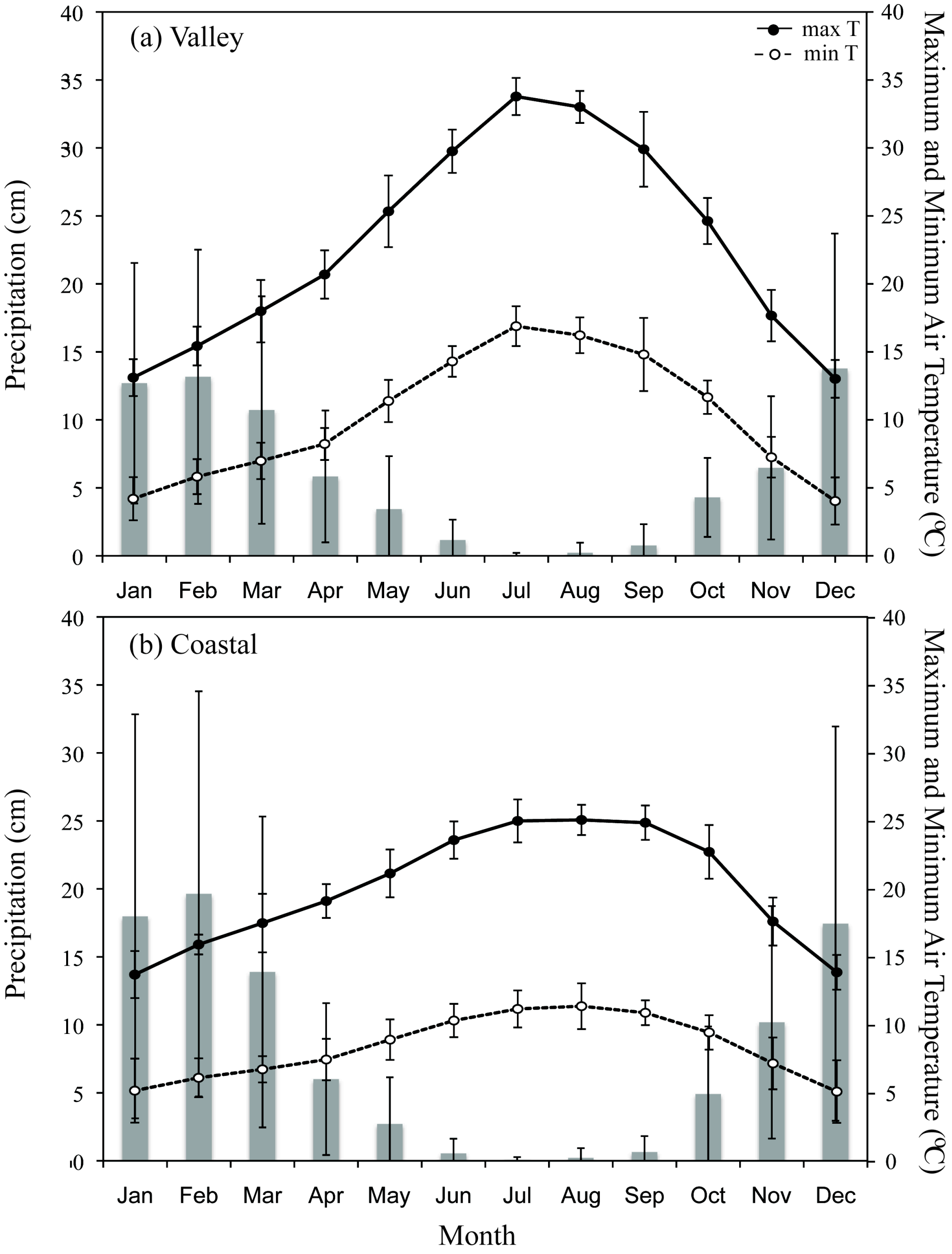


Figure 2. Simulated soil C (0 – 20 cm) during model spin up at the valley grassland. A similar pattern is observed with simulations at coastal grassland sites. DayCent was initiated with perennial grasses from Year 1 – 1749, followed by a shift to annual grasses after 1750.

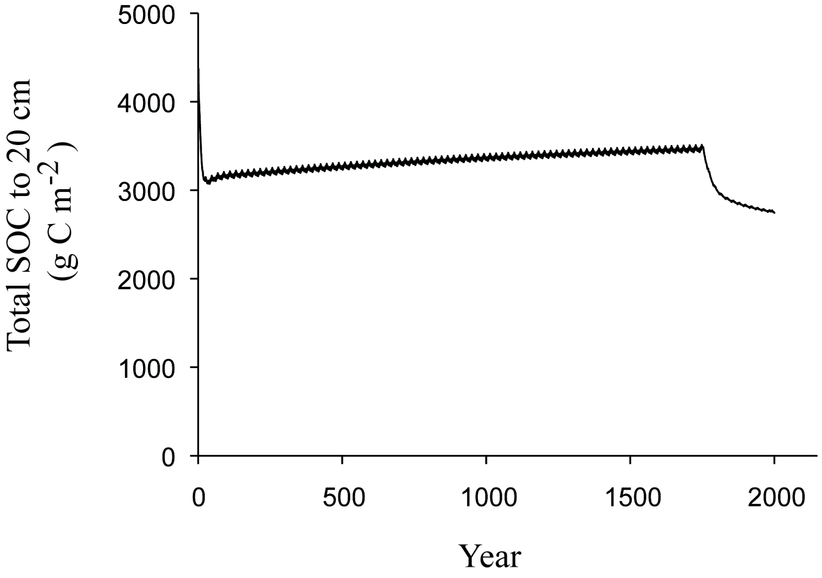


Figure 3. DAYCENT model validation as determined by the correlation between measured and modeled data. Validation relationships include valley and coastal grassland plot-level (a) 0-20 cm SOC, (b) ANPP, (b) BNPP, and (d) annual soil respiration (heterotrophic + autotrophic components) from 2009 – 2011. Red shades represent amended data. Blue shades represent control data. Circles are from valley plots, and squares are from coastal plots. * = analysis excludes coastal loam sites where measured SOC was consistently greater than modeled SOC.

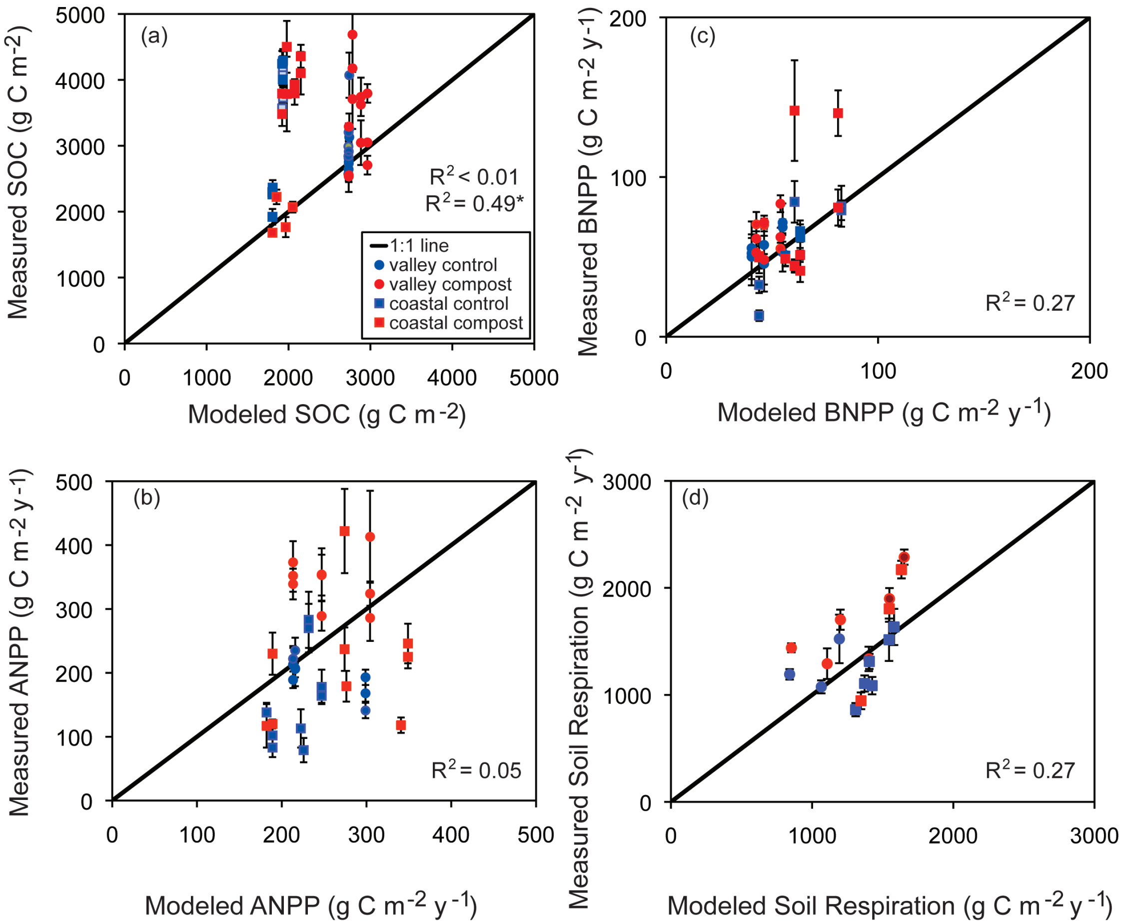


Figure 4. Simulated decomposition of the organic matter amendment following application to grassland soils.

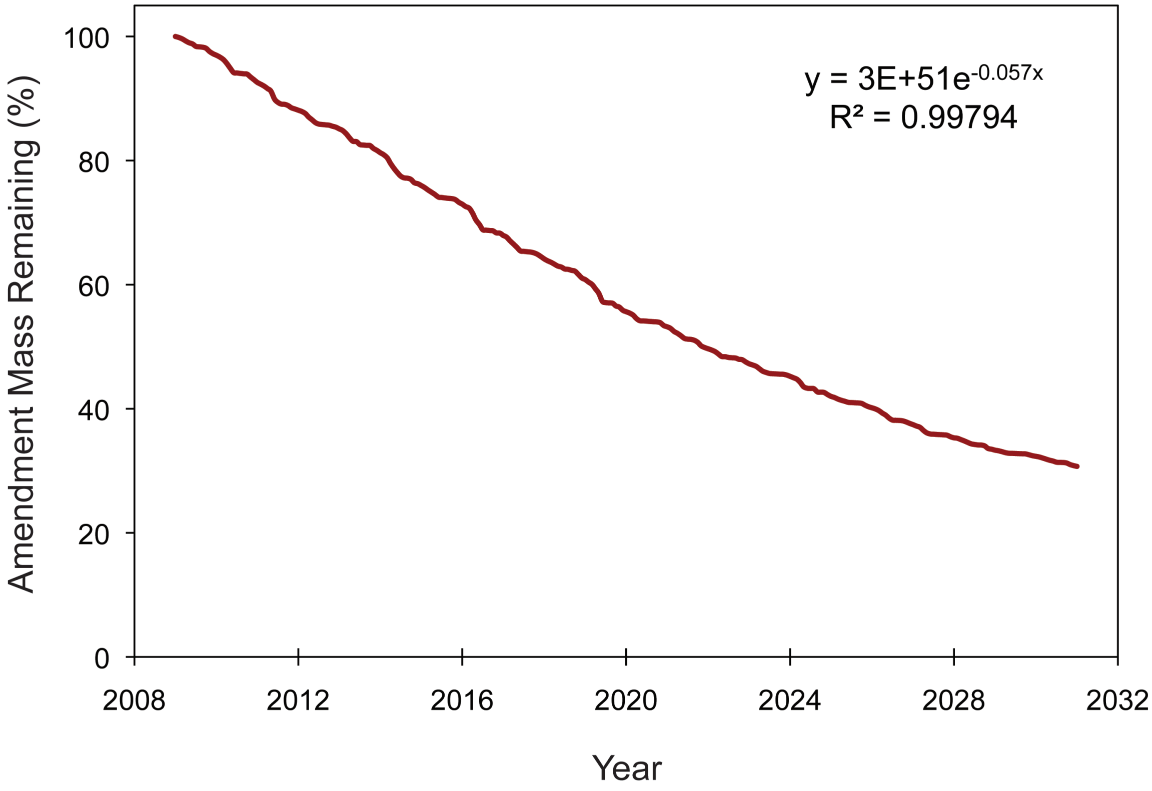


Figure 5. (a) Active, (b) slow, and (c) passive soil C pools in the DAYCENT model from control and S11 scenarios for the valley, coastal – loam, and coastal – sandy loam sites. (d) Treatment differences from control for each soil C pools is shown in units of g C m^{-2}.

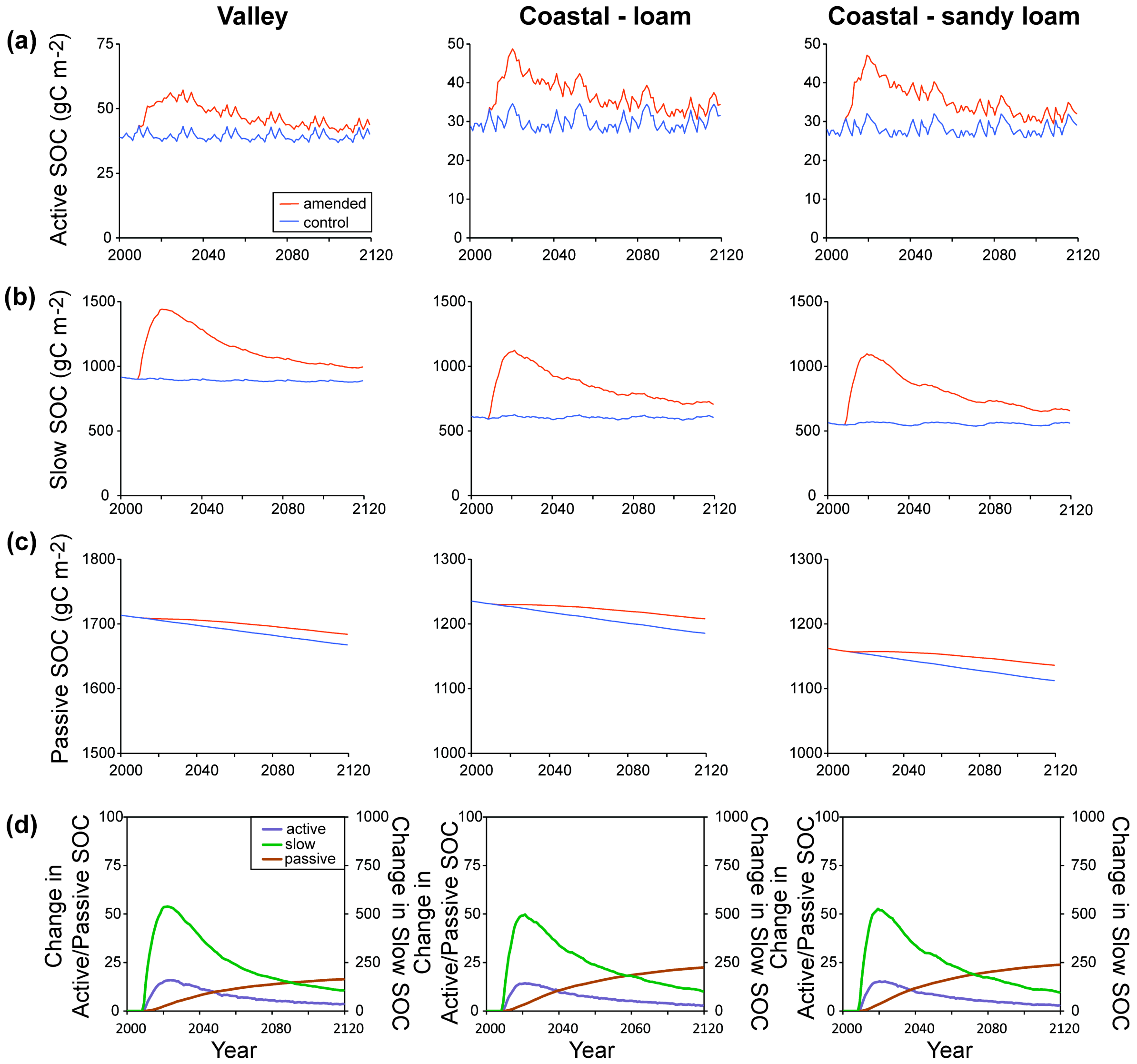


Figure 6. Treatment differences from control of (a) annual aboveground net primary production and (b) cumulative change in aboveground production over 10, 30 and 100 years under a range of amendment scenarios. S = single application of composted green waste (14.27 Mg C ha^{-1}). M = multiple annual applications of composted green waste (1.427 Mg C ha^{-1} y^{-1} for 10 consecutive years). 11 indicates C:N ratio of 11.1. 20 indicates C:N ratio of 20. 30 indicates C:N ratio of 30. Values above 0 indicate greater aboveground C in amendment scenario versus control.

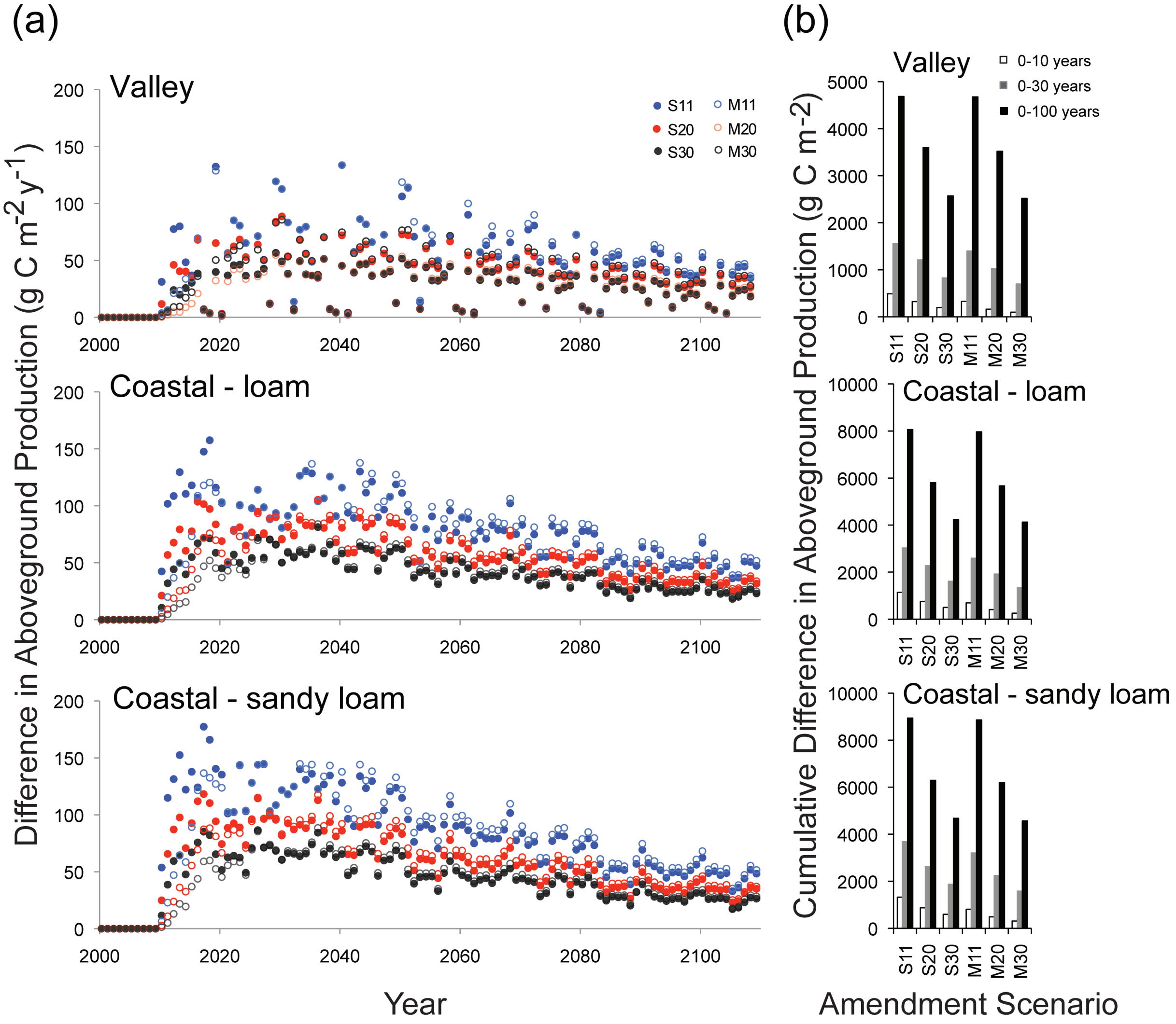


Figure 7. Treatment differences from control of (a) annual belowground net primary production and (b) cumulative change in belowground production over 10, 30 and 100 years under a range of amendment scenarios. S = single application of composted green waste (14.27 Mg C ha^{-1}). M = multiple annual applications of composted green waste (1.427 Mg C ha^{-1} y^{-1} for 10 consecutive years). 11 indicates C:N ratio of 11.1. 20 indicates C:N ratio of 20. 30 indicates C:N ratio of 30. Values above 0 indicate greater belowground C in amendment scenario versus control.

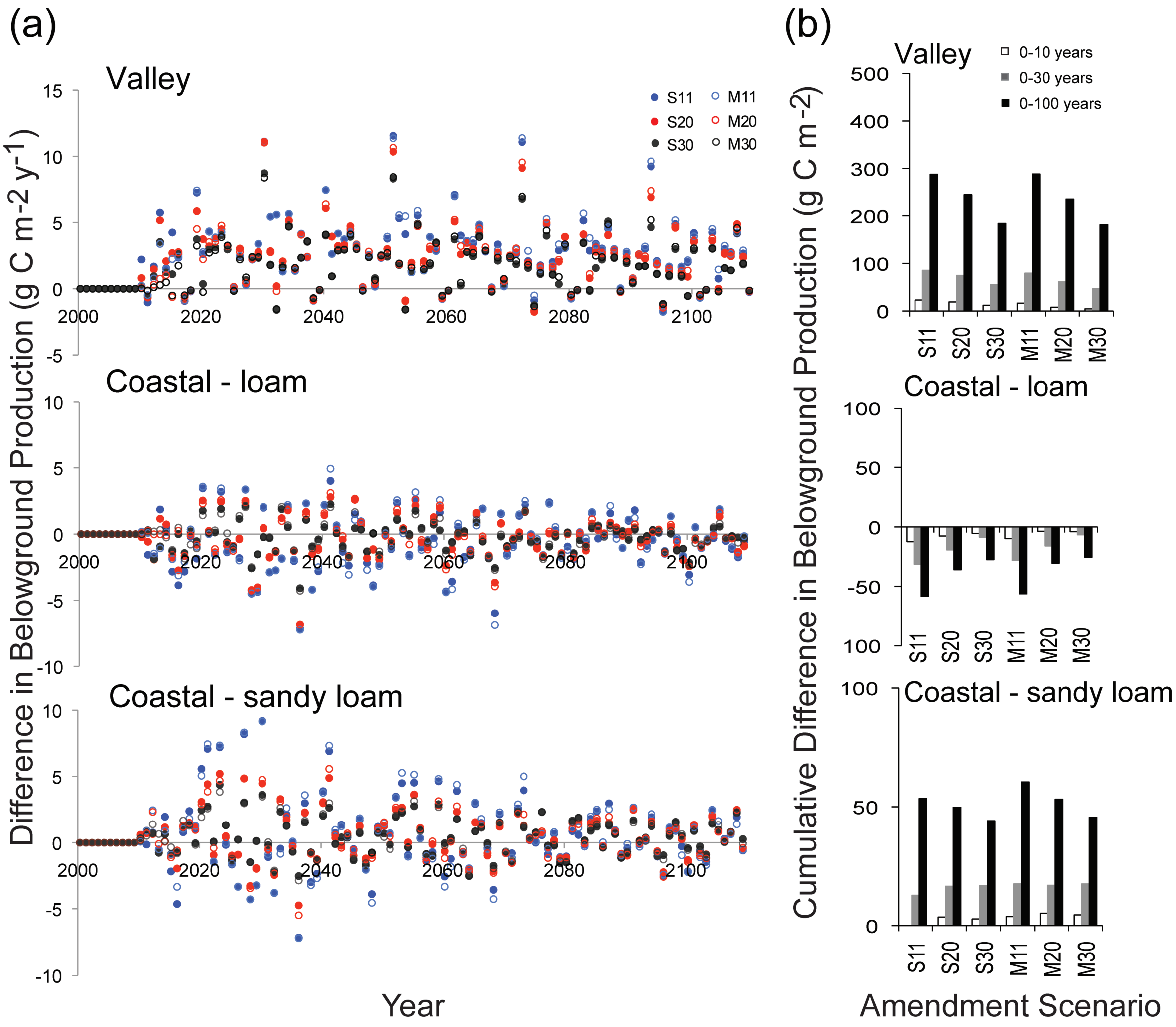


Figure 8. Changes to soil C pools with a range of soil amendment scenarios from the valley grassland simulations. Soil C pools are defined in the DAYCENT model by their turnover times: 1-5 years for the active pool, 20-40 years for the slow pool, and 200-1500 years for the passive pool. Values above zero indicate greater soil C in amendment scenario versus control.

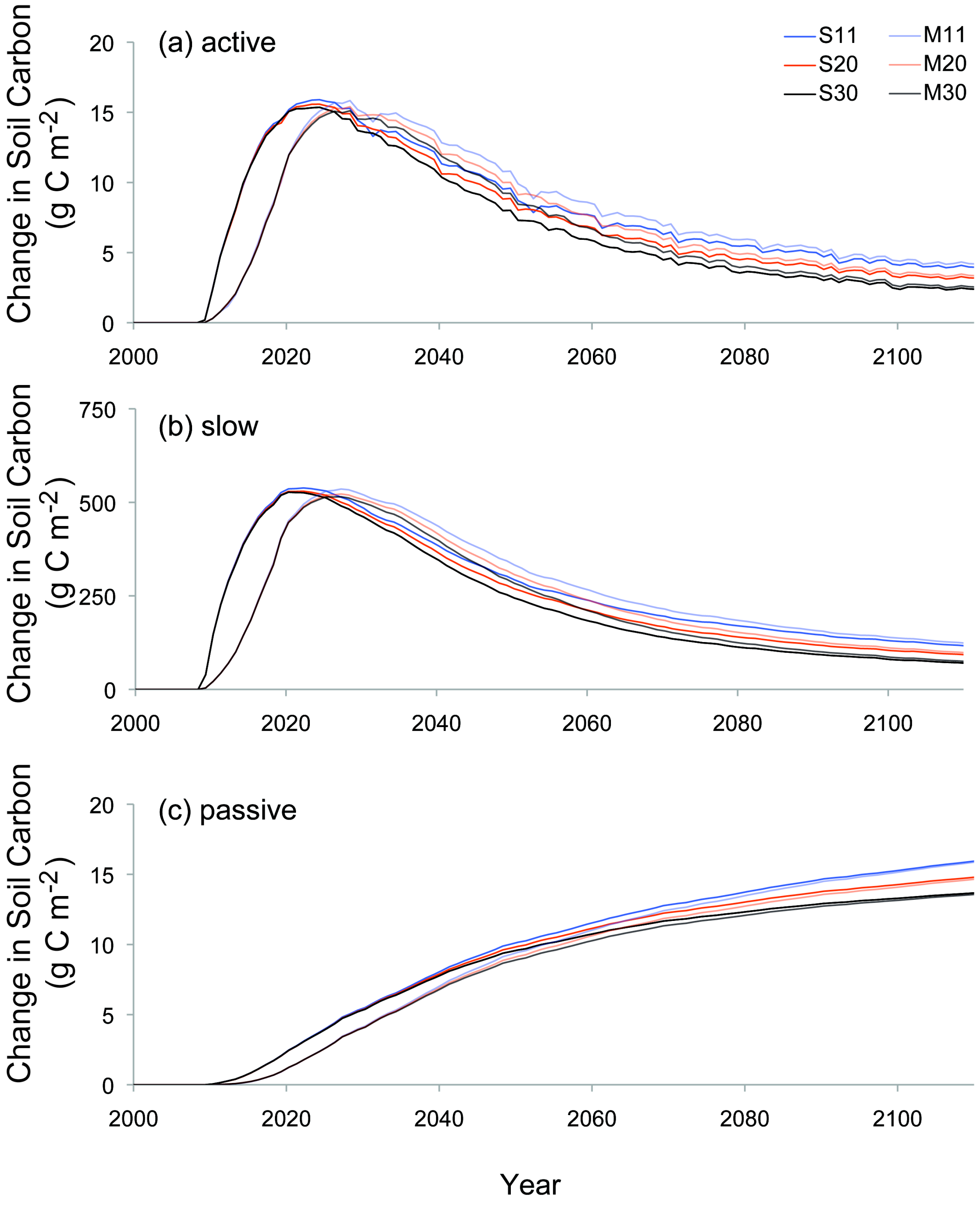


Figure 9. Treatment differences from control in daily soil N_2O, CH_4, and heterotrophic respiration for three years. Positive values indicate greater gaseous loss to the atmosphere from amended soils compared to control soils. For illustrative purposes, only scenarios S11, M11, and S30 are shown here.

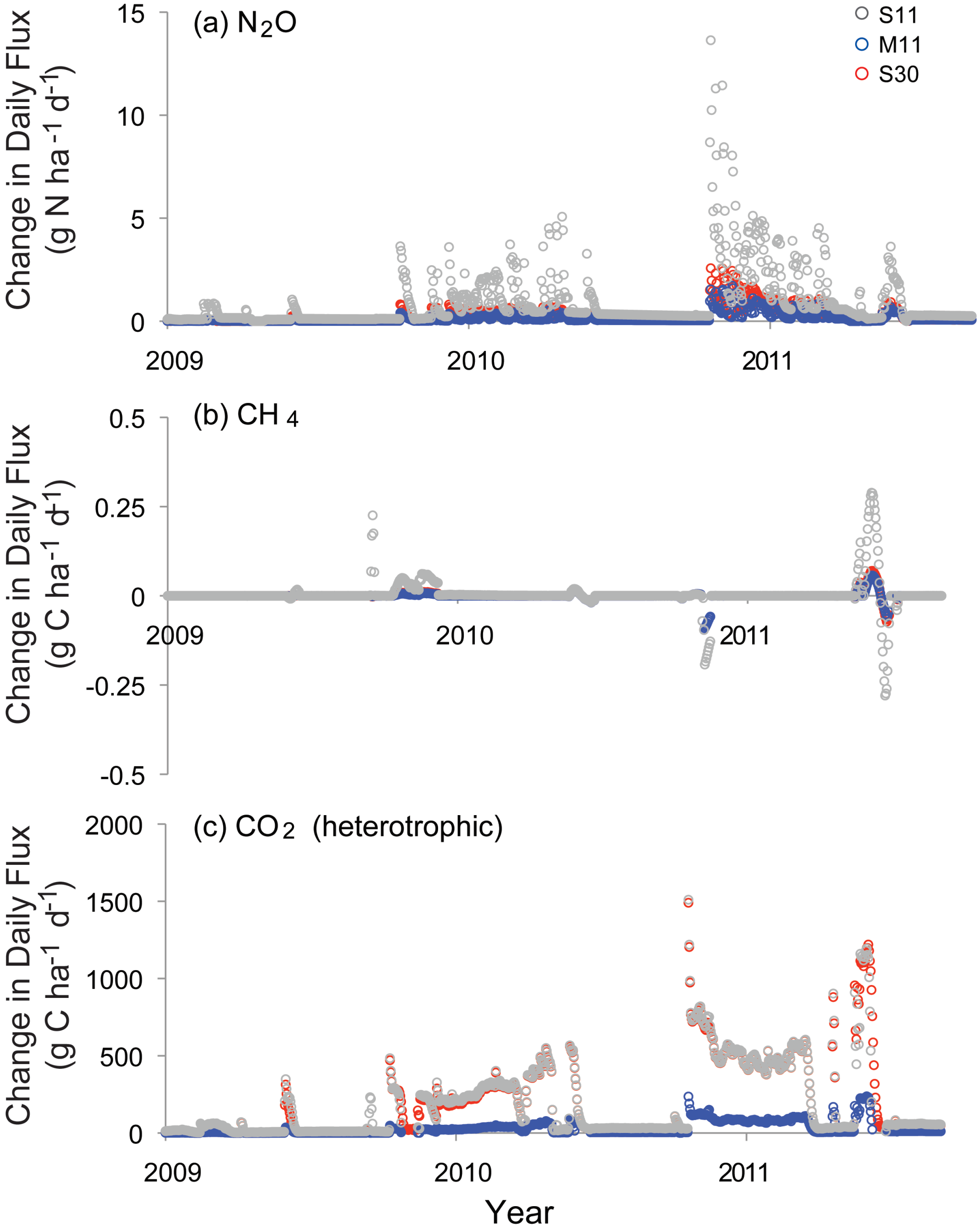


Figure 10. Treatment differences from control of (a) annual NO_3^- leaching and (b) cumulative change in NO_3^- leaching over 10, 30 and 100 years under a range of amendment scenarios. Positive values indicate greater loss of N via leaching from amended soils compared to control soils.

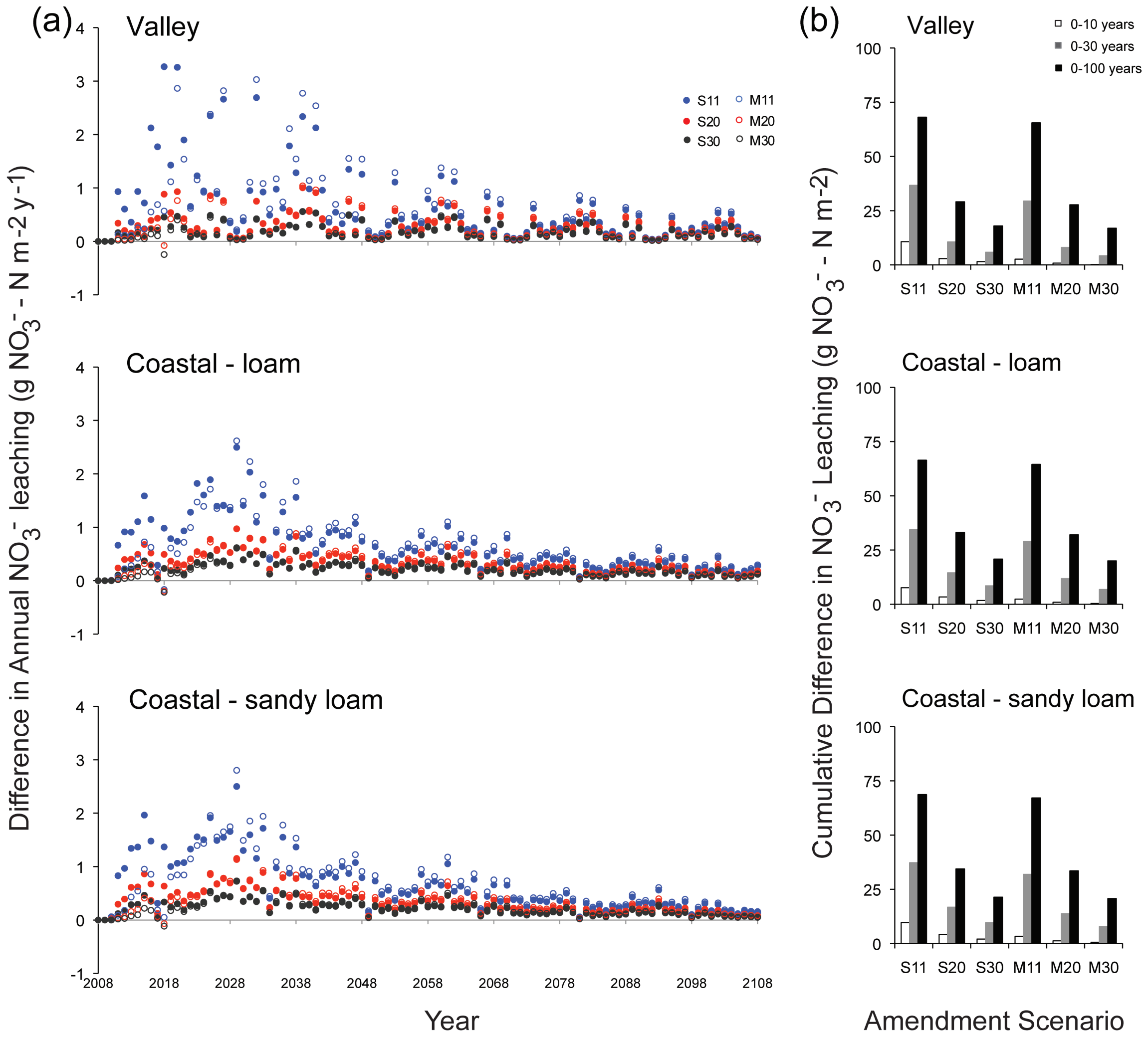
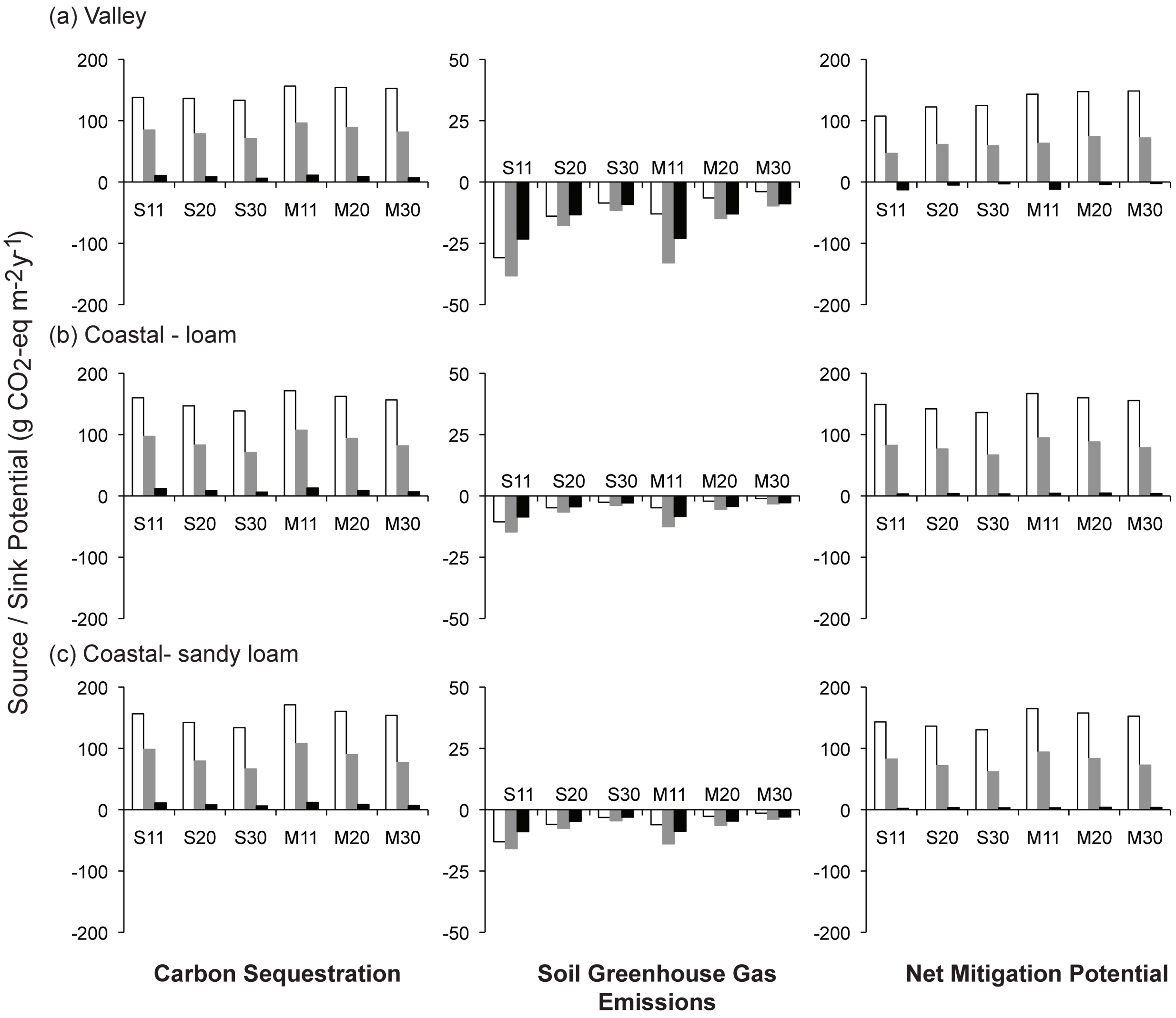


Figure 11. Potential for climate change mitigation due to C sequestration, changes in soil greenhouse gas emissions, and net climate change mitigation for a range of organic matter amendment scenarios based on DAYCENT simulations listed in Table 1. Negative values indicate a net source of CO_2-eq to the atmosphere. Positive values indicate that the ecosystem is a net sink of CO_2-eq.



Chapter 4: Are there tradeoffs in vegetation characteristics in rangelands managed for carbon sequestration?

Abstract

Rangeland management practices to increase carbon sequestration can have secondary affects on plant community dynamics such as aboveground biomass, plant chemistry, floristic diversity, or the abundance of noxious weed species. I assessed impacts of a one-time application of compost application with and without and keyline plowing on plant dynamics in two grazed grassland ecosystems in California: a coastal grassland and valley grassland. I observed large increases in aboveground biomass and plant N content following compost amendment, which persisted for four years. There were no major shifts in species richness or abundance at either grassland site. The valley grassland was dominated by exotic annual grasses and was less species rich than the coastal grassland, which consisted of a mix of perennial and annual grasses and forbs. Overall, plant communities at both grasslands were relatively resistant to these management events, but responses of some individual species were observed. Notably, we detected a short-term positive response of a noxious forb (*C. lanatus*) at the coastal grassland and a negative response of a noxious annual grass (*T. caput-medusae*) at the valley grassland to a combination of compost application and keyline plowing. Comparing two common vegetation survey methods, we found that the cover class method produced higher estimates of species richness and of diminutive and less common species compared to the point intercept method. Our results suggest that a single application of the keyline plow produces short-term (< 1 y) tradeoffs in aboveground biomass and noxious weeds, and that a single application of compost amendments produced sustained increases in forage production and nutritional quality without greatly affecting the floristic communities of these exotic annual grasslands.

Introduction

Rangelands have the potential to provide valuable ecosystem services, including forage production for livestock grazing, soil carbon (C) sequestration, and floristic diversity (Sala and Paruelo, 1997; Herrero *et al.*, 2009). Management interventions that aim to increase one type of service may have secondary effects that negatively impact other (Chan *et al.*, 2011; Fisher *et al.*, 2011). Rangelands occupy a large geographic area and are thought to have considerable capacity to increase global soil C storage even at low rates of sequestration (Lal, 2004). As a result, there has been appreciable recent interest in managing rangelands for climate change mitigation via C sequestration (Betts, 2003; Smith, 2004; Derner and Schuman, 2007; Lipper *et al.*, 2010; Morgan *et al.*, 2010; Conant *et al.*, 2011; Dean *et al.*, 2012; Srivastava *et al.*, 2012). In these ecosystems, the primary mechanism for C sequestration is increased C inputs to the soil through enhanced plant production or belowground allocation (Conant *et al.*, 2001) Yet, few studies have explored potential tradeoffs or co-benefits associated with strategies that promote C sequestration (Olander *et al.*, 2012).

Compost amendments and keyline plowing are two techniques that have been proposed for soil C sequestration in agricultural and rangelands (Yeomans, 1954; Cabrera *et al.*, 2009; Ryals and Silver, in press). Both strategies are assumed to increase plant production. Compost amendments can increase nitrogen (N) and water availability and thus stimulate net primary production (Eghball and Power, 1999; Ippolito *et al.*, 2010; Kowaljow *et al.*, 2010; McFarland *et al.*, 2010; D'Hose *et al.*, 2012; Powlson *et al.*, 2012). Keyline plowing, often used in permaculture and holistic management, is thought to promote plant growth by redistributing water resources (Mollison, 1988). The impact of these practices on net primary production (NPP), plant community structure and forage nutritional quality are not well known, but critical for understanding tradeoffs between rangeland C sequestration and economic sustainability.

Increases in C storage, especially if achieved by boosting plant production, may change species diversity and the abundance of noxious weeds (DiTomaso, 2000; Lemaire, 2012). Excessive loading of N to grassland ecosystems has been identified as a primary mechanism responsible for the overall loss of plant species (Stevens *et al.*, 2004; Bobbink *et al.*, 2010; De Schrijver *et al.*, 2011) and increased abundance of invasive species (Young *et al.*, 1998; DiTomaso, 2000; Corbin and D'Antonio, 2004b; Blumenthal, 2005; D'Antonio, 2007). Physical disturbance to soil can also encourage plant invasions because it creates vacant niches that alien species can occupy (Masters and Sheley, 2001). Noxious plant species decrease the sustainability of rangelands (DiTomaso, 2000), and generally reduce the capacity of rangelands to provide others goods and services (Masters and Sheley, 2001).

In California, rangelands occupy approximately 23 million hectares, of 40 % of the state’s land area (Brown *et al.*, 2004). It is thought that the plant communities in these ecosystems were once dominated by perennial bunchgrasses (Bartolome *et al.*, 1986). In the 19th century, the region experienced intensification of grazing, severe drought, and the introduction of seeds from European Mediterranean regions. The conditions led to a widespread shift from perennial to annual-dominated communities (Biswell, 1956). Exotic annual grasses are opportunistic and thrive in resource-pulse-driven environments. These invisibility traits have led to their persistence and altered nutrient cycling that prevent establishment or recovery of perennial grasses (D'Antonio, 2007). In more recent decades, secondary invaders, such as *Taeniatherum caput-medusae* (medusahead), have spread throughout exotic annual plant communities and further disrupt the resiliency of annual grassland ecosystems ((Young *et al.*, 1969; Young and Evans, 1971).

Determining changes to vegetation community composition, species richness, or the abundance of particular species may be sensitive to survey methodology (Kirby *et al.*, 1986; Kercher *et al.*, 2003). Cover class methods score percent cover of each individual species present within a given area into a predetermined percentage range, and have been used to detect temporal changes in communities (Mitchell *et al.*, 1988). The subjective nature and associated bias between observers of the cover class approach has been questioned (Leps and Hadincova, 1992). In contrast, point intercept methods calculate percent cover as a proportion of hits by theoretically dimensionless points arranged on a grid or transect. While this approach is more objective, some researchers argue that point or line intercept methods do not accurately assess cover in a heterogeneous stand of herbaceous species (McCune and Grace, 2002). Despite the potential pitfalls of applying either method to grassland studies, few studies compare results or efficacy of these common methods (Sykes *et al.*, 1983; Kercher *et al.*, 2003). The extent to which methodology approaches influence the evaluation of C sequestration management effects on grassland plant communities is largely unknown.

In this study, we investigated the separate and combined effects of compost amendments and keyline plowing on plant growth, community composition, and plant N and protein content in two rangeland ecosystems in California. Management approaches were applied at the beginning of the first growing season, and followed for three years. I tested the hypothesis that compost amendments and keyline plowing increases plant production and nutritional quality by increasing nutrient and water availability to plants. To test this hypothesis, forage quantity and quality were measured for four years. I also hypothesized that these heavily invaded grasslands would be resistant to shifts in community composition following these management events because invaded grasses thrive in the resource pulse-driven conditions and native grasses are unlikely to reestablish without intervention. To test this hypothesis, we determined whether management events altered the abundance, richness, and diversity of grasses, forbs, legumes, and bulbs. I also focused specifically on two prominent invaders of rangelands in the western US: *Carthamus lanatus*, an annual forb (woolly distaff thistle) and *Taenthaneum caput-medusae*, an annual grass (medusahead). Finally, we compared the results of two common vegetation survey approaches, the line intercept and cover class methods.

Methods

Study sites

The study was conducted on valley grasslands at the U.C. Sierra Foothill Research and Extension Center (Browns Valley, CA; 39.2 ºN, 121.3 ºW) and on privately-owned coastal grasslands (Nicasio, CA; 38.1 ºN, 122.7 ºW). Annual rainfall amounts at the valley grassland during experimental years were 380, 641, 843, and 549 mm/y, compared to a 22 y mean of 720 mm/yr. At the coastal grassland, annual rainfall was 771, 1050, 1163, and 778 mm/y, compared to a 38 y mean of 950 mm/y. Mean air temperatures range from 2 °C in January to 35 °C in July at the valley grassland and from 6 °C in January to 20 °C in July at the coastal grassland. Valley grassland soils are xeric inceptisols and alfisols in the Auburn-Sobrante complex (Beaudette and O'Geen 2009, http://casoilresource/lawr.ucdavis.edu/soilsurvey). Coastal grassland soils are mollisols in the Tocaloma-Saurin-Bonnydoon series (Beaudette and O'Geen 2009, http://casoilresource/lawr.ucdavis.edu/soilsurvey). Both sites are grazed by cattle using rotational grazing to 130 g/m^2 residual dry matter.

Experimental design

Treatments were applied one-time in December 2008 and consisted of untreated control, organic matter amendment, keyline plow, and amendment + keyline plow. Plots were 25 m by 60 m buffered by a ≥ 5 m strip arranged in three randomized complete blocks at each site. The organic matter amendment consisted of a single application of commercially available composted organic green waste (Feather River Organics, Marysville, CA) with a N concentration of 1.87 % and a C:N ratio of 11. Compost was applied as a thin surface dressing approximately 1.3 cm thick, equivalent to 1.42 kg C m^{-2}, 129 g total N m^{-2}, and 7.0 kg dry matter m^{-2}. Soils were plowed with a vertical subsoiler (Yeomans Plows, Queensland, Australia) extending 10 cm below rooting depth (approximately 30 cm depth) and moved along the contour of the hillside according to traditional keyline methods (Yeomans, 1954).

Measurements of aboveground biomass and forage quality

All plots were grazed during the spring for up to four weeks; the timing was dependent on the amount of available forage. Valley grassland plots were grazed February through March, while coastal grassland plots were grazed later in the growing season, ranging from mid-April through mid-June. During grazing, cattle were not isolated within plots, but instead allowed to graze the entire block.

Aboveground plant biomass was measured no more than 24 hours prior to grazing events. Aboveground biomass was determined by clipping vegetation to the soil surface from a 200 cm^2 area (n = 9 per plot). Biomass samples were then dried at 65 °C and weighed. Aboveground biomass was also measured no more than 24 hours after removal of cows from plots to assess grazing impacts. I estimated the amount of forage consumed by cows as the difference between pre-grazed and post-grazed biomass.

Plant N content and N concentration served as indicators of forage nutritional quality. Forage quality measurements were made on biomass samples collected < 24 hours prior to grazing events in order to better relate rangeland management practices to cattle production. Biomass samples were ground finely and analyzed for N concentration on a Carlo Erba Elantech elemental analyzer (Lakewood, NJ). Nitrogen content of aboveground biomass was determined by multiplying N concentration of each biomass sample by its mass and expressed in units of mass per area. I recognize that forages harvested at early growth stages generally have higher plant N content than mature forages harvested at flowering stages (Stokes and Prostko, 1998). Due to differences in the timing of grazing, we compared treatment effects on forage quality within sites and did not compare between sites. For the plowed plots, forage quality was assessed only for years 1 and 2, as no treatment difference from control, respectively, were detected.

Measurements of plant diversity and abundance

Plant diversity indices were estimated from annual vegetation surveys at the end of each growing season from 2009 – 2012. Vegetation surveys were conducted using point intercept and cover class methods, with the goal of comparing the effectiveness of detecting treatment effects using these two common approaches. Two 40 m transects were set diagonally in each plot (Figure 1). For the point intercept method, a pin was placed every 0.5 m along each transect. All individuals touching the pin were identified to the species level. Each individual was only recorded once per pin. Relative cover of each species was calculated as the number of times a species was recorded divided by the number of points. Species richness was determined as the number of unique species identified from all pin points. Within a plot, cover classes were determined according to the Daubenmire subplot method (Daubenmire, 1959). A 1 x 1 m quadrat was placed at every 8 m along each transect, for a total of 10 Daubenmire subplots per treatment plot. Within each Daubenmire subplot, percent cover per square meter was visually estimated into one of seven categories: 0, <1, 1-5, 5-25, 26-50, 51-75, 76-100 %. Relative percent cover was averaged across subsplots using the midpoint value for each category. Cover estimates were then used to calculate evenness values for each treatment plot based on the Shannon Index (McCune and Grace, 2002). Species richness was determined as the number of unique species per 1 m^2 averaged across subplots for each treatment plot (Hurlbert, 1971).

In addition to the community survey, we examined treatment effects of two important invasive weed species that pose risks to the productivity and ecological status of rangelands in the western U.S. At valley grasslands, the grass species *Taeniatherum caput-medusae* (common name: medusahead) is an abundant, aggressive competitor and low-quality forage for cattle and other grazers (Young *et al.*, 1969; Young and Evans, 1971). *T. caput-medusae* is classified as high impact and a serious risk to native ecosystems by the California Invasive Plant Council and classified as a noxious weed (List C) by the California Department of Food and Agriculture. The forb, *Carthamus lanatus* (common name: woolly distaff thistle) is a common invader of coastal grasslands. Although it does not pose health problems for cattle, *C. lanatus* is an invasive, noxious annual forb that is commonly treated with herbicides to prevent competition with other forages and to eliminate physical obstructions that restrict cattle movement. *C. lanatus* is classified as moderate impact and a serious risk to native ecosystems by the California Invasive Plant Council and classified as a noxious weed (List B) by the California Department of Food and Agriculture. I monitored the changes to the relative abundance of *T. caput-medusae* and *C. lanatus* for four years following single the management events.

Statistical analysis

The R package, vegan, was used to calculate species richness, Shannon Weiner diversity index, Shannon's evenness, and relative abundance. I used analysis of variance (ANOVA) to assess the effects of compost amendment and keyline plowing on these diversity indices as well as aboveground biomass, forage consumption by cows, and aboveground plant N content and concentration. Treatment was used as a fixed-effect and block identifiers were included as a blocking factor. Site and years were analyzed separately as grazing and plant biomass and quality analyses were performed at different stages of plant growth. Biomass and plant N variables were log-transformed to achieve normality and homoscedasticity. I tested the effect of treatment, time, and the interaction between treatment and time on these variables using repeated measures MANOVA.

To evaluate treatment effects on the relative abundance of individual plant species, we calculated the log response ratio (Hedges *et al.*, 1999; Stein *et al.*, 2010) for each plant species as

$$\ln RR(i) = \ln \frac{C_{treatment}(i)}{C_{control}(i)}$$

where RR(*i*) is the response ratio of cover of each species (*i*). The terms $C_{\text{treatment}}$ and C_{control} are the average species' cover in the treatment and control plots, respectively. Log response ratios were calculated separately for each block and reported as block means ± 1 standard error. Sites were analyzed separately. Only species that occurred in ≥ 8 % of plots within each site were included in this analysis (e.g. 13 species at valley grasslands and 19 species at coastal grasslands). Greater cover of a species in treatment relative to control plots is represented by positive lnRR values, whereas negative lnRR values indicate less cover in treatment relative to control plots. For this analysis, we present data only from cover class vegetation surveys.

I compared point intercept and cover class vegetation survey methods by comparing species richness and relative abundance results. The relative abundances of each species calculated with data from either survey method were compared using ANOVAs. To compare diversity indicies, species richness was expressed as the percent difference between treatments and control at each block for each of the survey methods, such that:

$$\Delta SpeciesRichness(SP) = \frac{SR_{treatment} - SR_{control}}{SR_{control}} \times 100$$

Analyses were performed using either 7.0.2 JMP software (SAS Institute Inc. 2007) or R version 2.15.1 (The R Foundation for Statistical Computing). Statistical significance was determined as $p < 0.05$ unless otherwise noted. Data are reported as mean ± 1 standard error and plotted separately for each site.

Results

Effects of rangeland management on aboveground biomass and quality

Significant treatment differences in pregrazed aboveground biomass were detected at both grassland sites within the first year following the application of management treatments (Figure 2a). Plots that received compost amendments had 38 ± 21 and 51 ± 41 % (block mean ± 1 standard error) more aboveground biomass prior to grazing than control plots at the valley and coastal grasslands, respectively ($p < 0.05$ at each site). Aboveground biomass in the plowed plots was 30 ± 4 and 13 ± 25 % less than control plots at the valley and coastal grasslands, respectively ($p < 0.05$). Plowing did not offset gains in biomass production in the combined amended + plowed plots. The positive response of biomass production in plots receiving compost amendments persisted throughout the four-year study. The interaction between time and treatment was not significant although rainfall conditions ranged widely throughout the years of the experiment. At the time of grazing, composted plots contained an average of 100 ± 16 and 128 ± 6 g m^{-2} y^{-1} more aboveground biomass in the valley and coastal grasslands, respectively.

Aboveground biomass measured immediately after grazing events provided an indication of grazing intensity. For most years, the amount of aboveground biomass left standing immediately after grazing events did not differ among treatments (Figure 2b). At the coastal grassland, consumption of forage by cows was considerably variable in space, showing no clear preference for forage from treatment plots (Figure 2c). At the valley grassland, significantly more forage was consumed from compost and compost + plowed plots relative to control plots ($p < 0.0001$ for both treatments and all years). There was no significant difference between the amount of forage consumed from plowed and control plots. Cows consumed an average of 136 ± 6 and 98 ± 18 g m^{-2} y^{-1} more forage from compost and compost + plowed plots, respectively, relative to control plots.

Compost amendments significantly affected plant N content. For compost treatments, plant N content of pregrazed biomass was significantly greater at both sites for all four years of the study (Figure 3). At the valley grassland, aboveground biomass in compost amended plots increased by 8.0 ± 1.2 g N m^{-2} over four years, equivalent to an increase of 89 ± 6 % each year. At the coastal grassland, aboveground plant N content increased by 5.7 ± 1.1 g N m^{-2} over four years, or by 44 ± 10 % each year. The effect of compost amendments on aboveground plant N concentration was more variable. At the valley grassland site, vegetation from composted plots had higher N concentration than control plots during year 1 ($p < 0.0001$) and 4 ($p < 0.10$). At the coastal grassland site, plant N concentrations were greater in composted relative to control plots in year 2 ($p < 0.05$) and slightly lower in year 3 ($p < 0.10$). The interaction between treatment and time was significant at $p < 0.10$ for N concentrations at both grassland types. However, the response of plant N content did not depend on time and showed consistent increases across all four years of the study

Effects of rangeland management on plant diversity and species composition

Species richness at the coastal grasslands was greater than that of valley grasslands by 1.82 ± 0.28 species m^{-2}, averaged across all treatments and years ($p < 0.05$). Overall, species richness and evenness of vegetation communities at both grassland sites were relatively resistant to management-induced changes (Figure 4). Management treatments did not significantly alter species richness or evenness at the valley grassland. At the coastal grassland, species richness during year 1 and evenness during years 1 and 2 were significantly greater in the composted and compost + plowed plots than either plowed or control plots ($p < 0.05$). There was a significant effect of time and a time by treatment interaction for evenness ($p < 0.0001$), and a significant effect of time for species richness ($p < 0.0001$) at both sites.

Both grassland types were dominated by non-native annual grasses and forbs, which is typical of most rangelands in California (Biswell, 1956). The relative abundance of two species of concern appeared to respond to management treatments in the short term (within the first year of management application), but the effect diminished by the following growing season (Figure 5a). During the first year, the relative abundance of *T. caput-medusae* at the valley grassland was significantly greater in plowed plots (46 %; $p < 0.05$) and lower in compost + plowed plots (15 %; $p < 0.01$) compared to control (31 %) and composted plots (35 %). Significant reductions in *T. caput-medusae* were also detected in composted and compost + plowed plots during year 3. There was significant interaction between time and treatment. At the coastal grassland, the relative abundance of *C. lanatus* was significantly greater in the compost + plowed (30 %; $p < 0.0001$) and plowed (9 %; $p < 0.01$) plots relative to control (0.6 %) and compost (3 %) plots in year 1 (Figure 5b). The abundance of *C. lanatus* was < 4 % across all treatment over the next three years. Analysis of log response ratios (lnRR) of *T. caput-medusae* and *C. lanatus* show similar treatment effects.

I analyzed lnRR of the 13 and 19 most common species at the valley and coastal grasslands, respectively (Figure 6 and 7). There were no major shifts in the plant communities regarding taxonomic placement or life history strategy. Increases, decreases, and no changes to individual species abundance were observed at both sites. At the valley grassland, compost amendments significantly increased three annual grass species (*F. perennis*, *B. diandrus*, and *H. murinum*) and decreased the abundance of three common forbs and bulbs (*E. botrys*, *S. bipinnatifida*, and *D. capitatum*). Plowing generally had similar effects on species abundances with increases observed with two annual grasses (*T. caput-medusae* and *F. perennis*) and decreases observed with three forbs and bulbs (*S. bipinnatifida*, *D. capitatum*, and *T. hirtum*). At the coastal grassland compost amendments significantly increased four grasses (*F. perennis, V. bromoides, B. diandrus,* and *B. carinatus*), one forb (*G. dissectum*), and one legume (*M. polymorpha*) species and decreased one grass (*A. barbata*) and one forb (*H. glabra*) species. The plow treatment significantly increased three of the same grasses and decreased two grasses (*A. barbata* and *B. hordeaceus*) and two legume (*V. sativa* and *M. polymorpha*) species. The combined treatment generally showed similar responses as compost treatment. At both grassland types, the largest treatment responses were observed from species of low to mid-abundance.

Comparison of vegetation survey methods

I assessed the effectiveness of two common vegetation survey methods on detecting treatment impacts to species abundances. Generally, the point intercept and cover class methods found similar treatment effects to the relative abundances of individual species. For all treatments, the relative abundances of two species, *Trifolium spp. and F. perennis,* were lower using the point intercept method compared to the cover class method. Abundance of some less commonly occurring species, particularly forbs, was underestimated using the point intercept. Differences were not consistent across treatments, and were restricted almost entirely to forbs, legumes, and bulb species. Method differences included underestimated abundances by the point intercept method of *S. binnatifida*, *E. botrys, and D. capitatum* (control), *T.laxa* and *E. botrys* (plowed plots), *H. murinum* and *B. madritensis* (composted plots), and *C. lanatus* and *R. crispus* (compost + plowed plots). The only two grasses which differed between methods, *H. murinum* and *B. madritensis*, occurred on just 40 and 14 % of the Daubenmire subplots at the valley grassland. These annual grass species tend to be shorter and less common than more dominant species.

I detected significant differences between methods when assessing treatment differences in species richness. Averaged across sites and years, no treatment differences in species richness were detected using the point-intercept method. In contrast, positive treatment responses were detected from plowed and compost + plowed plots using the Daubenmire cover class method (Figure 8).

Discussion

Effects of rangeland management on aboveground biomass and forage quality

Keyline plow techniques, which aim to increase water infiltration and distribution across the landscape, are increasingly used in permaculture, agroecology, and holistic agriculture to increase production, particularly in dryland ecosystems (Yeomans, 1954; Mollison, 1988). I know of no studies that have investigated the effects of keyline plowing in Mediterranean grasslands. Contrary to expectations, keyline plowing did not increase aboveground biomass. In our study, a single application of the keyline plow technique reduced aboveground biomassfor grazing events by an average of 13 % at the coastal grassland and 29 % at the valley grassland during the first year of study. It is notable that this year was considerably drier compared to historical average rainfall. Aboveground biomass did not increase significantly relative to controls in subsequent years.

The application of composted organic material is commonly used in agricultural lands to boost crop production (Cassman *et al.*, 2002; Blair *et al.*, 2006) and has been also proposed as a means of increasing plant production and sequestering C in rangelands (Cabrera *et al.*, 2009; Ippolito *et al.*, 2010; Powlson *et al.*, 2012; Ryals and Silver, in press). Aboveground biomass just before the first grazing event (3-5 months after compost applications) was 69 % (coastal) and 38 % (valley) greater in plots receiving compost amendments, separately or in combination with the keyline plow. This was not surprising, as most rangelands are N-limited or co-limited by N and water (Harpole *et al.*, 2007). Compost is a rich source of N and generally increases soil water holding capacity (Gagnon *et al.*, 1998). What was surprising was that the increase in biomass production was sustained throughout four years with no apparent signs of diminishing, despite a wide a range of annual rainfall amounts (380 to 843 mm/y at valley grasslands and 771 to 1163 mm/y at coastal grasslands). This result indicates that treatment effects did not depend strongly on rainfall or other environmental conditions of a given year. This carry-over effect can likely be attributed to the nature of compost acting as a slow release fertilizer and elevated available plant nutrients and soil water holding capacity for several years following application (Gerzabek *et al.*, 1997; Blair *et al.*, 2006; Ryals and Silver, in press).

Nitrogen content of plants increased with a result of a single compost application, and the effect persisted for all four years of the study. Plant N content increased by 44 % and 89 % at coastal and valley grasslands, respectively. A similar result was reported following the application of composted biosolids to disturbed perennial grasslands (McFarland *et al.*, 2010). Aboveground plant N concentration responded positively in most years at the valley grassland and showed variable responses at the coastal grassland. The interaction between treatment and time for N concentrations at both grassland types suggests that treatment responses depend somewhat on conditions within a given year. Thus, the large positive response of plant N content availability is largely driven by large and sustained increases in aboveground biomass. These long lasting increases in plant production and N content have positive implications for improving rangeland sustainability by decreasing dependency on forage supplements and by buffering the impacts of high interannual variability in plant production. Increases in plant N content correspond directly to increases in crude protein, a common index used by specialists to assess nutritional quality of forage (Stokes and Prostko, 1998; McFarland *et al.*, 2010). During rotational grazing events at the valley grassland, cows consumed an average of 136 g m^{-2} y^{-1}, or 450 %, more forage from composted compared to control plots. However, the amount of residual biomass left after grazing was similar or slightly less in composted plots, indicating that grazing pressure was similar across treatments. Similar, albeit more variable, trends were observed at the coastal grassland where grazing events occurred toward the end of the growing season.

Effects of rangeland management on plant diversity and species composition

The Mediterranean and Eurasian annual grasses that dominate California's rangelands are aggressive competitors of native perennial grasses and thrive in the resource pulse-driven semiarid conditions (Norton *et al.*, 2007). Physical disturbance of soil structure and external nutrient inputs can be important drivers of management-induced changes to species diversity and community composition in most grassland ecosystems (Faust *et al.*, 2012). However, the impacts of management practices on vegetation communities in heavily invaded grasslands are less well known. State and transition model theory suggests that the restoration pathways are difficult to achieve in invaded grasslands in a post-threshold state (Briske *et al.*, 2008), particularly because annual exotic grasses perpetuate and enhance the resource-pulse driven environment in which they thrive (Kourtev *et al.*, 2002; Ehrenfeld, 2003).

I hypothesized that the grasslands in our study, which are dominated by exotic annual species, would be relatively resistant to shifts in community composition with one-time management events that physically disturb soil (keyline plow) and/or increase nutrient availability (compost amendments). For four years following management events, species richness varied significantly through time, but showed little affect of management treatments, supporting our hypotheses. Despite the lack of major shifts in species diversity, we found that several grass and forb species responded to management.

Extensive research has explored the role of reactive N inputs on species loss in terrestrial ecosystems worldwide (Foster and Gross, 1998; Samuel and Hart, 1998; Sala *et al.*, 2000; Stevens *et al.*, 2004; Clark and Tilman, 2008; Cole *et al.*, 2008; Bobbink *et al.*, 2010). Most of these studies explore natural N deposition gradients or experimental inorganic N additions, due to the rapid rise in atmospheric N deposition and its large impact on the N status of many ecosystems (Galloway *et al.*, 2004). Observed increases in biomass production with N additions are generally accompanied by declines in species richness (Tilman, 1993) and increases in fast-growing weedy invasive species (Davis *et al.*, 2000), although results of empirical studies have varied (Clark *et al.*, 2007). Compost amendments provide a slow release of N over several years (Ryals and Silver, in press), which has the potential to increase mineral N availability for invasive species. However, compost amendments also provide other nutrients, increase soil water holding capacity, and supply labile C. In restoration studies, labile C amendments (commonly sugar or sawdust) were applied to immobilize N thereby potentially increasing competitiveness of native grass species (DiTomaso, 2000; Corbin and D'Antonio, 2004a; Perry *et al.*, 2010; Mitchell and Bakker, 2011).

I found that the abundances of a few individual grass and forbs species responded either positively (e.g. *F. perennis* and *B. diandrus* at both sites) or negatively (e.g. *D. capitatum* at valley grasslands and *H. glabra* at coastal grasslands) to compost amendments, but did not lead to major shifts in plant functional types at either grassland site. In plowed plots, individual species' responses to a one-time keyline plow event showed a similar trends in species responses at both sites, with the exception of two legumes that were reduced in plowed plots and increased in composted and composted + plowed plots at the coastal grassland. It is difficult to determine if individual species' responses are truly responses to management or a function of spatial heterogeneity because pretreatment surveys were not conducted,. My observations that species responses were similar through time and that responses to plow events and compost amendments were similar in magnitude and direction for most species suggests that preexisting differences may be responsible for observed changes, rather than management. These results are contrary to studies of composted biosolid application to rangelands, which demonstrated increases in perennial grass cover (Sullivan *et al.*, 2006; Ippolito *et al.*, 2010), but in agreement with findings that composted biosolid application did not change vegetation communities in a rangeland dominated by invasive species (McFarland *et al.*, 2010).

Aboveground biomass in composted and composted + plowed plots increased without reducing species diversity over four years. However, management had an apparent short-lived effect on the abundance of two secondary invaders that threaten the resiliency of these grassland communities (Dahl and Tisdale, 1975; DiTomaso, 2000). At the coastal grassland, the abundance of a noxious annual forb, *C. lanatus*, was one to two orders of magnitude greater than controls in plots that receive a one-time keyline plow event either with or without the combination of compost amendment. It is not clear why a plow event would favor thistle in the short-term. No treatment differences in *C. lanatus* abundance were detected for the following three years, but long-term effects of a productive season are unknown. *C. lanatus* has large seedbanks that can persist in the soil for up to 10 years (Sindel, 1997; Grace *et al.*, 2002a; Grace *et al.*, 2002b) and some studies report a lag in shifts in grassland communities up to 9 years (Faust *et al.*, 2012). At the valley grassland, the abundance of a prevalent noxious annual grass, *T. caput-medusae* significantly decreased in compost + plowed plots relative controls during the first year. There is evidence that *T. caput-medusae* invasion increases with N enrichment (Dahl and Tisdale, 1975; Blank and Sforza, 2007) and reduces with labile C additions (Young *et al.*, 1998). Seeds of *T. caput-medusae* are highly adapted for germination in the litter layer (Young *et al.*, 1971). Thus, microbial immobilization of excess N from labile compost C substrate in combination with a disturbance of the litter layer may explain the short term reduction in *T. caput-medusae* in the compost + plowed plots. However, the reduction of *T. caput-medusae* due to the one-time management event did not persist to the end of this study.

Comparison of vegetation survey methods

Accurately determining changes to vegetation communities may be influenced by type of sampling method used (Rich and Smith, 1996; Kercher *et al.*, 2003). I assessed species richness and relative abundance based on data collected using the point intercept and cover class methods. Averaged over both sites and four years, the relative response of management treatments to species richness was positive using the cover class method. In contrast, no change is detected using the point intercept method. The likely explanation of this discrepancy is that the line intercept is less likely to capture rare or less common plants (Goodall, 1952) and has been shown to produce lower species counts compared to cover class methods (Kercher *et al.*, 2003). Overall, there were general agreements in the relative abundance of most individual species. However, compared to the point intercept method, the cover class method resulted in greater estimates of diminutive species (e.g. *Trifolium spp.*), rigid grasses with small awns (e.g. *F. perennis*), and less frequently occurring forbs and bulbs. Therefore, care should be taken when assessing management impacts on vegetation communities based on sampling method, especially when considering impacts to new invaders or other species of concern.

Conclusions

Monitoring changes in aboveground biomass, forage nutritional quality, and vegetation community structure associated with management is necessary to determine the magnitude of these key tradeoffs or co-benefits of rangeland C sequestration strategies. I found that keyline plowing reduced aboveground biomass in the short term (< 1 y) but did not significantly affect aboveground biomass or nutritional quality over four years. In contrast, one-time compost amendments greatly increased aboveground biomass and improved forage quality at both grassland types. During the first year, management treatments increased the abundance of a noxious thistle at the coastal grassland and decreased the abundance of a noxious annual grass at the valley grassland. Responses did not last beyond the first year. The lack of major shifts in vegetation or prolonged responses of noxious species suggests that there are few trade-offs between plant production and species diversity in these grasslands.

References

Bartolome, J.W., Klukkert, S.E., Barry, W.J., 1986. Opal phytoliths as evidence for displacement of native Californian grassland. Madrono 33, 217-222.

Betts, K., 2003. Demonstrating carbon sequestration. Environmental Science & Technology 37, 354A-355A.

Biswell, H.H., 1956. Ecology of California grasslands. Jour Range Management 9, 19-24.

Blair, N., Faulkner, R.D., Till, A.R., Poulton, P.R., 2006. Long-term management impacts on soil C, N and physical fertility - Part 1: Broadbalk experiment. Soil & Tillage Research 91, 30-38.

Blank, R.R., Sforza, R., 2007. Plant-soil relationships of the invasive annual grass *Taeniatherum caput-medusae*: a reciprocal transplant experiment. Plant and Soil 298, 7-19.

Blumenthal, D., 2005. Ecology - Interrelated causes of plant invasion. Science 310, 243-244.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L., De Vries, W., 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecological Applications 20, 30-59.

Briske, D.D., Bestelmeyer, B.T., Stringham, T.K., Shaver, P.L., 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology & Management 61, 359-367.

Brown, S., Dushku, A., Pearson, T., Shoch, D., Winsten, J., Sweet, S., Kadyszewski, J., 2004. Carbon supply from changes in management of forest, range, and agricultural lands of California. Winrock International, for the California Energy Commission, PIER Energy-Related Environmental Research, Sacramento, CA, USA, p. 114.

Cabrera, V.E., Stavast, L.J., Baker, T.T., Wood, M.K., Cram, D.S., Flynn, R.P., Ulery, A.L., 2009. Soil and runoff response to dairy manure application on New Mexico rangeland. Agriculture Ecosystems & Environment 131, 255-262.

Cassman, K.G., Dobermann, A., Walters, D.T., 2002. Agroecosystems, nitrogen-use efficiency, and nitrogen management. Ambio 31, 132-140.

Chan, K.M.A., Hoshizaki, L., Klinkenberg, B., 2011. Ecosystem Services in Conservation Planning: Targeted Benefits vs. Co-Benefits or Costs? Plos One 6.

Clark, C.M., Cleland, E.E., Collins, S.L., Fargione, J.E., Gough, L., Gross, K.L., Pennings, S.C., Suding, K.N., Grace, J.B., 2007. Environmental and plant community determinants of species loss following nitrogen enrichment. Ecology Letters 10, 596-607.

Clark, C.M., Tilman, D., 2008. Loss of plant species after chronic low-level nitrogen deposition to prairie grasslands. Nature 451, 712-715.

Cole, L., Buckland, S.M., Bardgett, R.D., 2008. Influence of disturbance and nitrogen addition on plant and soil animal diversity in grassland. Soil Biology & Biochemistry 40, 505-514.

Conant, R.T., Ogle, S.M., Paul, E.A., Paustian, K., 2011. Measuring and monitoring soil organic carbon stocks in agricultural lands for climate mitigation. Frontiers in Ecology and the Environment 9, 169-173.

Conant, R.T., Paustian, K., Elliott, E.T., 2001. Grassland management and conversion into grassland: Effects on soil carbon. Ecological Applications 11, 343-355.

Corbin, J.D., D'Antonio, C.M., 2004a. Can carbon addition increase competitiveness of native grasses? A case study from California. Restoration Ecology 12, 36-43.

Corbin, J.D., D'Antonio, C.M., 2004b. Competition between native perennial and exotic annual grasses: Implications for an historical invasion. Ecology 85, 1273-1283.

D'Antonio, C.M., 2007. Ecology of invasive non-native species in California grasslands. In: Stromberg, M.R., Corbin, J.D., D'Antonio, C.M. (Eds.), California Grasslands: Ecology and Management. UC Press, Berkeley, CA.

D'Hose, T., Cougnon, M., De Vliegher, A., Willekens, K., Van Bockstaele, E., Reheul, D., 2012. Farm Compost Application: Effects on Crop Performance. Compost Science & Utilization 20, 49-56.

Dahl, B.E., Tisdale, E.W., 1975. Environmental factors related to medusahead distribution. Journal of Range Management 28, 463-468.

Daubenmire, R., 1959. A canopy coverage method of vegetation analysis. Northwest Science 33, 43-64.

Davis, M.A., Grime, J.P., Thompson, K., 2000. Fluctuating resources in plant communities: a general theory of invasibility. Journal of Ecology 88, 528-534.

De Schrijver, A., De Frenne, P., Ampoorter, E., Van Nevel, L., Demey, A., Wuyts, K., Verheyen, K., 2011. Cumulative nitrogen input drives species loss in terrestrial ecosystems. Global Ecology and Biogeography 20, 803-816.

Dean, C., Wardell-Johnson, G.W., Harper, R.J., 2012. Carbon management of commercial rangelands in Australia: Major pools and fluxes. Agriculture Ecosystems & Environment 148, 44-64.

Derner, J.D., Schuman, G.E., 2007. Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. Journal of Soil and Water Conservation 62, 77-85.

DiTomaso, J.M., 2000. Invasive weeds in rangelands: Species, impacts, and management. Weed Science 48, 255-265.

Eghball, B., Power, J.F., 1999. Composted and noncomposted manure application to conventional and no-tillage systems: Corn yield and nitrogen uptake. Agronomy Journal 91, 819-825.

Ehrenfeld, J.G., 2003. Effects of exotic plant invasions on soil nutrient cycling processes. Ecosystems 6, 503-523.

Faust, C., Storm, C., Schwabe, A., 2012. Shifts in plant community structure of a threatened sandy grassland over a 9-yr period under experimentally induced nutrient regimes: is there a lag phase? Journal of Vegetation Science 23, 372-386.

Fisher, B., Bradbury, R.B., Andrews, J.E., Ausden, M., Bentham-Green, S., White, S.M., Gill, J.A., 2011. Impacts of species-led conservation on ecosystem services of wetlands: understanding co-benefits and tradeoffs. Biodiversity and Conservation 20, 2461-2481.

Foster, B.L., Gross, K.L., 1998. Species richness in a successional grassland: Effects of nitrogen enrichment and plant litter. Ecology 79, 2593-2602.

Gagnon, B., Simard, R.R., Goulet, M., Robitaille, R., Rioux, R., 1998. Soil nitrogen and moisture as influenced by composts and inorganic fertilizer rate. Canadian Journal of Soil Science 78, 207-215.

Galloway, J.N., Dentener, F.J., Capone, D.G., Boyer, E.W., Howarth, R.W., Seitzinger, S.P., Asner, G.P., Cleveland, C.C., Green, P.A., Holland, E.A., Karl, D.M., Michaels, A.F., Porter, J.H., Townsend, A.R., Vorosmarty, C.J., 2004. Nitrogen cycles: past, present, and future. Biogeochemistry 70, 153-226.

Gerzabek, M.H., Pichlmayer, F., Kirchmann, H., Haberhauer, G., 1997. The response of soil organic matter to manure amendments in a long-term experiment at Ultuna, Sweden. European Journal of Soil Science 48, 273-282.

Goodall, D.W., 1952. Some considerations in the use of point quandrats for the analysis of vegetation. Australian Journal of Scientific Research Series B-Biological Sciences 5, 1-41.

Grace, B.S., Sheppard, A.W., Whalley, R.D.B., Sindel, B.M., 2002a. Seedbanks and seedling emergence of saffron thistle (*Carthamus lanatus*) in eastern Australian pastures. Australian Journal of Agricultural Research 53, 1327-1334.

Grace, B.S., Whalley, R.D.B., Sheppard, A.W., Sindel, B.M., 2002b. Managing saffron thistle in pastures with strategic grazing. Rangeland Journal 24, 313-325.

Harpole, W.S., Potts, D.L., Suding, K.N., 2007. Ecosystem responses to water and nitrogen amendment in a California grassland. Global Change Biology 13, 2341-2348.

Hedges, L.V., Gurevitch, J., Curtis, P.S., 1999. The meta-analysis of response ratios in experimental ecology. Ecology 80, 1150-1156.

Herrero, M., Thornton, P.K., Gerber, P., Reid, R.S., 2009. Livestock, livelihoods and the environment: understanding the trade-offs. Current Opinion in Environmental Sustainability 1, 111-120.

Hurlbert, S.H., 1971. Nonconcept of species diversity – critique and alternative parameters. Ecology 52, 577.

Ippolito, J.A., Barbarick, K.A., Paschke, M.W., Brobst, R.B., 2010. Infrequent composted biosolids applications affect semi-arid grassland soils and vegetation. Journal of Environmental Management 91, 1123-1130.

Kercher, S.M., Frieswyk, C.B., Zedler, J.B., 2003. Effects of sampling teams and estimation methods on the assessment of plant cover. Journal of Vegetation Science 14, 899-906.

Kirby, K.J., Bines, T., Burn, A., Mackintosh, J., Pitkin, P., Smith, I., 1986. Seasonal and observer differences in vascular plant records from British woodlands. Journal of Ecology 74, 123-131.

Kourtev, P.S., Ehrenfeld, J.G., Haggblom, M., 2002. Exotic plant species alter the microbial community structure and function in the soil. Ecology 83, 3152-3166.

Kowaljow, E., Julia Mazzarino, M., Satti, P., Jimenez-Rodriguez, C., 2010. Organic and inorganic fertilizer effects on a degraded Patagonian rangeland. Plant and Soil 332, 135-145.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma 123, 1-22.

Lemaire, G., 2012. Intensification of animal production from grassland and ecosystem services: a trade-off. CAB Reviews 7, 1-7.

Leps, J., Hadincova, V., 1992. How reliable are our vegetation analyses? Journal of Vegetation Science 3, 119-124.

Lipper, L., Dutilly-Diane, C., McCarthy, N., 2010. Supplying carbon sequestration from west African rangelands: Opportunities and barriers. Rangeland Ecology & Management 63, 155-166.

Masters, R.A., Sheley, R.L., 2001. Principles and practices for managing rangeland invasive plants. Journal of Range Management 54, 502-517.

McCune, B., Grace, J., 2002. Analysis of Ecological Communities. MJM Software Design, Gleneden Beach, Oregon.

McFarland, M.J., Vasquez, I.R., Vutran, M., Schmitz, M., Brobst, R.B., 2010. Use of biosolids to enhance rangeland forage quality. Water Environment Research 82, 455-461.

Mitchell, J.E., Bartling, P.N.S., Obrien, R., 1988. Comparing cover-class macroplot data with direct estimates from small plots. American Midland Naturalist 120, 70-78.

Mitchell, R.M., Bakker, J.D., 2011. Carbon addition as a technique for controlling exotic species in pacific northwest prairies. Northwest Science 85, 247-254.

Mollison, B., 1988. Permaculture: A Designer's Manual. Tagari Books, Tyalgun, TAS. Morgan, J.A., Follett, R.F., Allen, L.H., Jr., Del Grosso, S., Derner, J.D., Dijkstra, F., Franzluebbers, A., Fry, R., Paustian, K., Schoeneberger, M.M., 2010. Carbon sequestration in agricultural lands of the United States. Journal of Soil and Water Conservation 65, 6A-13A.

Norton, J.B., Monaco, T.A., Norton, U., 2007. Mediterranean annual grasses in western North America: kids in a candy store. Plant and Soil 298, 1-5.

Olander, L.P., Cooley, D.M., Galik, C.S., 2012. The potential role for management of U.S. public lands in greenhouse gas mitigation and climate policy. Environmental Management 49, 523-533.

Perry, L.G., Blumenthal, D.M., Monaco, T.A., Paschke, M.W., Redente, E.F., 2010. Immobilizing nitrogen to control plant invasion. Oecologia 163, 13-24.

Powlson, D.S., Bhogal, A., Chambers, B.J., Coleman, K., Macdonald, A.J., Goulding, K.W.T., Whitmore, A.P., 2012. The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study. Agriculture Ecosystems & Environment 146, 23-33.

Rich, T.C.G., Smith, P.A., 1996. Botanical recording, distribution maps and species frequency. Watsonia 21, 155-167.

Ryals, R., Silver, W.L., in press. Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grassland ecosystems. Ecological Applications.

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Biodiversity - Global biodiversity scenarios for the year 2100. Science 287, 1770-1774.

Sala, O.E., Paruelo, J.M., 1997. Ecosystem services in grasslands. Pages 237-253 in G.C. Daily, editor. Nature's services: Societal dependence on natural ecosystems. Island Press, Washington, D.C.

Samuel, M.J., Hart, R.H., 1998. Nitrogen fertilization, botanical composition and biomass production on mixed-grass rangeland. Journal of Range Management 51, 408-416.

Sindel, B.M., 1997. The persistence and management of thistles in Australian pastures. Ecology and Management of Weeds Proc. 50th N.Z. Plant Protection Conf., 453-456.

Smith, P., 2004. Soils as carbon sinks: the global context. Soil Use and Management 20, 212-218.

Srivastava, P., Kumar, A., Behera, S.K., Sharma, Y.K., Singh, N., 2012. Soil carbon sequestration: an innovative strategy for reducing atmospheric carbon dioxide concentration. Biodiversity and Conservation 21, 1343-1358.

Stein, C., Unsicker, S.B., Kahmen, A., Wagner, M., Audorff, V., Auge, H., Prati, D., Weisser, W.W., 2010. Impact of invertebrate herbivory in grasslands depends on plant species diversity. Ecology 91, 1639-1650.

Stevens, C.J., Dise, N.B., Mountford, J.O., Gowing, D.J., 2004. Impact of nitrogen deposition on the species richness of grasslands. Science 303, 1876-1879.

Stokes, S.R., Prostko, E.P., 1998. Understanding forage quality analysis. Texas A & M University AgriLife Extension Service.

Sullivan, T.S., Stromberger, M.E., Paschke, M.W., Ippolito, J.A., 2006. Long-term impacts of infrequent biosolids applications on chemical and microbial properties of a semi-arid rangeland soil. Biology and Fertility of Soils 42, 258-266.

Sykes, J.M., Horrill, A.D., Mountford, M.D., 1983. Use of visual cover assessments as quantitative estimators of some Brisish woodland taxa. Journal of Ecology 71, 437-450.

Tilman, D., 1993. Species richness of experimental productivity gradients – how important is colonization limitation. Ecology 74, 2179-2191.

Yeomans, P.A., 1954. The Keyline Plan. Waite & Bull, Sydney, Australia.

Young, J.A., Evans, R.A., 1971. Medusahead invasion as influenced by herbicides and grazing on low sagebrush sites. Journal of Range Management 24, 451-&.

Young, J.A., Evans, R.A., Eckert, R.E., 1969. Emergence of medusahead and other grasses from 4 seeding depths. Weed Science 17, 376-&.

Young, J.A., Evans, R.A., Kay, B.L., 1971. Germination of caryopses of annual grasses in simulated litter. Agronomy Journal 63, 551-&.

Young, J.A., Trent, J.D., Blank, R.R., Palmquist, D.E., 1998. Nitrogen interactions with medusahead (*Taeniatherum caput-medusae ssp. asperum*) seedbanks. Weed Science 46, 191-195.

Figures

Figure 1. Vegetation community sampling design, illustrated at one example block. Each block consisted of four randomly assigned treatment plots (control, amended, plowed, amended + plowed). Two transect line per plot were used for the point intercept method, in which vegetation was recorded every 0.5 m along transects. Ten 1 m x 1m Daubenmire plots were used for cover class assessment. Nine biomass samples (200 cm^2 each) were taken in each plot before and after grazing.

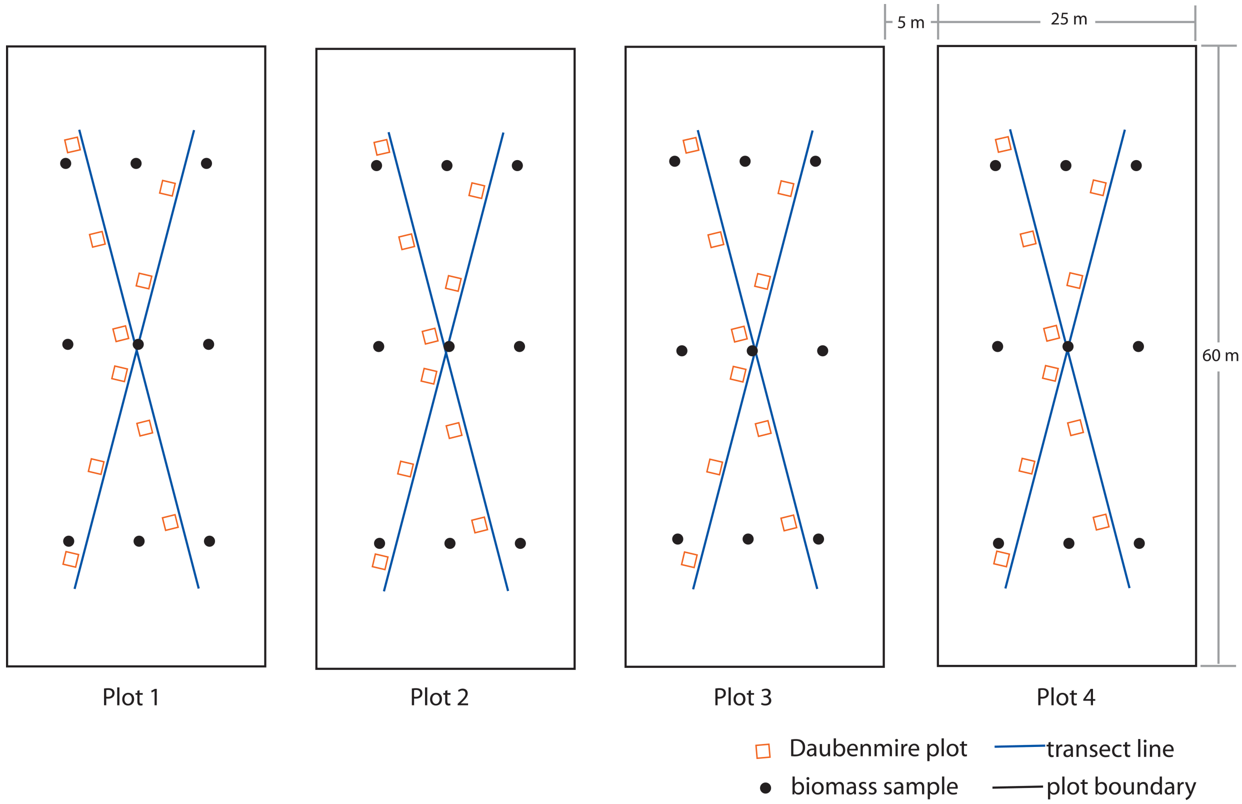
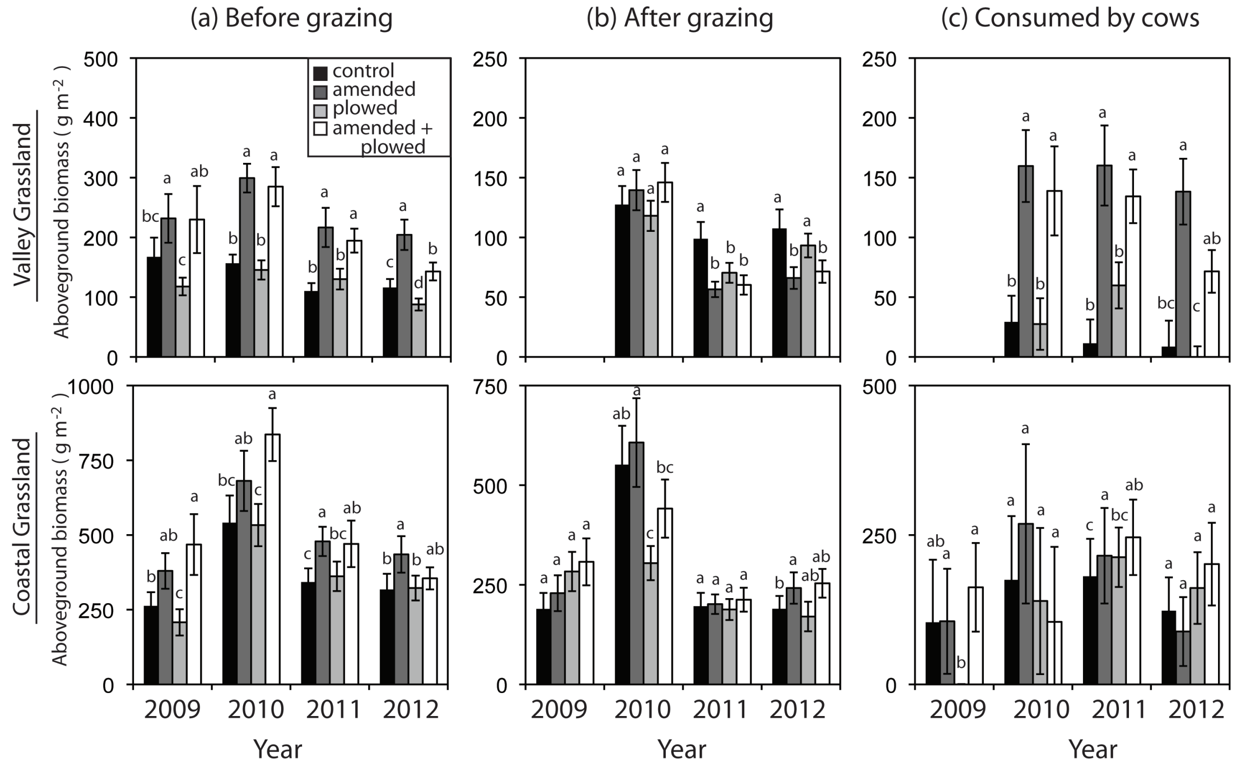


Figure 2. Dry weight of standing biomass (a) prior to grazing event and (b) < 24 hours after grazing event from four rangeland management practices. Biomass was clipped no more than 24 hours before or after the grazing event, which lasted 4 to 15 days, depending on standing biomass. (c) The amount of forage consumed by cows was calculated as the difference between (a) and (b). Data are shown for four years following treatment at a valley grassland (top panel) and coastal grassland (bottom panel). Bars are block means ± 1 standard error. Letters indicate significant differences at $p < 0.05$.



(a) Before grazing

(b) After grazing

(c) Consumed by cows

Figure 3. The top panel shows N content of aboveground biomass immediately prior to grazing events from control and amended plots at (a) valley and (b) coastal grasslands. Plant N content is a function of biomass (see Figure 2) and N concentration. The bottom panel shows the treatment effect on aboveground plant N concentration as the difference between control and amended plots. Error bars are ± 1 standard error. Asterisks indicate significance. * $p < 0.10$; ** $p < 0.05$; *** $p < 0.0001$.

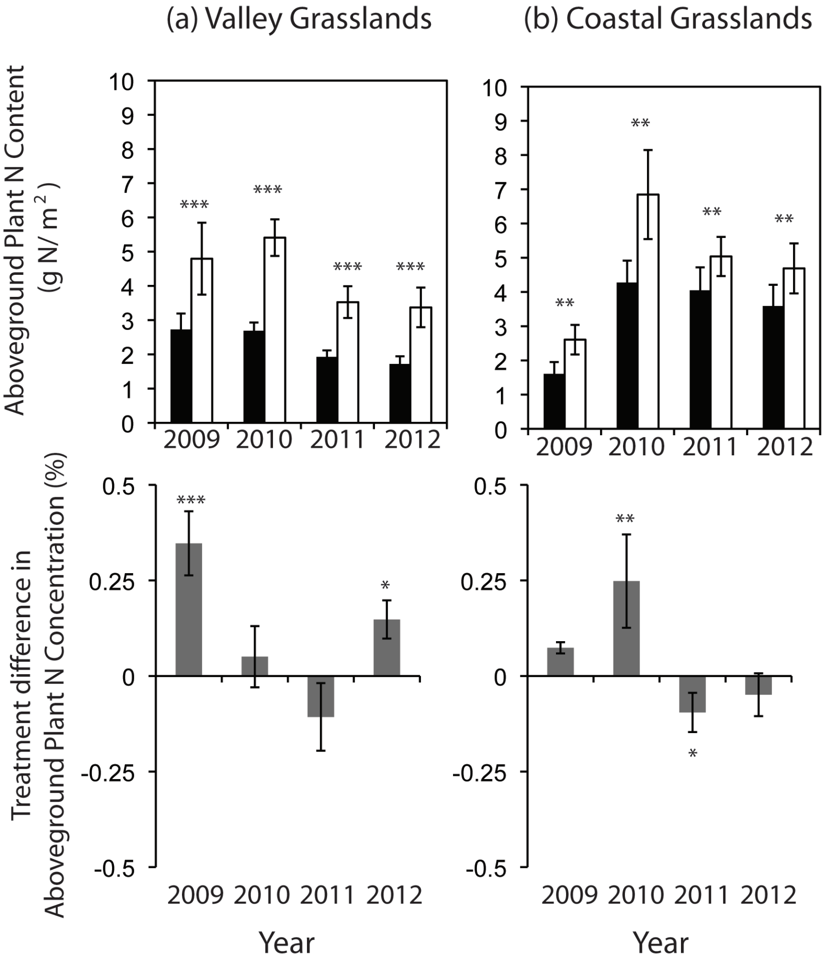


Figure 4. Plant species richness (top panels) and Shannon evenness index (bottom panels) in four treatment plots at (a) valley and (b) coastal grasslands. Symbols are block means ± 1 standard error.

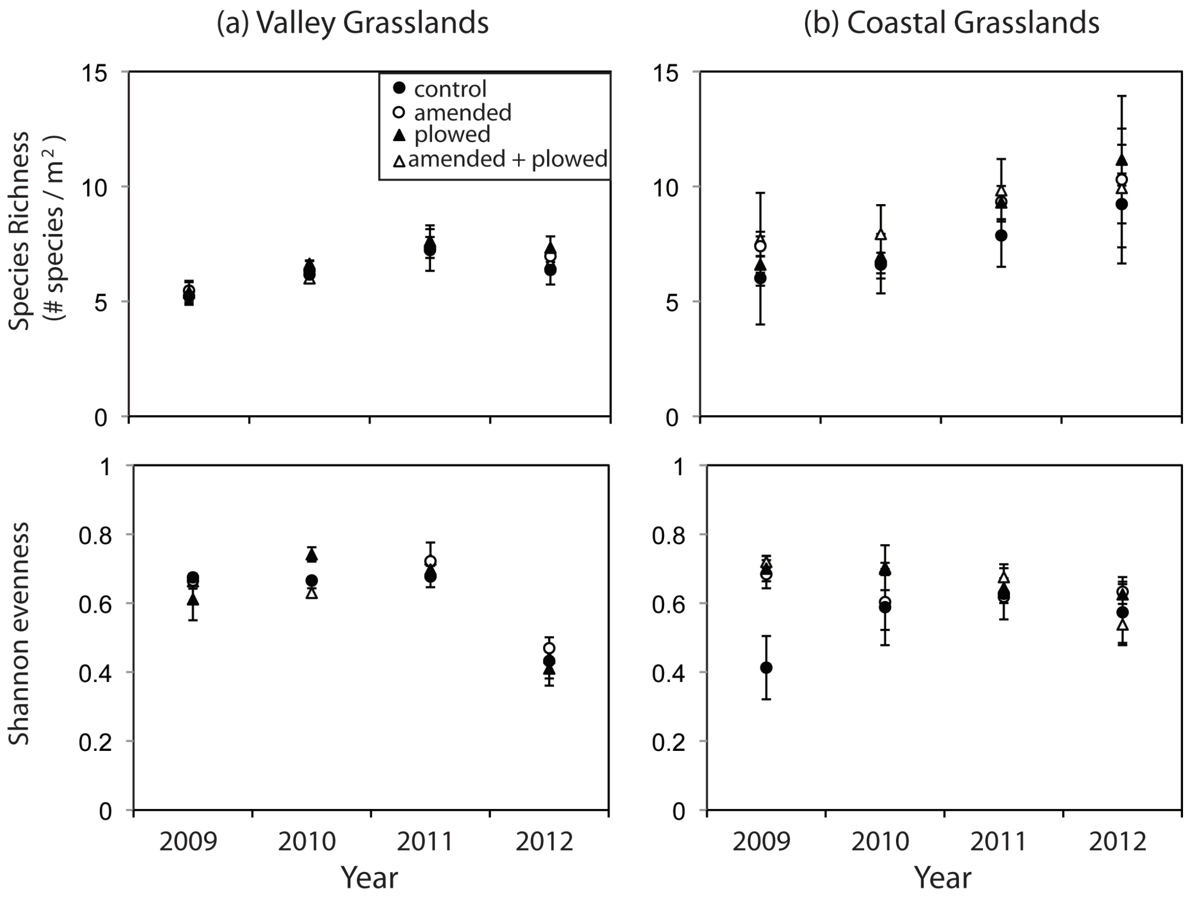


Figure 5. Relative abundance of *Taeniatherum caput-medusae* (common name: medusahead) at valley grasslands and *Carthamus lanatus* (common name: woolly distaff thistle) at coastal grasslands for four years following single management events. Symbols are treatment means and error bars are ± 1 standard error.

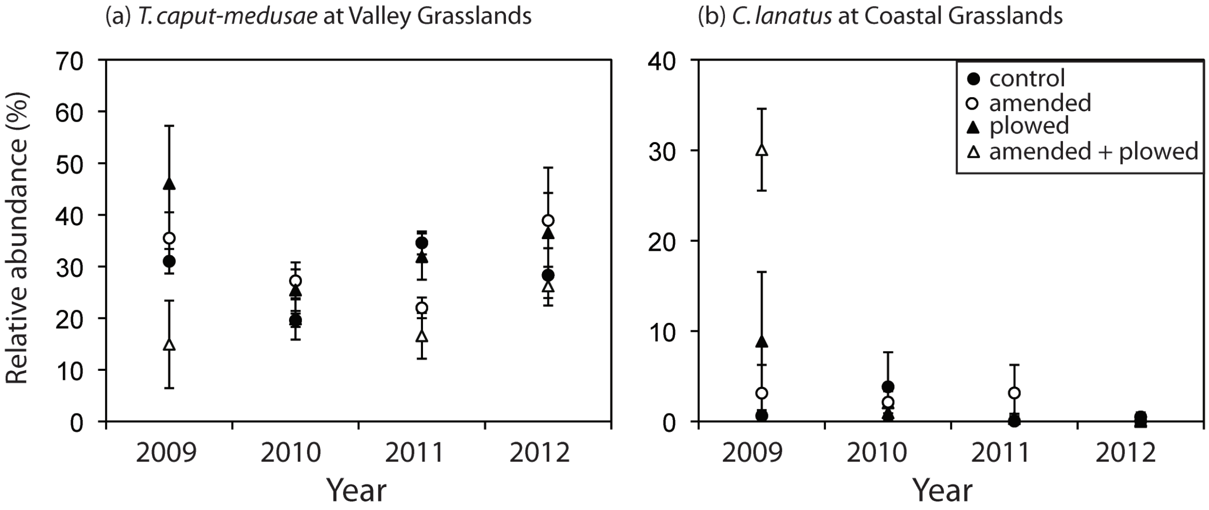


Figure 6. The influence of single management events on the relative percent cover of plant species with > 8 % occurrence (O) across all plots at the valley grasslands. Species are grouped by functional type (grasses, forbs, legumes, and forbs). Lifecycle (L) indicates annual (*a*) or perennial (*p*) species. Status (*S*) indicates ecological impact as designates by CalFlora, as either native (*n*), invasive (*i*), or exotic (*e*). Bars are paired plot treatment means of the log response averaged over four years. Positive values indicate that the relative cover of a plant species increased due to treatment. Negative values indicated a reduction of species cover. Error bars are ± 1 standard error. Asterisks indicate significance, where * $p < 0.05$, ** $p < 0.01$, and *** $p < 0.0001$.

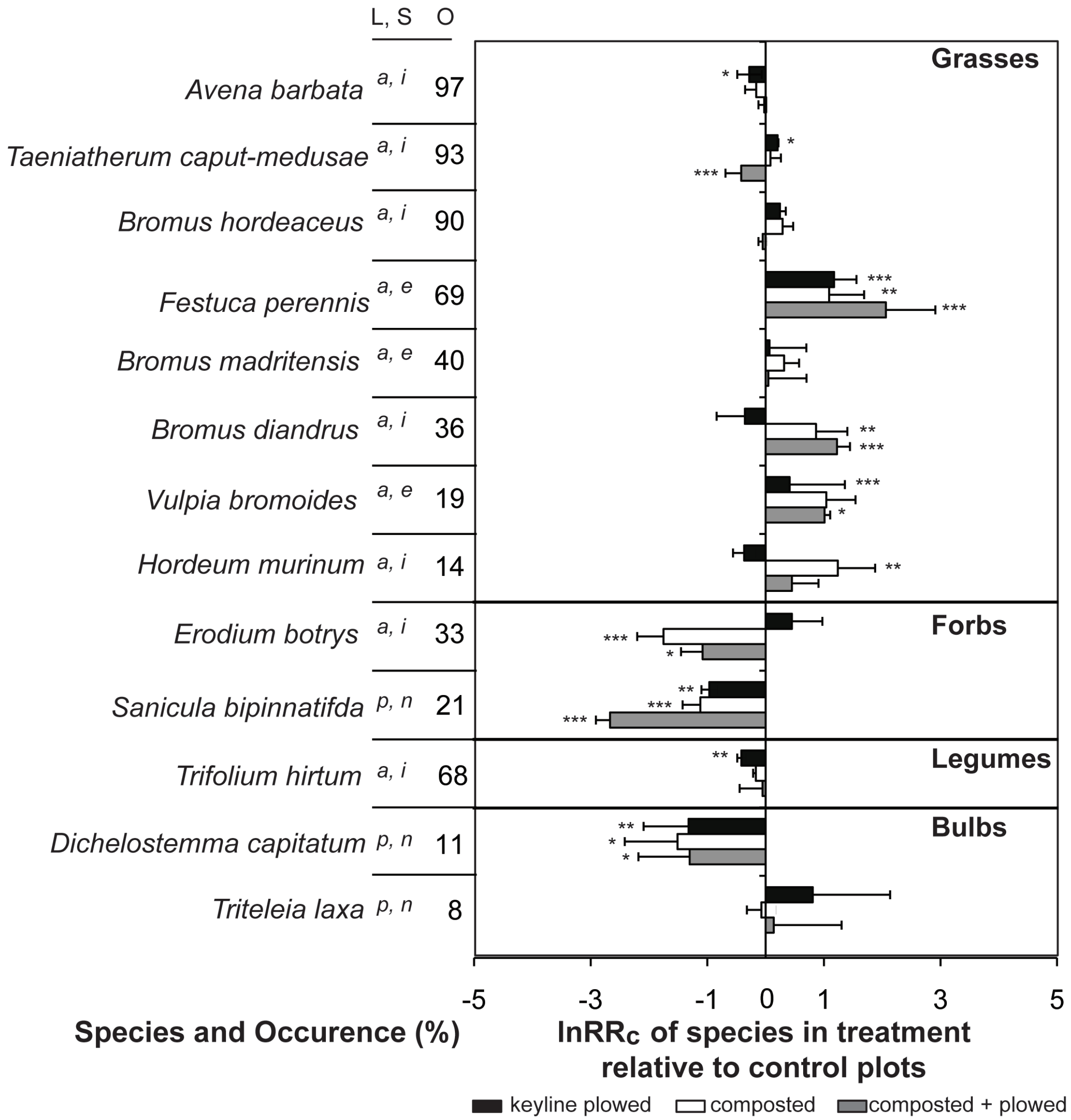


Figure 7. The influence of single management events on the relative percent cover of plant species with > 10 % occurrence (O) across all plots at the coastal grasslands. Species are grouped by functional type (grasses, forbs, and legumes). Lifecycle (L) indicates annual (*a*) or perennial (*p*) species. Status (*S*) indicates ecological impact as designates by CalFlora, as either native (*n*), invasive (*i*), or exotic (*e*). Bars are paired plot treatment means of the log response averaged over four years. Positive values indicate that the relative cover of a plant species increased due to treatment. Negative values indicated a reduction of species cover. Error bars are ± 1 standard error. Asterisks indicate significance, where * $p < 0.05$, ** $p < 0.01$, and *** $p < 0.0001$.

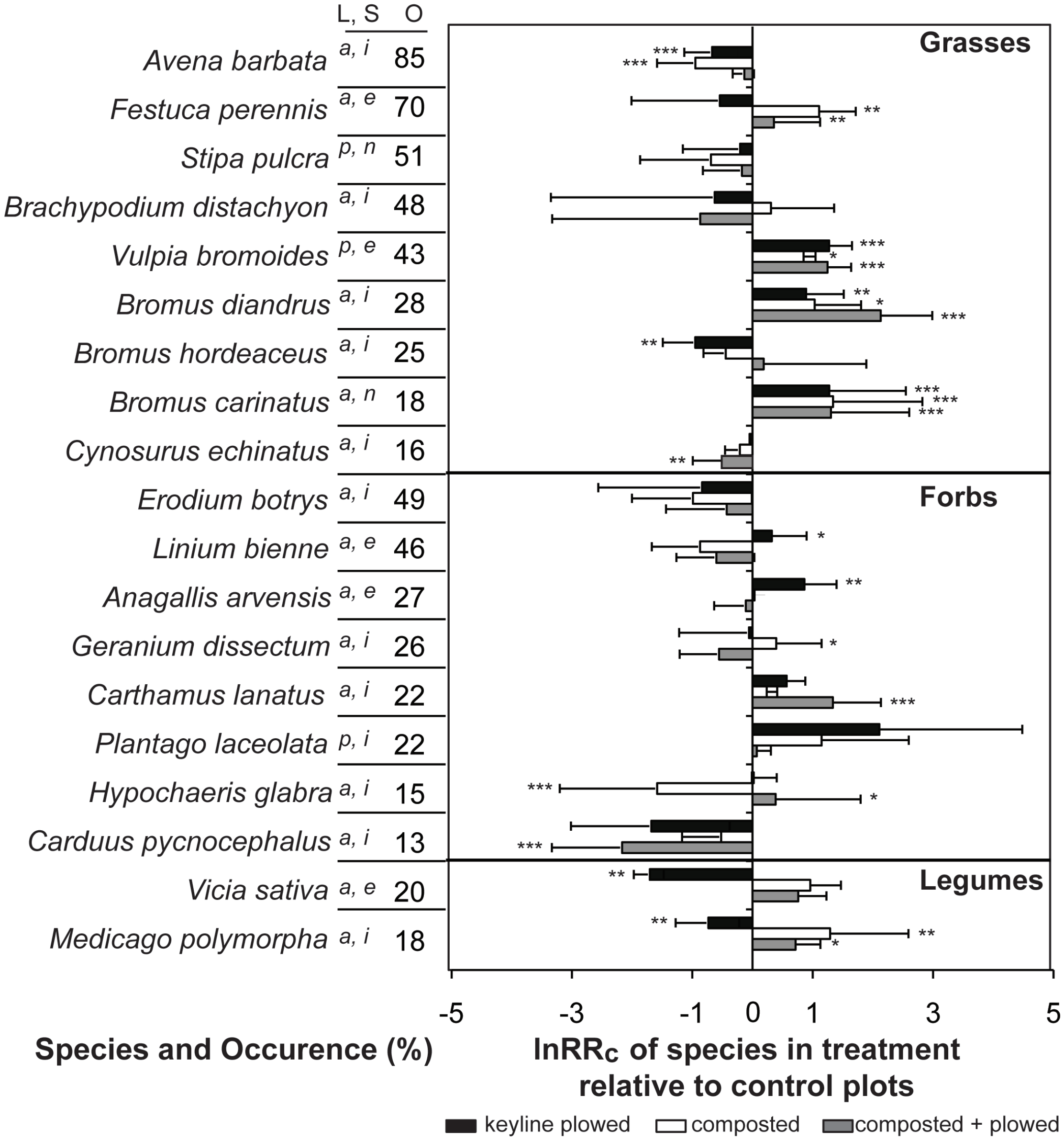
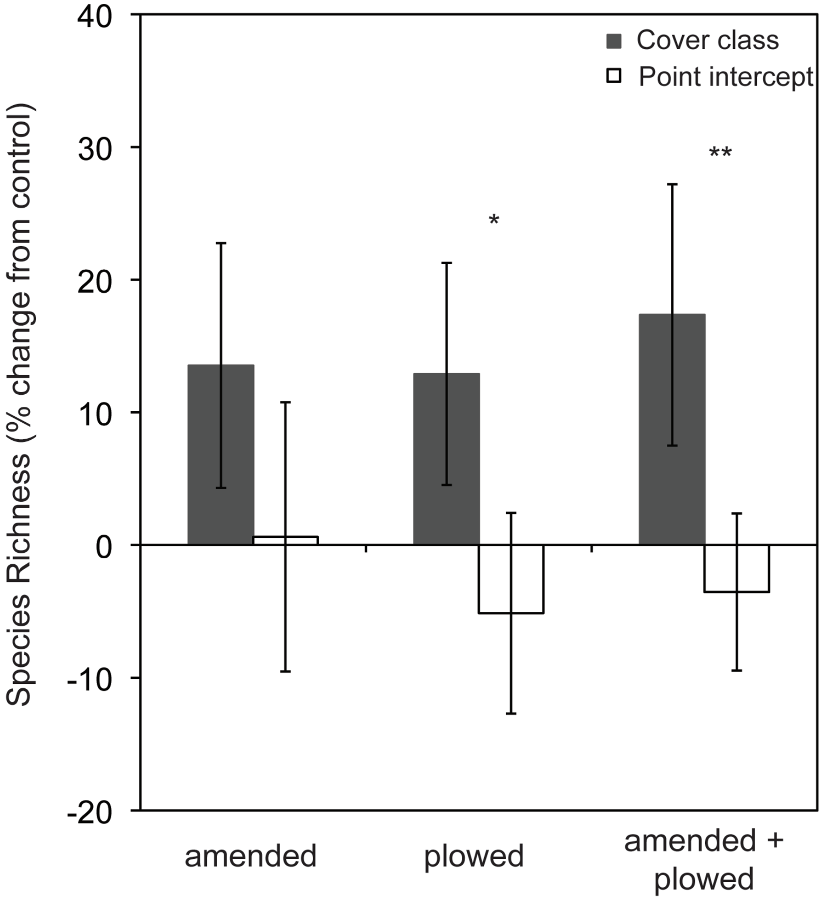


Figure 8. Comparison of cover class and point intercept vegetation survey methods. Bars represent change in species richness (averaged across sites and years) with management treatments. Error bars represent ± 1 standard error. * $p < 0.10$; ** $p < 0.05$.



Appendices

Appendix 1. Carbon pools and fluxes from control and organic matter amended plots for three water years at the valley and coastal grassland experimental sites. Values are plot averages, in units of mass of C per area, with standard errors in parentheses. ANPP = aboveground net primary production. BNPP = belowground net primary production (0-20 cm); Rs = total soil respiration. Net C storage is calculated using low (30 %), medium (50 %), and high (60%) scenarios of the fraction of heterotrophic respiration (Rh) to Rs.

Grassland	Treatment	ANPP (gC / m^2)	0-10 cm Root Biomass (gC / m^2)	10-20 cm Root Biomass (gC / m^2)	0-20 cm BNPP (gC / m^2)	NPP (gC / m^2)	Annual Rs (gC / m^2)	Net C Storage with 30% Rh (gC / m^2)	Net C Storage with 50% Rh (gC / m^2)	Net C Storage with 60% Rh (gC / m^2)
WY 1 (2008 – 2009)										
Valley 1	control	213 (26)	44.8 (5.5)	12.6 (1.5)	57.5 (5.7)	271 (26)	1190	-42.8	-252.0	-356.6
	amended	352 (25)	54.6 (3.9)	15.4 (1.1)	70.0 (4.0)	422 (25)	1442	68.1	-187.1	-314.8
Valley 2	control	189 (13)	35.7 (2.4)	10.1 (0.7)	45.7 (2.5)	234 (13)	1242	-104.5	-330.5	-443.8
	amended	373 (33)	43.3 (3.8)	12.2 (1.1)	55.5 (4.0)	428 (33)	1481	51.7	-218.9	-354.3
Valley 3	control	222 (20)	37.7 (3.4)	10.6 (1.0)	48.3 (3.5)	270 (20)	1145	-29.3	-229.1	-329.0
	amended	339 (24)	55.9 (3.9)	15.8 (1.1)	71.7 (4.1)	410 (24)	1397	58.4	-195.8	-322.8
Coastal 1	control	102 (22)	19.7 (4.3)	5.6 (1.2)	13.2 (3.3)	128 (23)	1053	-93.7	-241.2	-314.9
	amended	230 (33)	42.8 (6.1)	12.1 (1.7)	50.5 (7.8)	285 (34)	1256	64.4	-102.0	-185.2
Coastal 2	control	138 (15)	53.8 (5.7)	15.2 (1.6)	62.2 (8.2)	207 (16)	862	31.0	-86.5	-145.2
	amended	117 (34)	33.5 (9.6)	9.4 (2.7)	41.3 (7.0)	160 (36)	945	-19.0	-157.9	-227.4
Coastal 3	control	83 (15)	24.5 (4.5)	6.9 (1.3)	32.5 (5.1)	114 (16)	1160	-81.7	-212.4	-227.7
	amended	120 (7)	40.8 (2.4)	11.5 (0.7)	49.7 (9.9)	172 (8)	1417	-65.4	-243.4	-332.4

Grassland	Treatment	ANPP (gC / m^2)	0-10 cm Root Biomass (gC / m^2)	10-20 cm Root Biomass (gC / m^2)	0-20 cm BNPP (gC / m^2)	NPP (gC / m^2)	Annual Rs (gC / m^2)	Net C Storage with 30% Rh (gC / m^2)	Net C Storage with 50% Rh (gC / m^2)	Net C Storage with 60% Rh (gC / m^2)
WY 2 (2009 – 2010)										
Valley 1	control	206 (27)	44.0 (6.0)	11.6 (1.8)	55.5 (6.3)	261 (28)	1754	-214.6	-531.9	-690.6
	amended	354 (41)	55.5 (7.5)	14.9 (1.7)	70.4 (7.7)	424 (42)	1611	62.9	-220.5	-362.2
Valley 2	control	207 (14)	38.1 (4.5)	12.0 (1.8)	50.1 (4.9)	257 (15)	1301	-87.9	-317.8	-432.7
	amended	353 (32)	40.2 (3.0)	12.5 (2.7)	52.7 (4.0)	406 (33)	1799	-17.0	-341.6	-503.9
Valley 3	control	235 (20)	39.5 (3.0)	12.8 (0.2)	52.2 (3.0)	288 (21)	1515	-107.1	-370.2	-501.8
	amended	289 (23)	48.6 (4.5)	12.8 (1.1)	61.4 (4.6)	351 (24)	1697	-37.6	-339.2	-489.9
Coastal 1	control	270 (38)	29.9 (2.7)	14.5 (1.5)	44.5 (3.1)	315 (38)	2079	-231.9	-596.4	-778.6
	amended	237 (34)	30.0 (4.1)	14.3 (0.7)	44.3 (4.1)	281 (34)	2237	-247.9	-643.1	-840.7
Coastal 2	control	113 (30)	31.3 (5.5)	19.5 (3.7)	50.8 (6.7)	163 (31)	1515	-229.5	-491/5	-622.4
	amended	179 (24)	35.7 (3.8)	13.2 (2.5)	48.9 (4.5)	228 (24)	2171	-261.3	-630.3	-814.8
Coastal 3	control	283 (44)	63.5 (12.3)	21.0 (4.0)	84.5 (13.0)	367 (46)	2259	-212.1	-598.5	-791.7
	amended	422 (66)	98.9 (30.1)	42.7 (9.4)	141.6 (31.5)	563 (73)	2337	23.4	-379.0	-580.2
WY 3 (2010 – 2011)										
Valley 1	control	168 (13)	50.5 (4.0)	18.1 (2.9)	68.6 (4.9)	237 (14)	1029	-30.9	-209.6	-298.9
	amended	413 (72)	42.2 (2.7)	13.0 (2.6)	55.2 (3.7)	468 (72)	1243	216.0	-3.7	-113.5
Valley 2	control	193 (12)	37.0 (5.7)	14.0 (4.0)	71.5 (18.1)	265 (22)	1052	-30.8	-214.2	-305.9
	amended	286 (36)	61.1 (4.8)	22.2 (2.2)	83.3 (5.3)	370 (37)	1179	130.0	-81.6	-187.4
Valley 3	control	141 (12)	42.2 (2.0)	10.6 (2.1)	52.8 (3.0)	194 (13)	1144	-93.9	-285.9	-382.0
	amended	324 (19)	48.9 (5.2)	13.5 (2.6)	62.4 (5.8)	387 (20)	1452	94.2	-152.7	-276.2

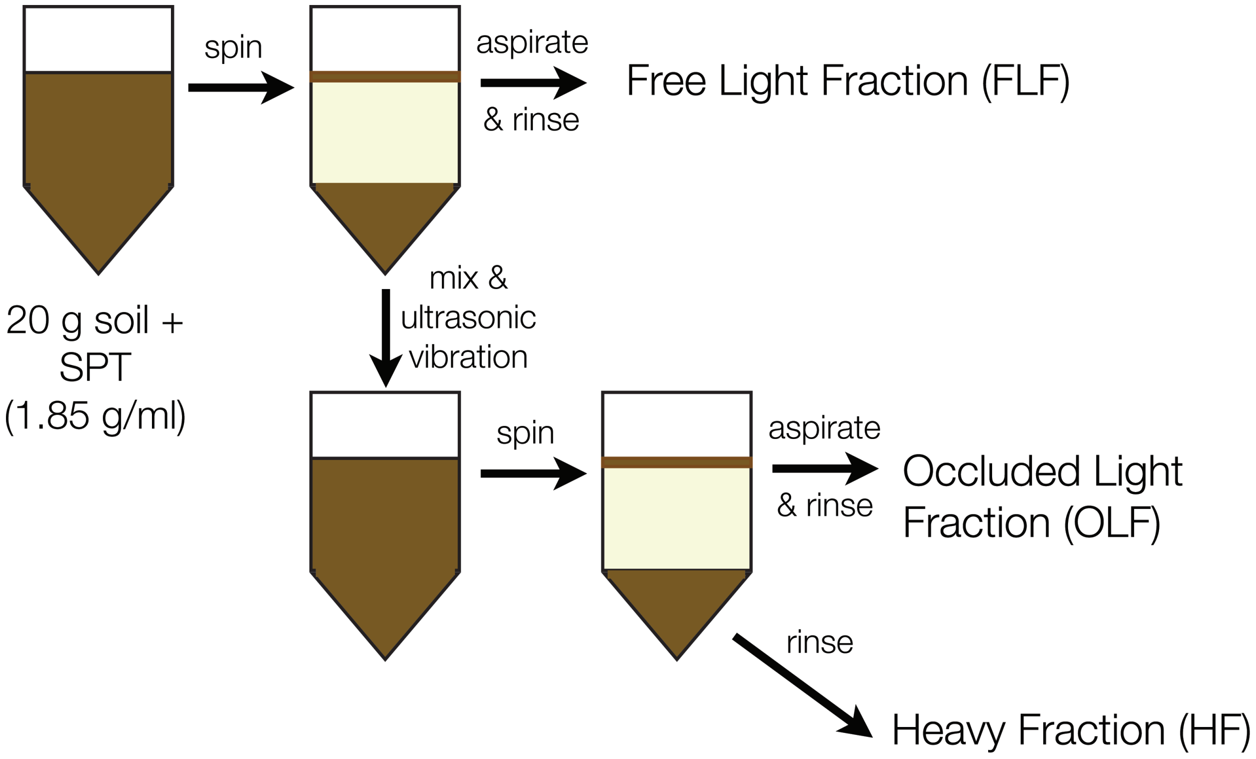
Grassland	Treatment	ANPP (gC / m^2)	0-10 cm Root Biomass (gC / m^2)	10-20 cm Root Biomass (gC / m^2)	0-20 cm BNPP (gC / m^2)	NPP (gC / m^2)	Annual Rs (gC / m^2)	Net C Storage with 30% Rh (gC / m^2)	Net C Storage with 50% Rh (gC / m^2)	Net C Storage with 60% Rh (gC / m^2)
Coastal 1	control	164 (11)	57.6 (9.8)	24.1 (8.3)	81.7 (12.8)	246 (17)	1774	-99.4	-329.5	-444.5
	amended	246 (31)	58.3 (10.6)	22.6 (4.0)	80.9 (11.3)	327 (33)	1828	23.8	-230.1	-357.0
Coastal 2	control	79 (19)	42.3 (4.6)	23.8 (4.8)	66.1 (6.7)	145 (20)	1313	-126.5	-307.8	-398.5
	amended	118 (12)	35.0 (4.0)	16.2 (1.7)	51.2 (4.4)	169 (13)	1804	-128.0	-378.1	-503.1
Coastal 3	control	178 (27)	53.4 (5.2)	25.8 (3.3)	79.2 (6.2)	257 (28)	1556	-32.7	-226.0	-322.6
	amended	225 (18)	84.8 (11.6)	55.1 (8.2)	140.0 (14.3)	365 (23)	1969	72.7	-174.0	-297.3

Appendix 2. Bulk carbon concentration and content from control and amended soils from valley and coastal grasslands. Pre-treatment samples were collected prior to compost application in November 2008 and at the end of each growing season, typically in April or May at the valley grassland and May or June at the coastal grassland. (Continued on next page)

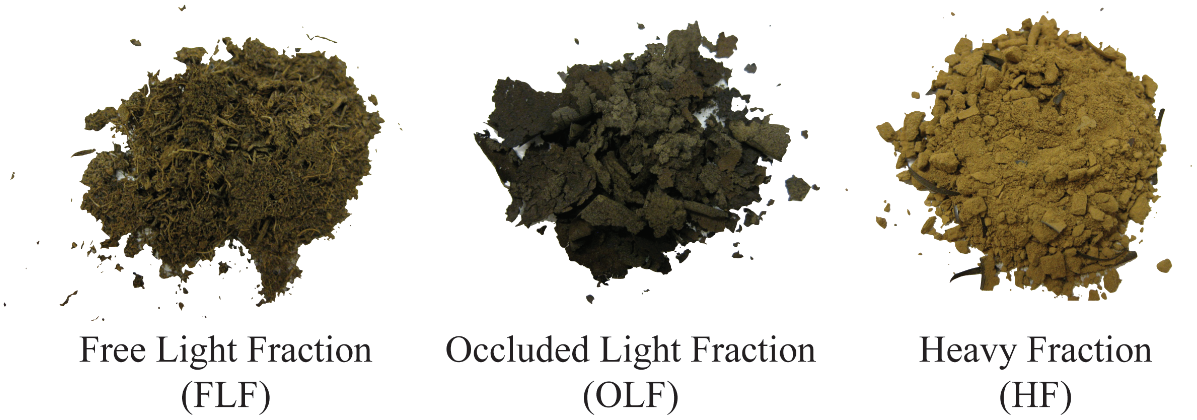
Grassland	Depth (cm)	Treatment	Bulk Density (g/cm^3)	Pre-Treatment				Year 1			
				C (%)	C (gC/m^2)	N (%)	N (gN/m^2)	C (%)	C (gC/m^2)	N (%)	N (gN/m^2)
Valley	0 – 10	control	0.98 (0.05)	2.73 (0.17)	2187 (114)	0.22 (0.02)	173 (13)	3.55 (0.35)	2867 (391)	0.28 (0.03)	223 (33)
		amended	1.05 (0.06)	2.93 (0.08)	2334 (246)	0.25 (0.01)	211 (11)	3.92 (0.10)	3563 (207)	0.32 (0.01)	290 (19)
	10 – 30	control	1.02 (0.04)	1.50 (0.17)	1226 (128)	0.11 (0.01)	92 (9)	nd	nd	nd	nd
		amended	1.03 (0.06)	1.41 (0.10)	824 (108)	0.11 (0.002)	65 (5)	nd	nd	nd	nd
	30 – 50	control	1.13 (0.06)	nd	nd	nd	nd	nd	nd	nd	nd
		amended	1.13 (0.07)	nd	nd	nd	nd	nd	nd	nd	nd
Coastal	0 – 10	control	1.06 (0.10)	3.11 (0.44)	2799 (177)	0.27 (0.03)	240 (6)	2.75 (0.63)	2656 (394)	0.23 (0.05)	224 (31)
		amended	0.87 (0.13)	2.91 (0.004)	2831 (90)	0.25 (0.001)	242 (9)	2.95 (0.54)	2808 (558)	0.25 (0.04)	237 (45)
	10 – 30	control	1.20 (0.06)	1.90 (0.43)	2215 (323)	0.17 (0.04)	202 (25)	1.60 (0.42)	1869 (455)	0.15 (0.03)	171 (35)
		amended	1.00 (0.04)	1.90 (0.23)	1610 (492)	0.17 (0.02)	129 (36)	1.59 (0.31)	1391 (261)	0.15 (0.03)	128 (24)
	30 – 50	control	1.11 (0.06)	0.88 (0.31)	1368 (234)	0.12 (0.02)	121 (13)	1.10 (0.27)	1188 (276)	0.10 (0.02)	113 (20)
		amended	1.11 (0.03)	1.45 (0.01)	1448 (135)	0.13 (0.0003)	129 (17)	1.05 (0.21)	1072 (136)	0.10 (0.01)	104 (9)
	50 - 100	control	1.26 (0.03)	0.47 (0.03)	578 (21)	0.06 (0.01)	73 (8)	0.79 (0.15)	976 (198)	0.08 (0.01)	95 (14)
		amended	1.27 (0.03)	0.57 (0.17)	712 (239)	0.06 (0.01)	81 (12)	0.81 (0.18)	1031 (232)	0.08 (0.01)	106 (14)

Grassland	Depth (cm)	Treatment	Year 2				Year 3			
			C (%)	C (gC/m^2)	N (%)	N (gN/m^2)	C (%)	C (gC/m^2)	N (%)	N (gN/m^2)
Valley	0 – 10	control	2.43 (0.18)	1954 (184)	0.19 (0.01)	151 (16)	2.87 (0.08)	2312 (144)	0.22 (0.01)	179 (12)
		amended	2.90 (0.19)	2615 (80)	0.24 (0.02)	212 (5)	3.00 (0.08)	2728 (221)	0.26 (0.01)	233 (14)
	10 – 30	control	1.14 (0.04)	1867 (82)	0.09 (0.004)	145 (6)	1.19 (0.10)	1867 (82)	0.09 (0.01)	77 (4)
		amended	1.27 (0.10)	1713 (313)	0.10 (0.002)	128 (17)	1.34 (0.11)	1713 (313)	0.11 (0.01)	75 (15)
	30 – 50	control	0.55 (0.01)	1085 (76)	0.05 (0.001)	94 (7)	0.73 (0.08)	1085 (76)	0.06 (0.01)	61 (6)
		amended	0.88 (0.15)	1890 (419)	0.07 (0.01)	142 (30)	0.68 (0.08)	1890 (419)	0.06 (0.01)	65 (11)
Coastal	0 – 10	control	2.55 (0.71)	2431 (480)	0.22 (0.06)	210 (39)	2.67 (0.63)	2569 (406)	0.24 (0.05)	229 (30)
		amended	2.60 (0.55)	2486 (589)	0.23 (0.05)	222 (50)	2.93 (0.53)	2797 (575)	0.26 (0.04)	249 (47)
	10 – 30	control	1.51 (0.44)	1766 (474)	0.14 (0.04)	163 (39)	1.69 (0.47)	1776 (347)	0.15 (0.04)	163 (26)
		amended	1.54 (0.36)	1357 (342)	0.14 (0.03)	127 (31)	1.72 (0.40)	1425 (382)	0.16 (0.03)	131 (32)
	30 – 50	control	1.13 (0.31)	1196 (269)	0.11 (0.02)	113 (20)	1.14 (0.29)	1331 (315)	0.11 (0.02)	124 (22)
		amended	1.03 (0.22)	1051 (210)	0.10 (0.02)	101 (18)	1.26 (0.32)	1135 (346)	0.11 (0.02)	101 (25)
	50 - 100	control	0.61 (0.16)	759 (210)	0.06 (0.01)	78 (16)	0.54 (0.08)	529 (83)	0.06 (0.0004)	61 (4)
		amended	0.54 (0.12)	681 (151)	0.06 (0.01)	76 (9)	0.70 (0.22)	739 (246)	0.07 (0.02)	73 (20)

Appendix 3. Schematic of soil physical density fractionation procedure.



Appendix 4. Images of soil density fractions. (Not to scale).



Appendix 5. Parameters used for DAYCENT simulations at valley, coastal- loam, and coastal-sandy loam grassland sites. Definitions for parameters can be found at http://www.nrel.colostate.edu/projects/daycent/.

A. Site Parameters (sitepar.in)

Note: Valley, coastal-loam, and coastal-sandy sites are simulated using the same site parameters, except for the first parameter in the file. Both extra weather drivers were used for the valley grassland simulations (0), whereas data for extra drivers was not available for the coastal sites (3).

```
0 or 3     / 0 = no extra drivers, 1 = PET drivers, 2 = psyn drivers, 3 = both
1.0        / sublimscale
0.18       / reflec - vegetation reflectivity (frac)
0.65       / albedo (frac) */
0.90       / fswcinit - initial swc, fraction of field capacity
0.000008  / dmpflux - in h2oflux routine (0.000001 = original value)
10         / hours_rain - duration of each rain event
0          / # of days between rainfall event and drainage of soil (-1=computed)
1  0       / watertable[month] - 0 = no water table, 1 = water table
2  0
3  0
4  0
5  0
6  0
7  0
8  0
9  0
10 0
11 0
12 0
-200       / hpotdeep - hydraulic water potential of deep storage layer (units?)
0.0003     / ksatdeep - saturated hydraulic conductivity of deep storage layer (cm/sec)
1  58      / cldcov[month] - cloud cover (%)
2  58
3  58
4  58
5  58
6  58
7  58
8  58
9  58
10 58
11 58
12 58
0.0 12.4   / min and max temperature for bottom soil layer (degrees C)
0.003      / damping factor for calculating soil temperature by layer
```

```
30.0      / timlag, days from Jan 1 to coolest temp at bottom of soil (days)
0.03      / min water/temperature limitation coefficient for nitrify
0 0       / turn off respiration restraint on denit between these Julian dates
0.6       / nitrification N2O adjustment factor (0.0-1.0)
1646.0    / elevation (meters)
0.0       / site slope, degrees
0.0       / site aspect, degrees
0.0       / site east horizon, degrees
0.0       / site west horizon, degrees
1  0.42   / solar radiation adjustment for cloud cover & transmission coeffient
2  0.50   / solar radiation adjustment for cloud cover & transmission coeffient
3  0.53   / solar radiation adjustment for cloud cover & transmission coeffient
4  0.57   / solar radiation adjustment for cloud cover & transmission coeffient
5  0.62   / solar radiation adjustment for cloud cover & transmission coeffient
6  0.69   / solar radiation adjustment for cloud cover & transmission coeffient
7  0.71   / solar radiation adjustment for cloud cover & transmission coeffient
8  0.66   / solar radiation adjustment for cloud cover & transmission coeffient
9  0.58   / solar radiation adjustment for cloud cover & transmission coeffient
10  0.52  / solar radiation adjustment for cloud cover & transmission coeffient
11  0.46  / solar radiation adjustment for cloud cover & transmission coeffient
12  0.45  / solar radiation adjustment for cloud cover & transmission coeffient
1.0       / slope for adjusting minimum temperature for VPD dewpoint calc
0.0       / intercept for adjusting minimum temperature for VPD dewpoint calc
1.16      / maximum carbon loss due to photodecomposition (ug C/KJ srad)
200.0     / litter biomass for full absorption of solar radiation (g biomass)
```

B. Soils parameters (soils.in) – Valley grassland

```
 0.0   2.0  1.00  0.42362  0.20159  0.80  0.30  0.42  0.16  0.05  0.08  0.00018  6.55
 2.0   5.0  1.00  0.42362  0.20159  0.20  0.24  0.42  0.16  0.03  0.06  0.00018  6.55
 5.0  10.0  1.00  0.42362  0.20159  0.00  0.18  0.42  0.16  0.01  0.04  0.00018  6.55
10.0  20.0  1.03  0.42362  0.20159  0.00  0.14  0.42  0.16  0.01  0.01  0.00018  6.55
20.0  30.0  1.03  0.42362  0.20159  0.00  0.09  0.42  0.16  0.01  0.00  0.00018  6.55
30.0  45.0  1.14  0.42362  0.20159  0.00  0.05  0.42  0.16  0.01  0.00  0.00018  6.55
45.0  60.0  1.14  0.42362  0.20159  0.00  0.00  0.42  0.16  0.01  0.00  0.00018  6.55
60.0  75.0  1.20  0.42362  0.20159  0.00  0.00  0.42  0.16  0.01  0.00  0.00018  6.55
75.0  90.0  1.20  0.42362  0.20159  0.00  0.00  0.42  0.16  0.01  0.00  0.00018  6.55
```

C. Soils parameters (soils.in) – Coastal-loam grassland

```
 0.0   2.0  0.91  0.28029  0.10059  0.80  0.30  0.42  0.16  0.05  0.08  0.00715  6.05
 2.0   5.0  0.91  0.28029  0.10059  0.20  0.24  0.42  0.16  0.03  0.06  0.00715  6.05
 5.0  10.0  0.91  0.28029  0.10059  0.00  0.18  0.42  0.16  0.01  0.04  0.00715  6.05
10.0  20.0  0.99  0.28029  0.10059  0.00  0.14  0.42  0.16  0.01  0.01  0.00715  6.05
20.0  30.0  0.99  0.28029  0.10059  0.00  0.09  0.42  0.16  0.01  0.00  0.00715  6.05
30.0  45.0  1.04  0.28029  0.10059  0.00  0.05  0.42  0.16  0.01  0.00  0.00715  6.05
45.0  60.0  1.04  0.28029  0.10059  0.00  0.00  0.42  0.16  0.01  0.00  0.00715  6.05
60.0  75.0  1.22  0.28029  0.10059  0.00  0.00  0.42  0.16  0.01  0.00  0.00715  6.05
75.0  90.0  1.22  0.28029  0.10059  0.00  0.00  0.42  0.16  0.01  0.00  0.00715  6.05
```

D. Soils parameters (soils.in) – Coastal-Sandy loam grassland

```
 0.0   2.0  1.04  0.23571  0.08404  0.80  0.30  0.42  0.16  0.05  0.08  0.00129  6.05
 2.0   5.0  1.04  0.23571  0.08404  0.20  0.24  0.42  0.16  0.03  0.06  0.00129  6.05
 5.0  10.0  1.04  0.23571  0.08404  0.00  0.18  0.42  0.16  0.01  0.04  0.00129  6.05
10.0  20.0  1.07  0.23571  0.08404  0.00  0.14  0.42  0.16  0.01  0.01  0.00129  6.05
20.0  30.0  1.07  0.23571  0.08404  0.00  0.09  0.42  0.16  0.01  0.00  0.00129  6.05
30.0  45.0  1.19  0.23571  0.08404  0.00  0.05  0.42  0.16  0.01  0.00  0.00129  6.05
45.0  60.0  1.19  0.23571  0.08404  0.00  0.00  0.42  0.16  0.01  0.00  0.00129  6.05
60.0  75.0  1.24  0.23571  0.08404  0.00  0.00  0.42  0.16  0.01  0.00  0.00129  6.05
75.0  90.0  1.24  0.23571  0.08404  0.00  0.00  0.42  0.16  0.01  0.00  0.00129  6.05
```

E. Crop parameters (crop.100)

Note: The same crop parameters were used for simulations of valley, coastal-loam, and coastal-sandy loam grassland sites.

	Annual Grass	Perennial Grass
'PRDX(1)'	1.5	1.7
'PPDF(1)'	20	23
'PPDF(2)'	45	47
'PPDF(3)'	2.5	2.5
'PPDF(4)'	2.5	1.5
'BIOFLG'	1	1
'BIOK5'	600	650
'PLTMRF'	1	1
'FULCAN'	100	100
'FRTCINDX'	2	1
'FRTC(1)'	0.5	0.6
'FRTC(2)'	0.1	0.2
'FRTC(3)'	90	108
'FRTC(4)'	0.3	0.8
'FRTC(5)'	0.3	0.2

'CFRTCN(1)'	0.7	0.7
'CFRTCN(2)'	0.4	0.4
'CFRTCW(1)'	0.25	0.75
'CFRTCW(2)'	0.3	0.45
'BIOMAX'	200	280
'"PRAMN(1,1)"'	25	35
'"PRAMN(2,1)"'	390	390
'"PRAMN(3,1)"'	340	90
'"PRAMN(1,2)"'	60	60
'"PRAMN(2,2)"'	390	390
'"PRAMN(3,2)"'	340	100
'"PRAMX(1,1)"'	30	35
'"PRAMX(2,1)"'	440	440
'"PRAMX(3,1)"'	440	100
'"PRAMX(1,2)"'	80	95
'"PRAMX(2,2)"'	440	440
'"PRAMX(3,2)"'	440	100
'"PRBMN(1,1)"'	40	50
'"PRBMN(2,1)"'	390	390
'"PRBMN(3,1)"'	340	100
'"PRBMN(1,2)"'	0	0
'"PRBMN(2,2)"'	0	0
'"PRBMN(3,2)"'	0	0
'"PRBMX(1,1)"'	50	55
'"PRBMX(2,1)"'	420	420
'"PRBMX(3,1)"'	420	100
'"PRBMX(1,2)"'	0	0
'"PRBMX(2,2)"'	0	0
'"PRBMX(3,2)"'	0	0
'"FLIGNI(1,1)"'	0.02	0.02
'"FLIGNI(2,1)"'	0.0012	0.0012
'"FLIGNI(1,2)"'	0.26	0.26
'"FLIGNI(2,2)"'	-0.0015	-0.0015
'"FLIGNI(1,3)"'	0.26	0.26
'"FLIGNI(2,3)"'	-0.0015	-0.0015
'HIMAX'	0	0
'HIWSF'	0	0
'HIMON(1)'	0	2
'HIMON(2)'	0	1
'EFRGRN(1)'	0.5	0
'EFRGRN(2)'	0.5	0
'EFRGRN(3)'	0.5	0
'VLOSSP'	0.04	0.15
'FSDETH(1)'	0.2	0.2
'FSDETH(2)'	0.7	0.7
'FSDETH(3)'	0.2	0.2

'FSDETH(4)'	150	160
'FALLRT'	0.2	0.2
'RDRJ'	0.05	0.05
'RDRM'	0.05	0.05
'RDSRFC'	0.14	0.14
'RTDTMP'	2	2
'CRPRTF(1)'	0.3	0
'CRPRTF(2)'	0	0
'CRPRTF(3)'	0	0
'MRTFRAC'	0.05	0.05
'SNFXMX(1)'	0	0
'DEL13C'	27	27
'CO2IPR'	1.1	1.3
'CO2ITR'	0.65	0.77
'''CO2ICE(1,1,1)'''	1.3	1
'''CO2ICE(1,1,2)'''	1	1
'''CO2ICE(1,1,3)'''	1	1
'''CO2ICE(1,2,1)'''	1.3	1.3
'''CO2ICE(1,2,2)'''	1	1
'''CO2ICE(1,2,3)'''	1	1
'CO2IRS'	1	1
'CKMRSPMX(1)'	0.1	0.1
'CKMRSPMX(2)'	0.15	0.15
'CKMRSPMX(3)'	0.05	0.05
'CMRSPNPP(1)'	0	0
'CMRSPNPP(2)'	0	0
'CMRSPNPP(3)'	1.25	1.25
'CMRSPNPP(4)'	1	1
'CMRSPNPP(5)'	4	4
'CMRSPNPP(6)'	1.5	1.5
'CGRESP(1)'	0.23	0.23
'CGRESP(2)'	0.23	0.23
'CGRESP(3)'	0.23	0.23
'NO3PREF(1)'	0.5	0.5
'CLAYPG'	6	6
'CMIX'	0.25	0.25
'TMPGERM'	10	10
'DDBASE'	1500	1500
'TMPKILL'	7	7
'BASETEMP(1)'	10	10
'BASETEMP(2)'	30	30
'MNDDHRV'	100	100
'MXDDHRV'	200	200
'CURGDYS'	120	120
'CLSGRES'	0.5	0.5
'CMXTURN'	0.12	0.12

'PS2MRSP(1)'	0.5	0.5
'AMAX(1)'	50	50
'AMAXFRAC(1)'	0.75	0.75
'AMAXSCALAR1(1)'	1	1
'AMAXSCALAR2(1)'	1	1
'AMAXSCALAR3(1)'	1	1
'AMAXSCALAR4(1)'	1	1
'ATTENUATION(1)'	0.57999	0.57999
'BASEFOLRESPFRAC(1)'	0.1	0.1
'CFRACLEAF(1)'	0.45	0.45
'DVPDEXP(1)'	-0.48	-0.48
'DVPDSLOPE(1)'	2.457	2.457
'GROWTHDAYS1(1)'	1	1
'GROWTHDAYS2(1)'	25	25
'GROWTHDAYS3(1)'	65	65
'GROWTHDAYS4(1)'	105	105
'HALFSATPAR(1)'	17.28	17.28
'LEAFCSPWT(1)'	270	270
'PSNTMIN(1)'	4	4
'PSNTOPT(1)'	24	24

F. Grazing parameters (graz.100)

Note: The same grazing parameters were used for simulations of valley, coastal-loam, and coastal-sandy loam grassland sites.

	GL Low Intensity	GM Moderate Intensity
'FLGREM'	0.1	0.1
'FDGREM'	0.05	0.01
'GFCRET'	0.3	0.3
'GRET(1)'	0.8	0.8
'GRET(2)'	0.95	0.95
'GRET(3)'	0.95	0.95
'GRZEFF'	0	1
'FECF(1)'	0.5	0.5
'FECF(2)'	0.9	0.9
'FECF(3)'	0.5	0.5
'FECLIG'	0.25	0.25

G. Organic matter amendment parameters (omad.100)

Note: Organic matter amendment parameters varied by scenario.

	SFREC-GREEN-WASTE-COMPOST					
	S11	S20	S30	M11	M20	M30
'OMADTYP'	2	2	2	2	2	2
'ASTGC'	1427	1427	1427	142.7	142.7	142.7
'ASTLBL'	1	1	1	1	1	1
'ASTLIG'	0.6	0.6	0.6	0.6	0.6	0.6
ASTREC(1)'	11	20	30	11	20	30
'ASTREC(2)'	99.2	99.2	99.2	99.2	99.2	99.2
'ASTREC(3)'	141.3	141.3	141.3	141.3	141.3	141.3

H. Site files – valley, coastal-loam, and coastal-sandy loam grasslands

*** Site and control parameters

	Valley	Coastal – loam	Coastal – sandy
'IVAUTO'	1	1	1
'NELEM'	1	1	1
'SITLAT'	39.24178	38.06703	38.06703
'SITLNG'	121.29561	122.71155	122.71155
'SAND'	0.41984	0.37783	0.54386
'SILT'	0.4158	0.43928	0.33255
'CLAY'	0.16436	0.18288	0.12359
'ROCK'	0	0	0
'BULKD'	1	1.0361	1.0361
'NLAYER'	5	5	5
'NLAYPG'	4	4	4
'DRAIN'	1	1	1
'BASEF'	0.5	0.5	0.5
'STORMF'	0	0	0
'PRECRO'	8	8	8
'FRACRO'	0.15	0.15	0.15
'SWFLAG'	0	0	0
'AWILT(1)'	0.2	0.2	0.2
'AWILT(2)'	0.2	0.2	0.2
'AWILT(3)'	0.2	0.2	0.2
'AWILT(4)'	0.2	0.2	0.2
'AWILT(5)'	0.2	0.2	0.2
'AWILT(6)'	0.2	0.2	0.2
'AWILT(7)'	0.2	0.2	0.2
'AWILT(8)'	0.2	0.2	0.2
'AWILT(9)'	0.2	0.2	0.2

'AWILT(10)'	0.3	0.3	0.3
'AFIEL(1)'	0.3	0.3	0.3
'AFIEL(2)'	0.3	0.3	0.3
'AFIEL(3)'	0.3	0.3	0.3
'AFIEL(4)'	0.3	0.3	0.3
'AFIEL(5)'	0.3	0.3	0.3
'AFIEL(6)'	0.3	0.3	0.3
'AFIEL(7)'	0.3	0.3	0.3
'AFIEL(8)'	0.3	0.3	0.3
'AFIEL(9)'	0.3	0.3	0.3
'AFIEL(10)'	0	0	0
'PH'	6.55	6.05	6.05
'PSLSRB'	1	1	1
'SORPMX'	2	2	2
*** External nutrient input parameters			
'EPNFA(1)'	0.5	0.5	0.5
'EPNFA(2)'	0.005	0.005	0.005
'EPNFS(1)'	30	30	30
'EPNFS(2)'	0.01	0.01	0.01
'SATMOS(1)'	0	0	0
'SATMOS(2)'	0	0	0
'SIRRI'	0	0	0
*** Organic matter initial values			
"'SOM1CI(1,1)'"	25	25	25
"'SOM1CI(1,2)'"	0	0	0
"'SOM1CI(2,1)'"	60	60	60
"'SOM1CI(2,2)'"	0	0	0
"'SOM2CI(1,1)'"	1415	1415	1415
"'SOM2CI(1,2)'"	0	0	0
"'SOM2CI(2,1)'"	1415	1415	1415
"'SOM2CI(2,2)'"	0	0	0
'SOM3CI(1)'	650	650	650
'SOM3CI(2)'	0	0	0
"'RCES1(1,1)'"	16	16	16
"'RCES1(1,2)'"	50	50	50
"'RCES1(1,3)'"	50	50	50
"'RCES1(2,1)'"	12	12	12
"'RCES1(2,2)'"	50	50	50
"'RCES1(2,3)'"	50	50	50
"'RCES2(1,1)'"	17	17	17
"'RCES2(1,2)'"	117	117	117
"'RCES2(1,3)'"	117	117	117
"'RCES2(2,1)'"	17	17	17
"'RCES2(2,2)'"	117	117	117
"'RCES2(2,3)'"	117	117	117
'RCES3(1)'	8	8	8

'RCES3(2)'	62	62	62
'RCES3(3)'	62	62	62
"'CLITTR(1,1)'"	50	50	50
"'CLITTR(1,2)'"	0	0	0
"'CLITTR(2,1)'"	35	35	35
"'CLITTR(2,2)'"	0	0	0
"'RCELIT(1,1)'"	66	66	66
"'RCELIT(1,2)'"	300	300	300
"'RCELIT(1,3)'"	300	300	300
"'RCELIT(2,1)'"	66	66	66
"'RCELIT(2,2)'"	300	300	300
"'RCELIT(2,3)'"	300	300	300
'AGLCIS(1)'	0	0	0
'AGLCIS(2)'	0	0	0
'AGLIVE(1)'	0	0	0
'AGLIVE(2)'	0	0	0
'AGLIVE(3)'	0	0	0
'BGLCIS(1)'	150	150	150
'BGLCIS(2)'	0	0	0
'BGLIVE(1)'	3	3	3
'BGLIVE(2)'	0.4	0.4	0.4
'BGLIVE(3)'	0.4	0.4	0.4
'STDCIS(1)'	25	25	25
'STDCIS(2)'	0	0	0
'STDEDE(1)'	0.8	0.8	0.8
'STDEDE(2)'	0.2	0.2	0.2
'STDEDE(3)'	0.2	0.2	0.2
*** Mineral initial parameters			
"'MINERL(1,1)'"	0.5	0.5	0.5
"'MINERL(2,1)'"	0.5	0.5	0.5
"'MINERL(3,1)'"	0.5	0.5	0.5
"'MINERL(4,1)'"	0.5	0.5	0.5
"'MINERL(5,1)'"	0	0	0
"'MINERL(6,1)'"	0	0	0
"'MINERL(7,1)'"	0	0	0
"'MINERL(8,1)'"	0	0	0
"'MINERL(9,1)'"	0	0	0
"'MINERL(10,1)'"	0	0	0
"'MINERL(1,2)'"	0.5	0.5	0.5
"'MINERL(2,2)'"	0	0	0
"'MINERL(3,2)'"	0	0	0
"'MINERL(4,2)'"	0	0	0
"'MINERL(5,2)'"	0	0	0
"'MINERL(6,2)'"	0	0	0
"'MINERL(7,2)'"	0	0	0
"'MINERL(8,2)'"	0	0	0

"'MINERL(9,2)'"	0	0	0
"'MINERL(10,2)'"	0	0	0
"'MINERL(1,3)'"	0.5	0.5	0.5
"'MINERL(2,3)'"	0	0	0
"'MINERL(3,3)'"	0	0	0
"'MINERL(4,3)'"	0	0	0
"'MINERL(5,3)'"	0	0	0
"'MINERL(6,3)'"	0	0	0
"'MINERL(7,3)'"	0	0	0
"'MINERL(8,3)'"	0	0	0
"'MINERL(9,3)'"	0	0	0
"'MINERL(10,3)'"	0	0	0
'PARENT(1)'	0	0	0
'PARENT(2)'	50	50	50
'PARENT(3)'	50	50	50
'SECNDY(1)'	0	0	0
'SECNDY(2)'	15	15	15
'SECNDY(3)'	2	2	2
'OCCLUD'	0	0	0

*** Water initial parameters

'RWCF(1)'	0	0	0
'RWCF(2)'	0	0	0
'RWCF(3)'	0	0	0
'RWCF(4)'	0	0	0
'RWCF(5)'	0	0	0
'RWCF(6)'	0	0	0
'RWCF(7)'	0	0	0
'RWCF(8)'	0	0	0
'RWCF(9)'	0	0	0
'RWCF(10)'	0	0	0
'SNLQ'	0	0	0
'SNOW'	0	0	0