

Long-term impacts of manure amendments on carbon and greenhouse gas dynamics of rangelands

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Abstract

Livestock manure is applied to rangelands as an organic fertilizer to stimulate forage production, but the long-term impacts of this practice on soil carbon (C) and greenhouse gas (GHG) dynamics are poorly known. We collected soil samples from manured and nonmanured fields on commercial dairies and found that manure amendments increased soil C stocks by 19.0 ± 7.3 Mg C ha⁻¹ and N stocks by 1.94 ± 0.63 Mg N ha⁻¹ compared to nonmanured fields (0–20 cm depth). Long-term historical (1700–present) and future (present–2100) impacts of management on soil C and N dynamics, net primary productivity (NPP), and GHG emissions were modeled with DayCent. Modeled total soil C and N stocks increased with the onset of dairying. Nitrous oxide (N₂O) emissions also increased by ~ 2 kg N₂O-N ha⁻¹ yr⁻¹. These emissions were proportional to total N additions and offset 75–100% of soil C sequestration. All fields were small net methane (CH₄) sinks, averaging -4.7 ± 1.2 kg CH₄-C ha⁻¹ yr⁻¹. Overall, manured fields were net GHG sinks between 1954 and 2011 (-0.74 ± 0.73 Mg CO₂e ha⁻¹ yr⁻¹, CO₂e are carbon dioxide equivalents), whereas nonmanured fields varied around zero. Future soil C pools stabilized 40–60 years faster in manured fields than nonmanured fields, at which point manured fields were significantly larger sources than nonmanured fields (1.45 ± 0.52 Mg CO₂e ha⁻¹ yr⁻¹ and 0.51 ± 0.60 Mg CO₂e ha⁻¹ yr⁻¹, respectively). Modeling also revealed a large background loss of soil C from the passive soil pool associated with the shift from perennial to annual grasses, equivalent to 29.4 ± 1.47 Tg CO₂e in California between 1820 and 2011. Manure applications increased NPP and soil C storage, but plant community changes and GHG emissions decreased, and eventually eliminated, the net climate benefit of this practice.

Keywords: carbon sequestration, DayCent, greenhouse gas emissions, manure, pasture, rangelands, soil

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Introduction

Greenhouse gas (GHG) emissions from livestock comprise approximately 20% of the total GHG emissions from agriculture globally, and nearly half of this is from livestock manure management (EPA, 2012). Manure application to rangelands is used to dispose of manure and provide an organic fertilizer to enhance forage production (Eghball & Power, 1999; Conant *et al.*, 2001; Bolan *et al.*, 2004; Kong *et al.*, 2005; Cabrera *et al.*, 2009). Long-term experiments on the effects of manure addition on soil C stocks and GHG fluxes are rare and have focused on cropland rather than rangeland (Christensen, 1988; Buyanovsky & Wagner, 1998; Gulde *et al.*, 2008; Sosulski *et al.*, 2014). Determining the degree to which ecosystem C sequestration is offset by GHG emissions is critical for evaluating the net climate change mitigation potential of organic matter amendments.

In the short-term, organic matter amendments such as manure typically increase soil C and nitrogen (N)

pools while improving other soil properties including aggregation, water-holding capacity, and erosion resistance (Clark *et al.*, 1998; Conant *et al.*, 2001; Kong *et al.*, 2005; Lynch *et al.*, 2005; Blair *et al.*, 2006; Risse *et al.*, 2006; Cabrera *et al.*, 2009; Simonetti *et al.*, 2012). Enhanced net primary production (NPP) contributes to greater soil carbon (C) content through increased litter inputs, root and microbial exudates, and root turnover (Conant *et al.*, 2001; Jones & Donnelly, 2004; Kell, 2011). Although a fraction of added organic matter is rapidly decomposed by microbes, some is incorporated into aggregates, physically protected from degradation by soil minerals, and/or biochemically protected (Six *et al.*, 2002; Jones & Donnelly, 2004; Kong *et al.*, 2005). Aggregated or physiochemically protected soil C and N pools are thought to turn over relatively slowly, on timescales of decades to millennia (e.g., Jastrow *et al.*, 1996). The proportion of added organic C and N that is sequestered in pools with slow turnover rather than rapidly decomposed is a function of soil characteristics, organic matter composition, and climate (Burke *et al.*, 1989; Pascual *et al.*, 1999; Jobbágy & Jackson, 2000; Lynch

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et al., 2005). Recent work suggests that relatively small additions of compost to rangeland soils can measurably increase soil C pools and NPP in the short term (Ryals & Silver, 2013) and, based on model simulations, may remain a net sink for several decades (Ryals *et al.*, 2015). The practice of compost additions to rangelands was recently incorporated into the offset methodology of the American Carbon Registry (Haden *et al.*, 2014).

Ecosystem C gains from some forms of organic matter amendments may be partially or fully offset by increased GHG emissions. Nitrous oxide (N₂O) emissions, in particular, typically increase with manure amendment (Lowrance *et al.*, 1998; Chadwick *et al.*, 2000; Bouwman *et al.*, 2002; Velthof *et al.*, 2003; Stehfest & Bouwman, 2006). The IPCC currently estimates that all types of organic matter amendments release 0.01 kg N₂O-N kg N_{added}⁻¹ yr⁻¹ (Lasco *et al.*, 2006). Actual N₂O emissions, however, depend on local climate, timing of amendment application, soil characteristics, and amendment composition (Simek *et al.*, 2002; Stehfest & Bouwman, 2006; Snyder *et al.*, 2009). Manure amendments are likely to have higher N₂O emissions than composted organic matter because composting slows rates of N mineralization (Stehfest & Bouwman, 2006; Aguilera *et al.*, 2013a).

We used fields on commercial dairies as long-term experiments (>60 years) testing the effects of different rangeland management practices on soil C and N dynamics. We hypothesized that manure additions would increase soil C content, but that increased GHG emissions could potentially offset some or all C gains over the long term. We used the DayCent model to simulate soil C and GHG gas fluxes from the sampled fields in order to evaluate the long-term global warming potential (GWP) of manure applications to rangelands.

Materials and methods

Field sampling and soil analysis

Soils were collected from fields on ten dairies in Marin and Sonoma counties, California, USA, between November 2011 and March 2012. These sites are rangelands as defined by the USDA (2009): land on which plant cover (climax, subclimax, or potential) is composed principally of grasses, grass-like plants, forbs, or shrubs suitable for grazing and browsing, including native and introduced plant species. The region has a typical Mediterranean climate with dry, warm summers and wet, mild winters. Mean annual temperature is 14.5 °C, and mean annual precipitation is 70 cm. The dairies had been operating for at least 60 years, often managed by several generations of the same family. Current and historical management practices were determined through interviews with the farmers and reviewing literature on local history (Livingston, 1993, 1995; Avery, 2009).

At each dairy, soils were collected from 2 to 4 grazed fields for a total of 26 fields with varying management approaches (Table S1). Eleven fields were amended only with solid manure, two only with liquid manure, four with both solid and liquid manure, and nine had no amendment. Other management techniques were recorded (e.g., aeration, seeding, and seasonal irrigation) and used as additional variables in our analyses. Typical of this region, all fields were vegetated with diverse exotic annual grasses which were introduced in the early 1800s and replaced native perennial and annual grasses (Jackson & Bartolome, 2007). Most manure-amended fields and some unamended fields had been more recently seeded with a non-native pasture mix (Table S1). Fields on the same dairy were located on similar soil types based on the NRCS online soil database (<http://websoilsurvey.nrcs.usda.gov/>) and had similar slopes. Soil was sampled volumetrically from 0 to 5 cm, 5 to 10 cm, 10 to 20 cm, 20 to 30 cm, and 30 to 50 cm depths using metal corers (a 6.5-cm-diameter corer from 0 to 20 cm and a 5.5-cm-diameter corer from 20 to 50 cm) at 5 locations along a 40-m-long transect.

The soils were air-dried and passed through a 2-mm sieve. The separated rocks were weighed to calculate rock concentration. One subsample was oven-dried at 105 °C to calculate oven-dry mass for calculating bulk density (as soil oven-dry mass divided by volume). Roots and other visible plant fragments were removed from an air-dried subsample that was then ground to a fine powder with a ball grinder (SPEX Sample Prep Mixer Mill 8000D, Metuchen, NJ, USA). The powdered samples were analyzed for total C and N concentration on a Carlo Erba Elantech elemental analyzer (Lakewood, NJ, USA) using atropine as a standard at a rate of one per ten samples. No carbonates were detected when the soils were tested with 2M HCl; thus, the total C analysis reflected total organic C content. Soil C and N contents were calculated by multiplying the C and N concentrations (%) by the oven-dry mass of the fine fraction (<2 mm) and dividing by the total sample volume (Throop *et al.*, 2012). Soil pH was measured in a 1:1 volumetric slurry of soil and deionized water. Texture was analyzed by hydrometer on soil samples from one location in each field. The samples were not treated with hydrogen peroxide to remove organic matter, but this likely introduced an error of <5% to the texture calculations (Bouyoucos, 1951).

Statistical tests were performed using JMPPro v.11 (SAS Institute, Inc., Cary, NC, USA). Field data (C, N, pH, and bulk density) considered by depth and management (manured vs. nonmanured) were not normally distributed and had unequal population sizes, but did have statistically equal variances (determined using analysis of means for variances with Levene, $\alpha = 0.1$). Therefore, means were compared using analysis of means (ANOM) with transformed rankings ($\alpha = 0.1$). Statistical significance was determined as $P < 0.10$.

DayCent modeling

In order to explore the long-term impacts of manure amendments to rangelands, we used the biogeochemical land surface model DayCent, the daily time-step version of the widely used CENTURY model (Parton *et al.*, 1998). It is parameterized with

site-specific soil and vegetation characteristics, daily weather data, and user-specified management events. The outputs include GHG fluxes and C and N content of vegetation and soil pools. Soil C and N contents are reported for total content and also for three pools operationally defined by their turnover times: fast (1–5 years), slow (20–40 years), and passive (200–1500 years). Net primary productivity is reported as aboveground productivity (ANPP) and belowground productivity (BNPP). We used the latest version of DayCentUV (July 2014) that includes subroutines for photosynthesis and photodecomposition of surface litter.

DayCent was parameterized using measured soil and site characteristics, including texture, bulk density, and pH. Daily climate data were obtained from the Western Regional Climate Center for the Petaluma Fire Station #3 weather station in Petaluma, CA, the nearest station to the study areas (ranging from 7 to 30 km distant). Climate data were available from 1954 through 2012 (the last complete year available when the model was run). These data were used in a repeating pattern until 1953 after which (beginning in 1954) the climate data were synchronized with the model date. Beyond 2012, the climate data were again applied in a repeating pattern. Management events were scheduled and parameterized in accordance with information provided by the farmers and historical records (Livingston, 1993, 1995; Avery, 2009). Event timing and manure application rates were iteratively adjusted until soil C stocks calculated by the model for 2011 were within 20% of the measured values (with three exceptions: A2, E2, and G3, see Results). Ammonia volatilization is not yet included as a N cycling pathway in DayCent, but 4% to as much as 90% of urine N may be volatilized, as well as 0–10% of solid manure N (Hristov *et al.*, 2011), depending on climate, vegetation, and manure properties (Petersen & Sommer, 2011). As a result, DayCent can overestimate the amount of ammonium N remaining in the soil, particularly at high N concentrations. We manipulated maximum daily nitrification rates to indirectly increase leaching of N as nitrate (rather than guess a volatilization factor), iteratively increasing from the default value of 0.4 g N m⁻² to 2.0–4.0 g N m⁻². The model was run for each field; then, the model outputs were pooled into manured ($n = 17$) and nonmanured ($n = 9$) fields for comparison. All types of manured fields were pooled due to the limited number of fields with liquid manure ($n = 2$) and both liquid and solid manure amendment ($n = 4$).

Manure spreading and the direct deposition of manure derived from supplemental feed were parameterized in DayCent's OMAD and FERT input files (Tables S2 and S3). Most farmers said that they try to apply solid manure as thinly as possible, but the total amount varies depending on the equipment used, the operator, and manure characteristics. Applications of 0.5 cm, 1 cm, and 2 cm thickness were parameterized, equivalent to 280 g C m⁻², 570 g C m⁻², and 1200 g C m⁻², respectively, assuming a bulk density of 0.5 g cm⁻³ and a C concentration of 12% based on field measurements of a solid manure pile and literature values (Heinrich, 2009; Eldridge *et al.*, 2013).

Liquid manure required both OMAD and FERT events to account for the addition of organic solids and dissolved

inorganic N. Liquid manure varies widely in composition depending on the duration of storage and degree of dilution for land application (e.g., Heinrich, 2009). The dairies in this study managed their liquid manure similarly, storing it in ponds and then diluting it for land application. Liquid manure data were provided by a farmer who had contracted a commercial laboratory. This slurry had an organic matter content of 3150 $\mu\text{g g}^{-1}$ (equivalent to 1575 $\mu\text{g g}^{-1}$ C assuming organic matter is 50% C), total N of 270 $\mu\text{g g}^{-1}$, with 26 $\mu\text{g g}^{-1}$ as organic N and 244 $\mu\text{g g}^{-1}$ as ammonium N (NH₄-N), and a bulk density of ~ 1 g cm⁻³, with approximately 1% solids. These values were within the range reported in the literature for liquid manure composition on California dairies (Heinrich, 2009). Most farmers said they applied approximately 2.5 cm of liquid manure; this application rate and a 30% heavier one of 3.3 cm were parameterized in the model. The 2.5-cm-deep application was equivalent to 40 g C m⁻² and 6 g NH₄-N m⁻², and the 3.3-cm-deep application was equivalent to 52 g C m⁻² and 8 g NH₄-N m⁻² (Tables S2 and S3). Liquid added during these events accounted for less than 5% of mean annual precipitation, and because it was added before the growing season, it was not likely to have much impact on plant growth or GHG fluxes (J.J. Owen and W.L. Silver, unpublished data).

During grazing events in DayCent, nutrients are automatically cycled from the consumed biomass back to the soil to mimic the deposition of manure on the fields from grazing livestock. In California and other regions with Mediterranean climates, however, cows are often on fields where vegetation has senesced for the summer and the cows are given supplemental feed. These cows do not remove much biomass, but they do deposit manure derived from the supplemental feed and we treated this as an external organic matter addition in the model. Feces were treated as an organic matter input to the surface litter pool and urine as a fertilizer input (Tables S2 and S3). Published values of cow manure excretion rates vary widely due to their dependence on cow age, breed, stage in milk, size, diet, and environmental factors (Nennich *et al.*, 2005). We used mean values from Nennich *et al.* (2005) of 43.2 \pm 0.3 kg feces day⁻¹ and 15.5 \pm 0.1 kg feces day⁻¹ for milk cows and heifers, respectively. We collected and analyzed fresh feces samples ($n = 60$) from milk cows in the region (J.J. Owen and W.L. Silver, in preparation) to determine its dry matter (DM) C content, which averaged 16 \pm 0.4% DM and 0.36 \pm 0.01 g C g⁻¹ DM. Grazing intensity was estimated based on interviews with farmers. A typical string of cows was 100 hd and spent a month on an area of 6 ha, with heifers spending 100% of their time on the field and milk cows spending 75% of their time. Over 30 days at the stocking density described above, heifers and milk cows deposited 43 and 93 g C m⁻², respectively.

Heifer and milk cows produce different amounts of urine (9.0 \pm 0.4 kg day⁻¹ and 23.1 \pm 0.3 kg day⁻¹, respectively) with different N content (59.7 \pm 3.9 and 216.5 \pm 2.8 g urine N day⁻¹, respectively) (Nennich *et al.*, 2005). Using the same field size, stocking density, and duration, the urine from heifers and milk cows applied 3 and 8 g N m⁻², respectively (Table S3). Urine N is almost entirely in the form of urea

(Bristow *et al.*, 1992) and is rapidly converted into NH_4 (Bristow *et al.*, 1992; Bussink & Oenema, 1998) so the N addition was modeled as 100% NH_4 .

The model baseline was initiated with a perennial grassland, the study region's historic vegetation cover (Heady *et al.*, 1977; Avery, 2009). Low-intensity grazing by native and introduced grazers was modeled for the growing season (December to May). Native Americans in the region practiced prescribed burning, and fire recurrence was modeled as every 20 years (Keeley, 2002). The intensity of these prescribed burns was very low (Avery, 2009) and likely negligible in a model where fire is not required to maintain the grass cover. Following standard protocol, the initial amount of passive soil C was adjusted for each field so that total soil C, passive soil C, and C:N ratio of the soil organic matter stabilized within 1000 years.

Exotic annual grasses were introduced by Spanish missionaries during the early 1800s, and widespread invasion and conversion to non-native annual grasses was complete by 1850 (Jackson & Bartolome, 2007). We set the transition from perennial to annual grasses at 1820 in the model and continued low-intensity grazing. The 1920s–1950s marked the onset of dairying characterized by less reliance on rangeland, more use of supplemental summer feed, and manure spreading. These practices were widespread by the mid-1950s.

Beginning in 1954, fields which were reported as having manure amendments were modeled as such and fields which reported as having no manure amendments were modeled with either no amendments or manure inputs derived from supplemental feed, all at rates that achieved the measured soil C and N in 2011. Of the nine nonmanured fields, two required no organic matter amendments at all (fields A1 and G3) and six required manure deposition derived from supplemental feed (four at the lower heifer rate and two at the higher milk cow rate); one required light solid manure spreading (field I2). For fields E1, F1, and H2, manure deposition derived from supplemental feed was modeled beginning in 1921 because starting in 1954 did not increase soil C to 2011 levels in time. DayCent was then used to project rangeland ecosystem properties and nutrient fluxes to 2100 using the business-as-usual scenarios established between 1954 and 2011. Modeled values represent our best estimates of potential long-term trends for Mediterranean annual grasslands and not exact values for any given field. As our primary goal was to determine the long-term effects of manure applications on ecosystem processes, these projections did not account for background global changes including climate change, rising atmospheric CO_2 concentrations, or increasing anthropogenic N deposition.

The net climate impacts (i.e., GWP) of the fields were calculated as the sum of net direct GHG emissions associated with manure additions and net changes in ecosystem C (the sum of changes in ANPP, BNPP, and total soil C in Mg C ha^{-1}). We did not include indirect N_2O to focus on direct emissions; thus, we have calculated minimum estimates of N_2O fluxes from the fields. All components were converted into CO_2 equivalents; the CO_2 -equivalent GWPs of

N_2O and CH_4 over 100 years were 298 and 34, respectively (Myhre *et al.*, 2013). We used the sign convention that positive indicates loss from the ecosystem and negative indicates gain/consumption. For soil nutrients and GHG fluxes, we compared time points coincident with the maximum response to management changes: 1750 represented the initial perennial grass system, 1921 was the annual grass system under light beef cattle grazing, 1954 was low-intensity dairying, 2011 was the current state of the system (intensive dairying), and 2100 was the potential future state assuming a business-as-usual scenario. Due to the large interannual variability of ANPP and BNPP typical of these rangelands, comparing values at individual time points was not representative of average, long-term ecosystem change. Thus, we used the differences between the mean values of ANPP and BNPP at peak biomass (May) from four time periods associated with predominant management practices: 1720–1820 (perennial baseline), 1821–1953 (extensive use and annual grass invasion), 1954–2011 (intensive use), and 2012–2100 (future intensive use).

Model sensitivity analysis

DayCent is a complex simulation model with thousands of interacting parameters, but only a small subset must be set by the user. Uncertainty arises from user adjustable parameters, such as organic matter input rates and the maximum nitrification rate. It also arises from the way the model handles some parameters, such as soil bulk density and pH which are held constant through time despite the occurrence of processes well known to change them. Our goal in using the model was to capture the general trajectories of soil C and GHG fluxes over time, but we wanted to know how sensitive our results were to these two sources of uncertainty. The complexity of DayCent makes it unamenable to traditional sensitivity tests such as Monte Carlo analysis, nor did we have 'control' vs. 'experimental' plots with which to analyze variance (Ogle *et al.*, 2007). Thus, to test the sensitivity of model outputs to varying manure amendment rates, nitrification rates, bulk density, and pH, the model was rerun for a representative field (B1) that had a bulk density profile similar to the mean of all the fields and no manure amendments except 4 months of summer grazing by heifers.

To show the effect of different manure amendments, we ran the model with no amendments, with 4 months of summer grazing by heifers and by dairy cows, and with annual additions of each of the manure amendments listed in Table S2 (except chicken manure as this was rarely applied), and then with the minimum and maximum combinations of solid and liquid manures (minimal solid with basic liquid and maximum solid and high liquid, respectively). To test the effect of different maximum daily nitrification rates, the model was run using the default value of $0.4 \text{ g N m}^{-2} \text{ d}^{-1}$, and with $2.0 \text{ g N m}^{-2} \text{ d}^{-1}$, $4.0 \text{ g N m}^{-2} \text{ d}^{-1}$, and $8.0 \text{ g N m}^{-2} \text{ d}^{-1}$ (with this last value $2\times$ greater than any used in the model runs). To test the sensitivity to bulk density and pH, the model was rerun using the maximum and minimum bulk densities and pH values of all fields for each soil depth.

Results

Field measurements

Total soil C and N contents (0–50 cm) ranged from 60.1 ± 1.8 to 222.8 ± 6.4 Mg C ha⁻¹ and 5.92 ± 0.14 to 18.79 ± 0.51 Mg N ha⁻¹, respectively (Table S3), with about half found in the surface (0–20 cm) soil (36.5 ± 2.4 to 96.6 ± 7.2 Mg C ha⁻¹ and 3.87 ± 0.25 to 9.31 ± 0.75 Mg N ha⁻¹, respectively, Table 1). Soil C concentrations varied as much within dairies as between them and generally decreased with depth (Fig. 1). Manure addition significantly increased mean soil C and N concentrations by $1.07 \pm 0.81\%$ C and $0.12 \pm 0.07\%$ N, respectively, in the 5- to 10-cm interval, and by $0.88 \pm 0.68\%$ C and $0.08 \pm 0.06\%$ N, respectively, in the 10- to 20-cm interval (Fig. 2). Although the difference between manured and nonmanured fields at 0–5 cm was not statistically significant, the trend was

consistent with the other near-surface soils. Mean depth-averaged values of soil C and N contents for the top 20 cm in manured fields were significantly greater than nonmanured fields by 19.0 ± 7.3 Mg C ha⁻¹ and 1.94 ± 0.63 Mg N ha⁻¹, respectively.

Manure addition significantly decreased bulk density in the 0- to 5-cm and 5- to 10-cm intervals by 0.19 ± 0.08 and 0.14 ± 0.06 g cm⁻³, respectively (Fig. 3). This difference was largely due to the accumulation of soil organic matter; soil bulk density was significantly negatively correlated with soil C concentration at all depths (Fig. 4). Manure addition significantly increased pH by 0.56 ± 0.25 pH units in the 20–50 cm depths.

Soil texture was unaffected by manure addition. Clay increased with depth in most fields, ranging from 10 to 14% in the top 20 cm and 12 to 20% in the deeper soil (Table S4). Mean clay concentration for all the fields was $15 \pm 1\%$ (to 50 cm). Clay concentration was

Table 1 Comparison between field measurements and model outputs for top 20 cm. Positive values in the ‘difference between field and model’ columns indicate that the modeled value was greater than the field-measured value

Field	Modeled		Measured		Modeled		Difference between field and model			
	Manured	Manure type	Soil C (Mg C ha ⁻¹)	Soil N (Mg N ha ⁻¹)	Total C (Mg C ha ⁻¹)	Total N (Mg N ha ⁻¹)	Soil C (Mg C ha ⁻¹)	Soil N (Mg N ha ⁻¹)	Soil C (%)	Soil N (%)
A1	N	N	36.5 ± 2.4	3.9 ± 0.2	40.4	3.5	4.0	-0.4	11	-11
A2	Y	S	45.0 ± 3.8	4.4 ± 0.3	57.2	4.8	12.2	0.5	27	11
B1	N	N	51.5 ± 2.1	4.7 ± 0.2	51.8	4.2	0.3	-0.5	1	-11
B2	Y	B	86.9 ± 4.8	7.6 ± 0.4	88.2	6.9	1.3	-0.7	1	-9
C1	Y	S	85.1 ± 4.8	8.2 ± 0.4	84.7	6.6	-0.4	-1.6	0	-19
C2	N	N	89.9 ± 1.9	8.2 ± 0.3	82.0	6.7	-7.8	-1.5	-9	-18
C3	Y	S	84.9 ± 4.8	7.7 ± 0.5	84.6	6.9	-0.3	-0.8	0	-11
C4	Y	S	96.6 ± 7.2	9.3 ± 0.7	107.4	8.3	10.8	-1.0	11	-11
D1	Y	S	71.2 ± 3.1	7.8 ± 0.6	75.5	5.7	4.3	-2.2	6	-27
D2	N	N	51.3 ± 1.5	4.8 ± 0.2	49.3	4.0	-2.0	-0.8	-4	-16
E1	N	N	77.6 ± 2.7	7.1 ± 0.2	77.7	6.4	0.0	-0.7	0	-10
E2	Y	S	65.3 ± 3.0	6.5 ± 0.3	90.6	6.0	25.3	-0.5	39	-7
E3	Y	B*	96.5 ± 2.3	8.7 ± 0.2	106.6	7.4	10.1	-1.3	10	-15
F1	N	N	53.7 ± 6.4	5.4 ± 0.6	51.1	4.2	-2.6	-1.2	-5	-22
F2	Y	L	63.0 ± 5.5	6.5 ± 0.5	58.2	4.9	-4.8	-1.6	-8	-25
G1	Y	B	54.4 ± 2.0	5.6 ± 0.2	63.5	5.2	9.1	-0.4	17	-6
G2	Y	S	49.3 ± 2.8	5.2 ± 0.3	57.0	4.7	7.7	-0.5	16	-10
G3	N	N	36.9 ± 1.0	3.4 ± 0.1	50.9	4.4	14.0	1.0	38	31
H1	Y	S	92.3 ± 8.6	8.7 ± 0.7	94.0	7.4	1.7	-1.4	2	-16
H2	N	N	60.1 ± 1.5	5.6 ± 0.1	67.0	5.6	6.9	0.0	11	-1
H3	Y	S	87.4 ± 2.8	8.5 ± 0.3	71.5	5.7	-15.9	-2.9	-18	-34
I1	Y	S	66.9 ± 5.2	6.1 ± 0.4	60.6	5.0	-6.3	-1.1	-9	-18
I2	Y*	S*	82.9 ± 4.6	7.8 ± 0.5	73.6	5.8	-9.3	-2.0	-11	-26
J1	Y	B	85.5 ± 5.8	8.0 ± 0.6	82.1	6.5	-3.4	-1.5	-4	-19
J2	Y	B	89.5 ± 6.4	8.4 ± 0.6	83.3	6.6	-6.1	-1.8	-7	-21
J3	Y	S	67.9 ± 6.0	6.6 ± 0.6	72.9	5.6	5.0	-1.0	7	-15

N, not managed this way; Y, yes, this management occurred; N, no manure amendments; S, solid manure amendment; L, liquid manure amendments; B, both liquid and solid manure amendments.

*The value is different from what was reported by the farmers.

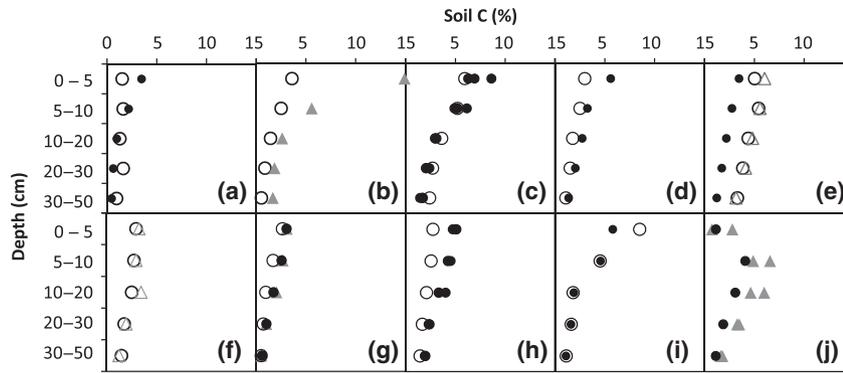


Fig. 1 Soil C concentration by depth by field on each dairy (a–j). Open circles indicate fields with no manure additions, closed circles indicated fields with solid manure additions, open triangles indicate fields with liquid manure additions, and solid triangles indicate fields with both solid and liquid manure additions. Standard error bars were smaller than marker size and are not shown.

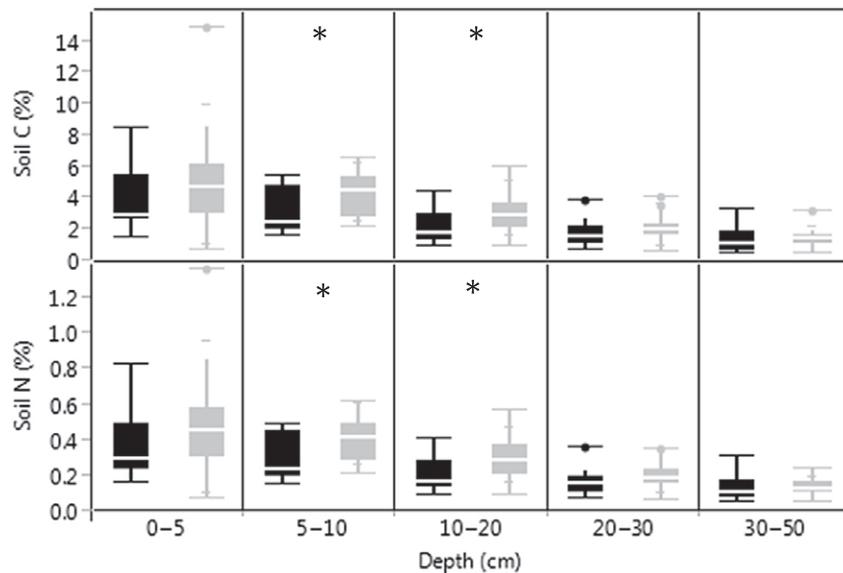


Fig. 2 Box plots of soil C and soil N concentration of nonmanured (black) and manured (grey) fields by sample interval. Outliers are labeled with solid dots. Significantly different concentrations are labeled with asterisks.

positively correlated with soil C concentration in the 10–20 cm ($r^2 = 0.38$, $P \leq 0.0008$) and 20–30 cm ($r^2 = 0.34$, $P < 0.003$) intervals. Additional management activities (seeding, aeration, etc.) did not significantly affect soil properties.

DayCent model parameterization

Modeled soil C contents matched field measurements (Fig. 5a, Table 1), with a few exceptions that revealed the sensitivity of the model to manure addition rate and frequency. Run without any manure additions, the soil C content of field G3 (reported as a nonmanured field) was overestimated by ~ 14 Mg C ha $^{-1}$ (38%), suggesting other, unknown management practices had

decreased soil C content. Two reportedly manured fields, A2 and E2, were modeled with the lightest solid manure addition possible yet the predicted soil C was ~ 12 and 25 Mg C ha $^{-1}$ greater than the field measured value, respectively, equivalent to overestimates of 27 and 39%. Rerunning the model with no inputs brought the modeled total C contents to within 4% of the measured values, suggesting that manure additions may have been less frequent. Although field I2 was reportedly too steep for manure amendment, manure additions (as light solid manure annually) were necessary to increase the modeled soil C content to that which was measured. The sampling transect was located near the top of the slope where the gradient was gentle; thus, the area which was sampled may have had manure

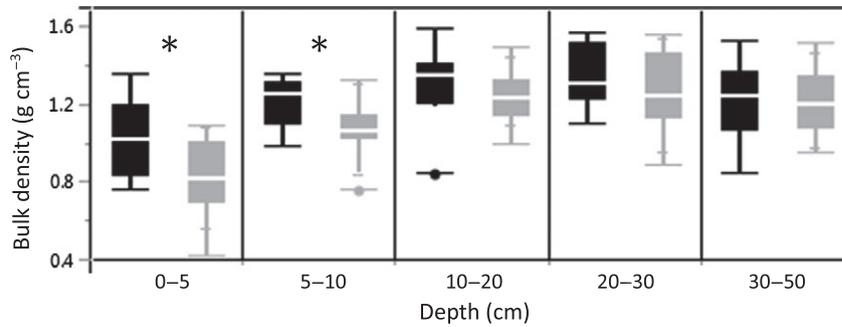


Fig. 3 Box plots of soil bulk density of nonmanured (black) and manured (grey) fields by sample interval. Outliers are labeled with solid dots, and significantly different concentrations are labeled with asterisks.

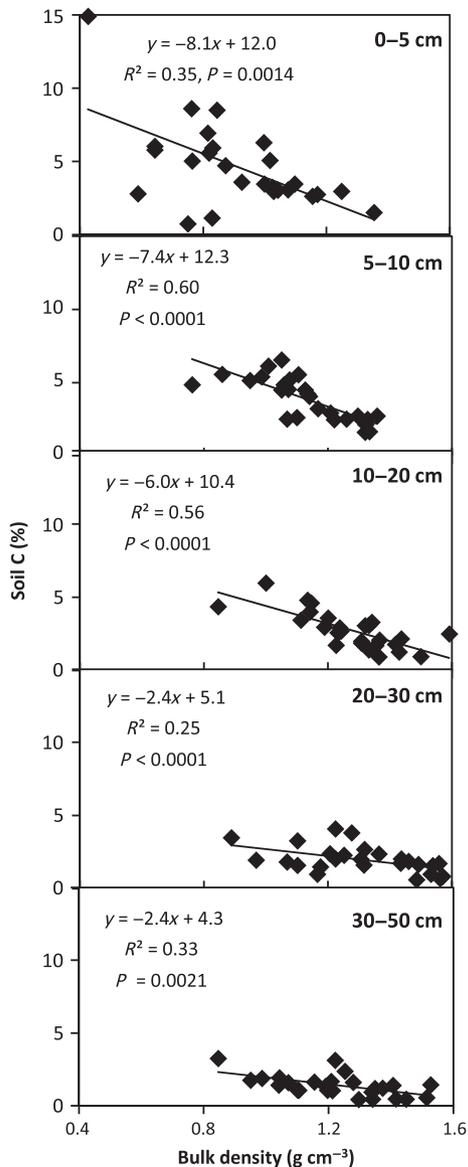


Fig. 4 Bulk density and soil C concentration by sample interval (in cm). All correlations are significant.

amendment, while the rest of the field did not. Lastly, field E3 reportedly had only liquid manure amendments but required annual additions of light solid manure, too, in order to increase soil C to the field measured value. Liquid manure on this dairy may have had higher solids content than our estimated value or solid manure may have been applied in the past, of which the current farmer may have been unaware. These examples highlight the challenge of modeling amendments with limited data and how various combinations of manure addition rates and timing could produce the same soil C results. Overall, management impacts on soil C were well represented in the model once uncertainties in management events were addressed.

Modeled and measured soil N contents were similar for soil N < 6.0 Mg N ha⁻¹, but were underestimated by 10–30% at higher soil N contents regardless of management (Fig. 5b). The differences between measured and modeled soil C content were positively correlated with the differences between measured and modeled soil N content ($r^2 = 0.64$, $P < 0.0001$), suggesting some systematic effect in the model. The correlation was offset from 0 such that good agreement between measured and modeled values of soil C for a given field did not imply that values of soil N were also in good agreement.

Management impacts on soil C and N

Modeled soil C was sensitive to management and responses varied among soil C pools (Fig. 6). The slow pool accounted for 40–60% of total C, while the passive pool accounted for 30–50% of total C, but was always smaller than the slow pool. Total soil C averaged 48.1 ± 13.0 Mg C ha⁻¹ during the perennial grass run-up (Fig. 6a). A small periodic variation was present during the run-up and mirrored natural patterns in precipitation. The shift from perennial to annual grasses in 1820 (and considered through 1920)

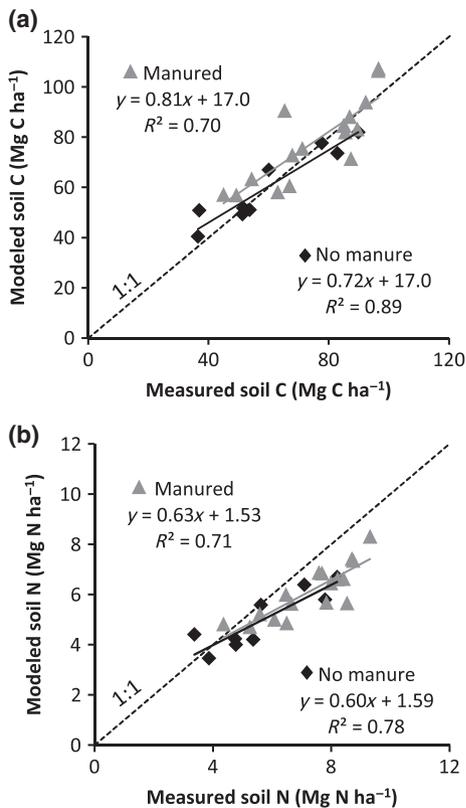


Fig. 5 Comparison between field data and modeled soil C (a) and N (b) content for the top 20 cm of soil (modeled data from 2011).

decreased total soil C by -62.4 ± 2.3 kg C ha⁻¹ yr⁻¹. Most of the soil C decrease occurred in the slow C pool (Fig. 6c), but the fast and passive pools also declined (Fig. 6b, d). With the onset of manure deposition derived from supplemental feed on some fields in 1921, total and slow soil C pools increased slightly, while the passive pool continued declining at the same rate as it had between 1820 and 1920 (~ -7 kg C ha⁻¹ yr⁻¹). The nonsignificant deviation in soil C between manured and nonmanured fields in 1921 (Fig. 6) was an effect of treating these fields separately beyond 1921 and parameterizing the model with modern soil bulk density.

Between 1954 (the onset of manure amendments) and 2011, total soil C increased by 697 ± 77 kg C ha⁻¹ yr⁻¹ in manured fields (a total of 36.9 ± 4.1 Mg C ha⁻¹), more than twice the increase of 276 ± 79 kg C ha⁻¹ yr⁻¹ in nonmanured fields (a total of 14.6 ± 4.2 Mg C ha⁻¹) (Fig. 6a). Most of this change occurred in the slow pool (Fig. 6c). Despite overall increases in total soil C, the passive C pool continued to decline: -3.26 ± 0.82 kg C ha⁻¹ yr⁻¹ in manured fields and -6.20 ± 0.69 kg C ha⁻¹ yr⁻¹ in the nonmanured fields. Organic matter C additions to the fields (including only

direct inputs from livestock derived from supplemental feed and manure amendments, not recycled C from grazing) ranged from 0 to 7.96 Mg C ha⁻¹ yr⁻¹ between 1954 and 2011, with mean total inputs to nonmanured fields about half of those to manured fields (2.45 ± 0.66 vs. 5.33 ± 0.45 Mg C ha⁻¹ yr⁻¹, respectively). Modeled total soil C content in 2011 increased linearly with total C inputs between 1954 and 2011 (Fig. 7). The slopes of the linear fits in Fig. 7 indicate that approximately $11 \pm 1\%$ of added C was retained, either directly or by increasing plant growth, during this period.

Projecting beyond 2011, manured fields were predicted to gain another 6.95 ± 4.22 Mg C ha⁻¹ and nonmanured fields to gain 1.60 ± 3.73 Mg C ha⁻¹ before soil C content stabilized. In manured fields, passive soil C pools increased 2.06 ± 1.35 kg C ha⁻¹ yr⁻¹ between 2011 and 2100, whereas in nonmanured fields, the passive soil C pool continued declining at a rate of -4.01 ± 1.09 kg C ha⁻¹ yr⁻¹. On average, only 2% of added C was retained in the soil during this period and, in some fields, continued organic matter additions could not offset soil C loss. Nitrogen pools followed the same patterns as C pools, and mineral N content was proportional to total C content.

Management impacts on NPP

The change in vegetation type and the increase in mineral N additions contributed to changes in NPP (Fig. 8). Net primary production varied widely due to its sensitivity to interannual climate variability, with a coefficient of variation of typically 20–30% and as high as 40% (Fig. 8a). In order to consider overall change in productivity due to management, we compared mean ANPP and BNPP for the periods 1720–1820, 1821–1953, 1954–2011, and 2012–2100. Total NPP during the perennial grassland run-up was 2.40 ± 0.03 Mg C ha⁻¹ yr⁻¹, with a greater proportion belowground than aboveground. The shift to annual grasses increased ANPP by 601 ± 49 kg C ha⁻¹ yr⁻¹ and decreased BNPP by 966 ± 13 kg C ha⁻¹ yr⁻¹. Subsequent management changes had smaller effects on ANPP and no effect on BNPP. Aboveground NPP increased an additional 0.24 ± 0.18 Mg C ha⁻¹ yr⁻¹ in manured fields between the 1821 and 1953 and the 1954–2011 blocks, and by 0.17 ± 0.11 Mg C ha⁻¹ yr⁻¹ in nonmanured fields. These increases were not significantly different between the manured and nonmanured fields. Average NPP values for the 1954–2011 and 2012–2100 blocks were not significantly different for either treatment.

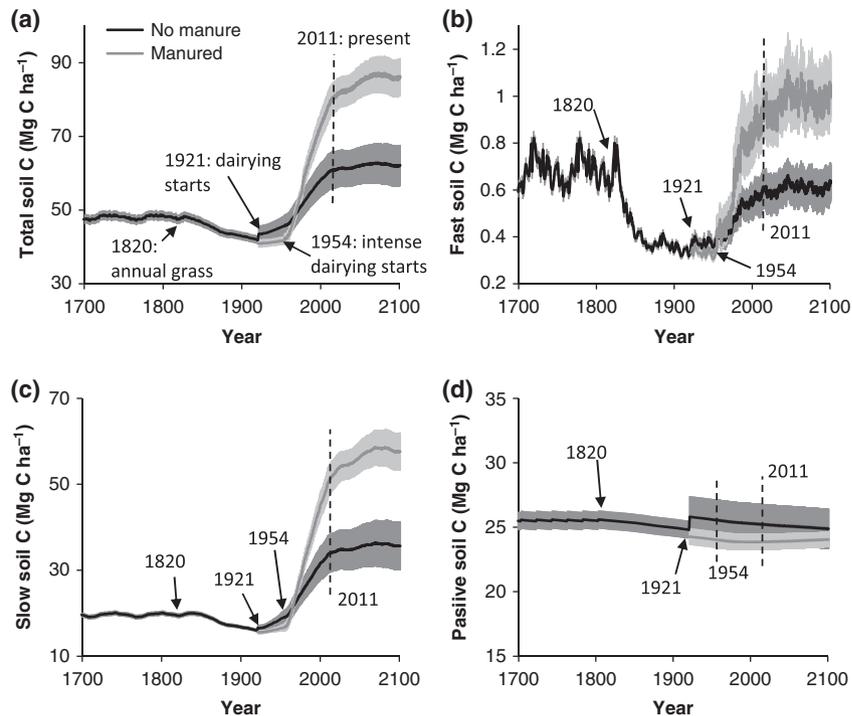


Fig. 6 Modeled soil C content for (a) total C, (b) fast pool, (c) slow pool, and (d) passive pool. Shaded areas around the data lines are standard errors. Vertical dashed lines denote years with important changes to management parameters. The apparently abrupt split in passive soil C pools (d) between manured and nonmanured fields in 1921 is an artifact of the model parameterization and data analysis.

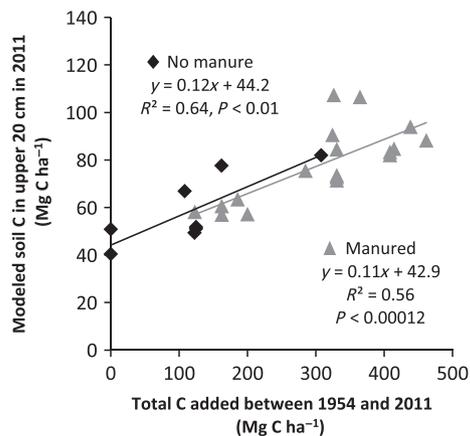


Fig. 7 Modeled soil C content vs. total C added as organic matter inputs from 1954 through 2011.

Modeled GHG emissions and net climate impacts of manure amendments

Mean annual N_2O emissions increased with manure additions from approximately $1.3 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ during the perennial run-up to $2.2 \pm 0.5 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ and $3.9 \pm 0.4 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ from nonmanured and manured fields, respectively, between

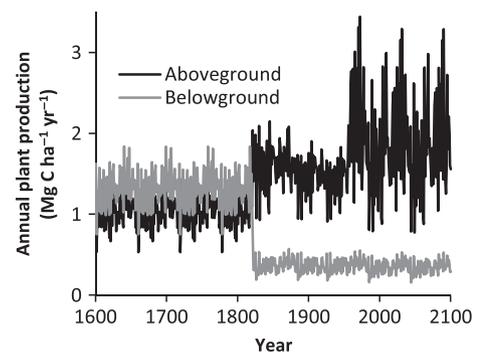


Fig. 8 Example of annual model data for aboveground and belowground production in field G2.

1954 and 2011 (Fig. 9). Nitrous oxide emissions were strongly positively correlated with total N additions (Fig. 10), and the slope of the regression lines indicated that approximately 0.75% of added N was subsequently emitted as $\text{N}_2\text{O-N}$.

Annual CH_4 oxidation rates were not affected by the change in vegetation type or management, averaging a net uptake by soil of $-4.7 \pm 1.2 \text{ kg CH}_4\text{-C ha}^{-1} \text{ yr}^{-1}$. Manured fields generally had lower annual CH_4 oxidation rates and smaller ranges in monthly CH_4 oxidation rates than nonmanured fields, but this difference was

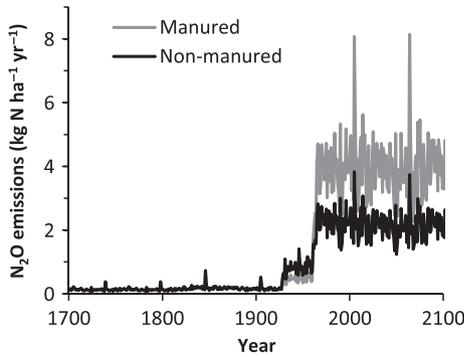


Fig. 9 Mean annual N_2O emissions from manured and nonmanured fields. The standard error was smaller than the interannual variability and is not shown for clarity.

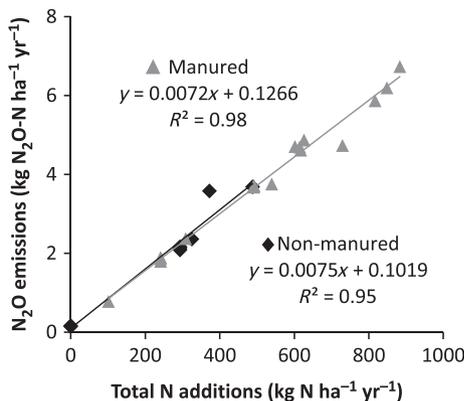


Fig. 10 Nitrous oxide emissions vs. total N additions (sum of N from fertilizer and manure) by field for the period 1954 to 2011.

not significant when the mean oxidation rates of all manured and all nonmanured fields were compared. Soil texture appeared to have the greatest influence on the range of monthly CH_4 oxidation rates; sand and silt contents (which were strongly, negatively correlated; $r^2 = 0.89$, $P < 0.0001$) were positively correlated with the range of monthly CH_4 oxidation rates ($r^2 = 0.49$, $P < 0.0001$).

The net climate impacts of vegetation change and dairying were calculated for the periods between 1820–1920, 1954–2011, and 2012–2100 which correspond to the shift in vegetation from perennial to annual grasses, intensive management, and continuing intensive management, respectively. The replacement of perennial grasses with annuals decreased soil C to such an extent that the ecosystem was a net source to the atmosphere from 1820 to 1920 ($16.4 \pm 2.6 \text{ Mg CO}_2\text{e ha}^{-1}$, Fig. 11). Although the decrease in BNPP was nearly twice as big as the gain in ANPP (3.54 ± 0.05 and $-2.05 \pm 0.05 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$, respectively), these changes in annual productivity were much

smaller than the cumulative impacts of soil C loss ($0.23 \pm 0.02 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$ for a cumulative $23.0 \pm 1.0 \text{ Mg CO}_2\text{e ha}^{-1}$) and GHG fluxes. Methane oxidation rate was constant at $-0.22 \pm 0.01 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$ for a cumulative $-22.3 \pm 1.0 \text{ Mg CO}_2\text{e ha}^{-1}$; N_2O emissions averaged $0.14 \pm 0.01 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$ for a cumulative $14.3 \pm 0.5 \text{ Mg CO}_2\text{e ha}^{-1}$. Between 1954 and 2011, manured fields were generally a net C sink ($-42.3 \pm 27.0 \text{ Mg CO}_2\text{e ha}^{-1}$) with high rates of soil C sequestration ($-2.38 \pm 0.26 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$) offsetting increased N_2O emission rates ($1.84 \pm 0.19 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$, more than an order of magnitude increase). In contrast, the net balance of nonmanured fields varied around zero ($-6.7 \pm 28.5 \text{ Mg CO}_2\text{e ha}^{-1}$, Fig. 11), with annual gains in soil C ($-0.94 \pm 0.27 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$) and continued CH_4 oxidation almost entirely offset by N_2O emissions ($1.04 \pm 0.21 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$). Projecting into the future, manured fields were large net sources between 2012 and 2100 ($146.0 \pm 22.3 \text{ Mg CO}_2\text{e ha}^{-1}$) and nonmanured fields were smaller net sources ($70.7 \pm 20.8 \text{ Mg CO}_2\text{e ha}^{-1}$). Mean soil CH_4 and N_2O flux rates did not change between 1954 and 2100; the change in ecosystem balance from source to sink was caused by soil C sequestration rates decreasing to 0.01 ± 0.004 and $-0.01 \pm 0.005 \text{ Mg CO}_2\text{e ha}^{-1} \text{ yr}^{-1}$ in the nonmanured and manured fields, respectively.

Model sensitivity analysis results

Soil C and N contents were very sensitive to organic matter inputs in the model. Carbon addition rate and the modeled soil C at 2100 were strongly linearly correlated ($r^2 = 0.85$, $P < 0.0001$) as were N addition rate and the modeled soil N at 2100 ($r^2 = 0.94$, $P < 0.0001$). Different amendment rates resulted in varying soil C accumulation trajectories which indicated that after about 5 decades, annual amendment rate was more important than the duration of annual amendments in determining soil C. Nitrous oxide fluxes increased linearly with N addition rate at a rate of $0.0073 \text{ g N}_2\text{O-N g N}_{\text{added}}^{-1}$ ($r^2 = 0.87$, $P < 0.001$), if the two highest inputs (high solid manure and high solid manure plus high liquid manure) were excluded. The high manure addition cases had lower-than-expected N_2O emissions and decreased the slope and strength of the correlation (slope = $0.0032 \text{ g N}_2\text{O-N g N}_{\text{added}}^{-1}$, $r^2 = 0.75$, $P \leq 0.001$). Methane fluxes were not affected by manure amendments.

Changing maximum daily nitrification rates had no effect on soil C and N, NPP, or CH_4 flux. Increasing nitrification rates from $0.4 \text{ g N m}^{-2} \text{ d}^{-1}$ to $2.0 \text{ g N m}^{-2} \text{ d}^{-1}$ tripled N_2O emissions after 1954 from an average value of $0.08 \text{ g N m}^{-2} \text{ yr}^{-1}$ to $0.22 \text{ g N m}^{-2} \text{ yr}^{-1}$, but further

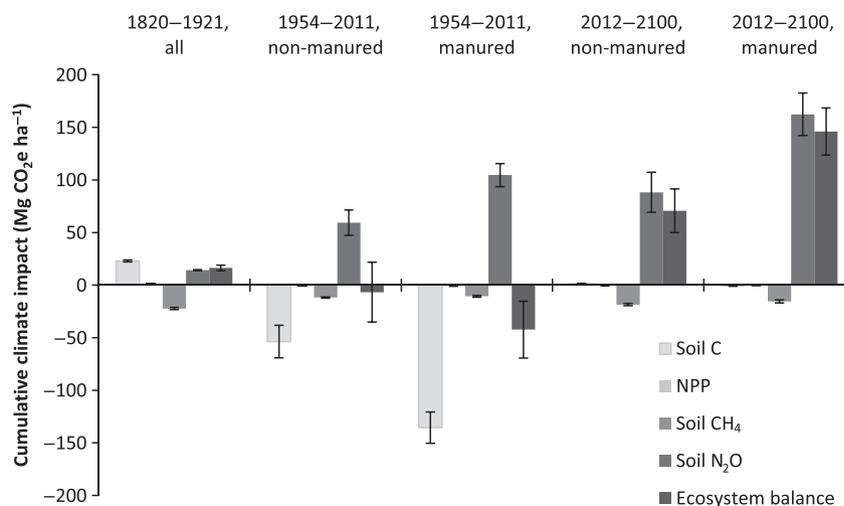


Fig. 11 The mean (\pm standard error) cumulative climate impacts of soil greenhouse gas emissions, changes in soil C storage and NPP, and the net climate impact from 1820 to 1921, 1954 to 2011, and 2012 to 2100 for nonmanured and manured fields. These timespans represent the changes due to the shift in vegetation from perennial to annual grasses, intensive management of the past 60 years, and continued intensive management, respectively.

increases in maximum daily nitrification rates had no effect.

Maximum and minimum bulk densities increased and decreased, respectively, total soil C by 10%, with similar scale impacts on individual soil C and N pools. The opposite was true for pH: low pH values increased soil C and N contents, while high pH values decreased them, with a range of $\pm 10\%$. Modeled N₂O emissions changed by less than 1% in response to changing bulk density, but were increased and decreased by approximately 5% by higher and lower pH, respectively. Soil pH did not affect modeled CH₄, but high bulk densities halved CH₄ oxidation rates and low bulk densities doubled them.

Discussion

Effects of manure amendments on soil C and N pools: field data

Our field data showed that manure amendments added 19.0 ± 7.3 Mg C ha⁻¹ in the top 20 cm of mineral soil relative to nonmanured fields. Other studies comparing the soil C and N contents of manured and nonmanured fields have found significant increases due to manure additions in agricultural fields (Edmeades, 2003; Blair *et al.*, 2006; Gulde *et al.*, 2008; Eagle *et al.*, 2012; Aguilera *et al.*, 2013b) and rangelands (Lynch *et al.*, 2005; Cabrera *et al.*, 2009). No other manure addition studies have reported soil C and N contents below 15–20 cm. We detected significant increases in soil C stocks for the 5–10 cm and 10–20 cm depths; average soil C in the

0- to 5-cm interval was greater in manured plots but, due to large variation, was not significantly so. Given the duration of manure addition to the amended fields, we hypothesized that soil C and N would have increased at all depths, rather than only in the near-surface soils (5–20 cm depth) as was observed. However, soil C decreased sharply below 20 cm in most fields, indicating that organic matter transport deeper into the soil was limited. Clay content increased with depth and was positively correlated with soil C concentrations from 10 to 30 cm depth, suggesting that complexation with clay minerals was a potential mechanism for C storage (Six *et al.*, 2002). Bulk density also increased with depth, and in combination with increasing clay content, this would have decreased pore volume and decreased movement of larger organic matter particles downward. This effect was likely exacerbated by the change in vegetation from perennial to annual grasses; the smaller, shallower roots of annual grasses would not have input C directly to deeper soils, nor opened larger pores for organic matter transport from above. The lack of increased C storage at depth may ultimately limit the residence time of added C in these soils.

The range in soil C stocks measured by this study was comparable to the range reported for the top 50 cm (28–137 Mg C ha⁻¹) of rangeland soils throughout California (Silver *et al.*, 2010). The range in surface (0–10 cm) soil C, 12.1–70.8 Mg C ha⁻¹, was comparable to C stocks calculated for 600 sites in the Great Plains of the USA (10–90 Mg C ha⁻¹) (Burke *et al.*, 1989). Thus, despite the small geographic extent of the sites in this study, the patterns in soil C stocks may be broadly

applicable to grass-dominated, managed rangelands across the western USA and globally (D'Antonio & Vitousek, 1992).

Evaluation of model parameterization and sensitivity

Modeled soil C and N contents were tuned to agree with measured values (within 20%) by adjusting the rate and duration of manure amendments within the constraints of typical management practices. The sensitivity analysis showed that the differences between fields were most dependent on manure amendments, with soil characteristics and other model parameters having much smaller effects, typically <10%. Thus, our modeling analysis was generally robust in representing trends in soil C and N in response to manure amendments.

Average total NPP, ANPP, and BNPP were modeled to be 2.20 ± 0.03 Mg C ha⁻¹, 1.84 ± 0.04 Mg C ha⁻¹, and 0.35 ± 0.01 Mg C ha⁻¹, respectively, during the 1954–2011 time block. For comparison, total NPP was similar at ~ 1.5 – 3.5 Mg C ha⁻¹ in an annual grass-dominated, degraded rangeland nearby (Ryals & Silver, 2013). Long-term forage production data from three inland rangelands in northern California indicated a historical range of 0.9–4.9 Mg dry matter ha⁻¹ (George *et al.*, 2001; Brownsey *et al.*, 2013); if forage is 40–50% C, this is equivalent to an ANPP of 0.36 Mg C ha⁻¹ at a minimum and 2.5 Mg C ha⁻¹ at maximum. The vegetation parameters used in our modeling produced NPP values comparable to the range of field measurements in California rangelands.

Average modeled CH₄ fluxes were -4.7 ± 1.2 kg CH₄-C ha⁻¹ yr⁻¹, similar to the average of a broad compilation of grassland CH₄ fluxes [-3.2 kg CH₄-C ha⁻¹ yr⁻¹, (Le Mer & Roger, 2001)]. Methane fluxes had high short-term variation but no response to long-term changes in management or vegetation. The absence of a management effect was due to the use of constant bulk density in the model, the variable to which CH₄ fluxes were most sensitive (-50% to $+100\%$ over the range of measured bulk densities). The field data showed that bulk density and soil C were strongly correlated, suggesting that as soil C accumulated in response to manure amendment, CH₄ oxidation should have increased. This would have offset some of the increase in N₂O emissions, but not enough to prevent the soil from becoming a net source. Methane oxidation would have to increase by at least a factor of 5 to completely offset N₂O emissions.

Modeled N₂O emissions were determined by N addition rate; they ranged from ~ 0.1 to 12 kg N₂O-N ha⁻¹ yr⁻¹ but most years had emissions of 0.1–0.5 kg N₂O-N ha⁻¹ yr⁻¹. These fluxes were within the range

measured in dairy rangelands in Australia and New Zealand [0.2 – 11 kg N₂O-N ha⁻¹ yr⁻¹, (Bolan *et al.*, 2004)], suggesting that our manipulation of the maximum daily nitrification rates produced reasonable N₂O fluxes.

The net ecosystem GWP was driven by changes in soil C and N₂O emissions, both of which were largely determined by manure amendments, with a smaller effect on soil C from vegetation change. The overall trend of fields shifting from net sinks to net sources was greater than the uncertainties considered here.

Long-term impacts of rangeland management and manure amendments: model results

Modeled total soil C declined due to the shift from perennial to annual grasses at an average rate of -62.4 ± 2.3 kg C ha⁻¹ yr⁻¹. While total soil C recovered and increased with the onset of intensive dairy management, the passive soil C pool continued declining. The decline was almost twice as large in the non-manured fields compared to the manured fields; thus, the model was cycling manure amendment C into the passive pool. Given the widespread conversion of grasslands from perennial to annual species in many regions across the globe (D'Antonio & Vitousek, 1992; Clary, 2012), this suggests a large and ongoing C loss. For example, non-native annual grass-dominated rangelands cover ~ 7 million ha of California alone (excluding desert and shrublands, University of California Davis, 2014) and our modeled decline in the passive soil C pool was, conservatively, -6.0 ± 0.3 kg C ha⁻¹ yr⁻¹ from 1820 to 2011. This suggests that California rangelands may have lost up to $42,000 \pm 2,100$ Mg C yr⁻¹ derived from the passive soil C pool since the shift in plant community composition. The total climate impact of this loss is 29.4 ± 1.47 Tg CO₂e, equivalent to one year's worth of emissions from 7.7 coal-burning power plants (using the US EPA GHG equivalencies calculator, <http://www.epa.gov/cleanenergy/energy-resources/calculator.html>, accessed February 11, 2014). Soil C loss has been measured at depths greater than the 20 cm modeled by DayCent and from all soil C pools in response to the change from annual to perennial grasses (Koteen *et al.*, 2011; Ryals & Silver, 2013); thus, we are likely underestimating C losses from these rangeland soils.

Manure additions had a smaller impact on ANPP and BNPP than the change in vegetation type. The transition to annual grass increased ANPP by 0.56 ± 0.03 Mg C ha⁻¹ yr⁻¹ and manure additions further increased ANPP by 0.25 ± 0.05 Mg C ha⁻¹ yr⁻¹, with a combined effect of nearly doubling rangeland ANPP. Belowground NPP declined 0.96 ± 0.02 Mg C ha⁻¹

yr⁻¹ with the introduction of annual plants and was insensitive to manure additions. There was no significant difference between the increase in nonmanured and manured fields with the onset of intensive dairying. This may reflect the relatively low degree of nitrogen limitation in the region which could have been mitigated by increased summer grazing on nonmanured fields beginning in 1921 or 1954. It may also indicate that the vegetation parameters used for annual and perennial grasses in DayCent need refinement, but that was beyond the scope of this work.

Organic matter amendments primarily affected the slow soil C pool and both the active and slow C pools reached equilibrium within the time frame of the model. Some researchers have proposed that soils have an upper limit for organic matter and C storage (Six *et al.*, 2002; Stewart *et al.*, 2007; Gulde *et al.*, 2008), but modeled soil C increased linearly with estimated C additions. These results suggested that there is an upper limit of soil C storage at a given amendment rate, but that further increases in amendment rate result in soil C increasing to a new stable state. In the model forecasts, nonmanured fields were approaching stable total C contents in 2011, whereas manured fields continued to sequester C for several more decades. Field experiments have shown that this is a common phenomenon as a balance between addition and decomposition is established (Gulde *et al.*, 2008; Thomsen & Christensen, 2010). Critically, soil C stability coincided with the ecosystem becoming a net source of C to the atmosphere. The average modeled C sequestration rate was 0.7 Mg C ha⁻¹ yr⁻¹ for the period 1954–2011; this was within the range reported by Eagle *et al.* (2012) for literature data on manured agricultural fields (0.2–5.1 Mg C ha⁻¹ yr⁻¹). However, this rate was not sustained for more than a few decades into the future under constant manure application rates. Although future soil C content could be boosted by increasing manure application rates, excess manure additions can negatively impact plant growth as well as air and water quality (Hall, 1915; Fenn *et al.*, 1998; Hooda *et al.*, 2000; Kellogg *et al.*, 2000; Ebeling *et al.*, 2002; Stokstad, 2014).

Effects on greenhouse gas emissions and global warming potential

Rangeland soils are generally a sink for atmospheric CH₄ (Smith *et al.*, 2000). The model predicted no change in CH₄ oxidation rates with the onset of manure additions. The lack of manuring effect on average modeled annual grassland CH₄ fluxes is reasonable given the competing effects of CH₄ oxidation enhancement via decreased N limitation for methanotrophs (Bodelier & Laanbroek, 2004) and inhibition of CH₄ oxidation by

increased ammonium concentrations (Le Mer & Roger, 2001), which have been observed in field studies (Del Grosso *et al.*, 2000).

Nitrous oxide emission rates responded strongly to N addition, consistent with field observations elsewhere (Bouwman, 1996; Bouwman *et al.*, 2002). Nitrous oxide emissions from manure inputs offset 75–100% of soil C sequestration between 1954 and 2011. When C sequestration decreased in the decades following 2011, continued N₂O emissions and decreasing NPP resulted in the rangelands becoming net sources. Minimizing N₂O emissions is the most effective approach to maximize the climate mitigation potential of land application of manure without compromising forage production. One possible solution is aerobically composting the manure, which, by diverting manure from anaerobic storage systems, can result in lower GHG emissions when storage, processing, and land application are considered (Brown *et al.*, 2008; DeLonge *et al.*, 2013; Owen & Silver, 2015). Other field and modeling work has shown that applications of composted organic matter to soils (rather than noncomposted manure) can significantly increase soil C and NPP with less GHG production and result in a large climate change mitigation potential (Pascual *et al.*, 1999; Ginting *et al.*, 2003; Dalal *et al.*, 2010; Aguilera *et al.*, 2013a; Ryals & Silver, 2013; Ryals *et al.*, 2015). However, more research is needed to determine optimum composting methods for manure to minimize GHG emissions during the process (Osada *et al.*, 2001; Pattey *et al.*, 2005; Maeda *et al.*, 2010; Ahn *et al.*, 2011).

Organic matter addition to rangelands offers an opportunity to help mitigate climate change but quantitative, long-term studies of both soil C and GHG fluxes are rare. Our work used field measurements and biogeochemical modeling to quantify the impact of manure additions on soil characteristics and ecosystem processes. Both approaches showed that long-term land application of manure increased soil C and N in the top 20 cm by 19.0 ± 7.3 Mg C ha⁻¹ and 1.94 ± 0.63 Mg N ha⁻¹, respectively, compared to nonmanured fields. The greatest change in ANPP occurred as a result of the change in vegetation type from perennial to annual grasses (~0.56 Mg C ha⁻¹ yr⁻¹), and the additional N in manured fields increased ANPP by ~0.25 Mg C ha⁻¹ yr⁻¹. These N additions also increased N₂O emissions by ~1.7 kg N₂O-N ha⁻¹ yr⁻¹ compared to nonmanured fields, which offset at least 75% of the climate change mitigation from increased soil C and NPP. Soil C and N pool sizes were predicted to stabilize in the coming decades, at which point GHG emissions would exceed C sequestration and the ecosystems would become large net sources. Thus, optimizing organic matter amendment quantity, quality, and timing is

likely critical in maximizing its climate change mitigation potential of rangelands.

Acknowledgements

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

- Table S1.** Field management history and characteristics.
Table S2. Parameterization of manure amendments and direct deposition in DayCent OMAD file.
Table S3. Parameterization of manure amendments and direct deposition in DayCent FERT file.
Table S4. Soil characteristics for each field (mean ± standard error, $n = 5$). *Avg/sum* rows are the depth-weighted averages or sums (in the case of N and C contents) to 50 cm for each field.