Title

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Journal

Ecological Applications, 27(2)

ISSN

1051-0761

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Publication Date

2017-03-01

DOI

10.1002/eap.1465

Peer reviewed

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First published: 11 November 2016
https://doi.org/10.1002/eap.1465
Cited by: 2
UC-eLinks
Corresponding Editor: David S. Schimel.

Abstract

Livestock agriculture is a major source of anthropogenic greenhouse gas (GHG) emissions, with a substantial proportion of emissions derived from manure management. Accurate estimates of emissions related to management practices and climate are needed for identifying the best approaches to minimize, and potentially mitigate, GHG emissions. Current emissions models such as those of the IPCC, however, are based on emissions factors that have not been broadly tested against field-scale measurements, due to a lack of data. We used a diverse set of measurements over 22 months across a range of substrate conditions on a working dairy to determine patterns and controls on soil-based GHG fluxes. Although dairy soils and substrates differed by management unit, GHG fluxes were poorly predicted by these or climate variables. The manure pile had the greatest GHG emissions, and though temperature increased and O_2 concentration decreased following mixing, we detected almost no change in GHG fluxes due to mixing. Corral fluxes were characterized by hotspots and hot moments driven by patterns in deposition. Annual scraping kept the soil and accumulated manure pack thin, producing drier conditions, particularly in the warm dry season. Summed over area, corral fluxes had the greatest non-CO_2 global warming potential. The field had net CH_4 consumption, but CH_4 uptake was insufficient to offset N_2O emissions on an area basis. All sites emitted N_2O with a similar or greater climate impact than CH_4. Our results highlight the importance of N_2O emissions, a less commonly measured GHG, from manure management and present potential opportunities for GHG emissions reductions.

Introduction

Greenhouse gas (GHG) emissions from livestock agriculture are large and increasing (EPA 2012, 2015). In 2005, methane (CH_4) and nitrous oxide (N_2O) emitted from livestock manure management alone accounted for nearly 7% of the total non-carbon dioxide (CO_2) GHG emissions from agriculture globally and are projected to increase by 15–20% by 2030 (EPA 2012). Estimates of current and projected GHG emissions are critical for directing climate change mitigation efforts and policy. Recent analyses suggested that the lack of field data may lead to significant underestimates of GHG emissions both regionally and globally (Owen and

Silver 2015). The greatest limitation to improving large-scale model estimates of GHG fluxes has been the lack of field-scale GHG flux measurements with accompanying climate, soil, and manure biogeochemical data collected over longer time scales (>1 yr) and frequencies capable of capturing short- and long-term changes.

Controls on the production of CO_2, CH_4, and N_2O from manure management are the same as from other forms of organic matter decomposition and nutrient mineralization, which include environmental conditions such as temperature, moisture content, and oxygen (O_2) availability, as well as substrate chemical and physical properties (Conrad 1996, Davidson et al. 2000, Ryan and Law 2005, De Klein et al. 2006, Dong et al. 2006). These factors are influenced by climate and management activities, such as pile mixing (Eghball et al. 1997, Hao et al. 2001, Ahn et al. 2011), scraping of corrals (Ellis et al. 2001, Gao et al. 2011), amendment applications (Ginting et al. 2003, Simonetti et al. 2012, Ryals et al. 2014, Owen et al. 2015), and tillage (Venterea et al. 2005, Blair et al. 2006, van Kessel et al. 2013). Carbon dioxide emissions from manure are considered climate neutral by the IPCC under the assumption that they are part of a tightly coupled photosynthesis–respiration cycle (Dong et al. 2006); however, some have argued that these CO_2 emissions are not benign, due to the decline in soil C and on-going deforestation associated with livestock agriculture (Goodland 2014). Manure stored in solid or liquid form is typically a net source of CH_4 due to anaerobic conditions (Dong et al. 2006), whereas agricultural fields to which manure is applied may be net sinks of CH_4 (Lasco et al. 2006, Conrad 2007). Fewer studies have measured N_2O fluxes (Owen and Silver 2015). Manure storage and land application may be a source of N_2O because of relatively high N availability and heterogeneous composition resulting in aerobic and anaerobic zones where nitrification and denitrification can both occur (Conrad 1996, De Klein et al. 2006).

Accurate estimates of GHG fluxes from manure management and land application is important for informing research and policy-making, but is challenging due to the complex interactions between factors controlling decomposition, the temporal and spatial heterogeneity of manure management systems, and the limited data available for comparison. This is particularly true for soil-based manure management, which is not as well studied as liquid manure storage (Owen and Silver 2015). The few available field measurements suggest that manure pile and corral N_2O emissions may be underestimated by an order of magnitude (Owen and Silver 2015). Owen and Silver (2015) also found that field measurements were lacking in general, but were especially poor for several key dairying regions including California, the largest dairy producer in the United States.

We measured GHG fluxes from common soil-based manure management strategies (dry lot corral, solid manure pile, field amendments), which are among the least characterized of manure management approaches, together with manure and soil properties on a pasture-based dairy in California for nearly two years. We coupled GHG flux measurements with measurements of substrate temperature and moisture, total C and N content, pH, bulk density, and mineral N pools and fluxes, and with observations of management practices as potential drivers of the observed patterns. For the manure pile, we also explored the role of O_2concentrations as a potential predictor of GHG emissions.

Methods

Study sites

The study sites were located on an organic, pasture-based dairy located in Marin County, California, USA. Local daily climate data (precipitation accumulation; minimum, average, and maximum temperature) were available for three nearby (<10 km) weather stations (*available online*).[1] One station (KCAMARSH3) provided the bulk of the weather data and the other two (KCAINVER3 and MPREC1) were used to evaluate the consistency between stations and fill any data gaps. Precipitation and air temperature followed typical coastal Mediterranean climate patterns with most rainfall falling in winter and early spring (Fig. 1) and temperatures peaking in late August to early September (Fig. 1). The second year of the study (2014) was drier and warmer than the first (2013).

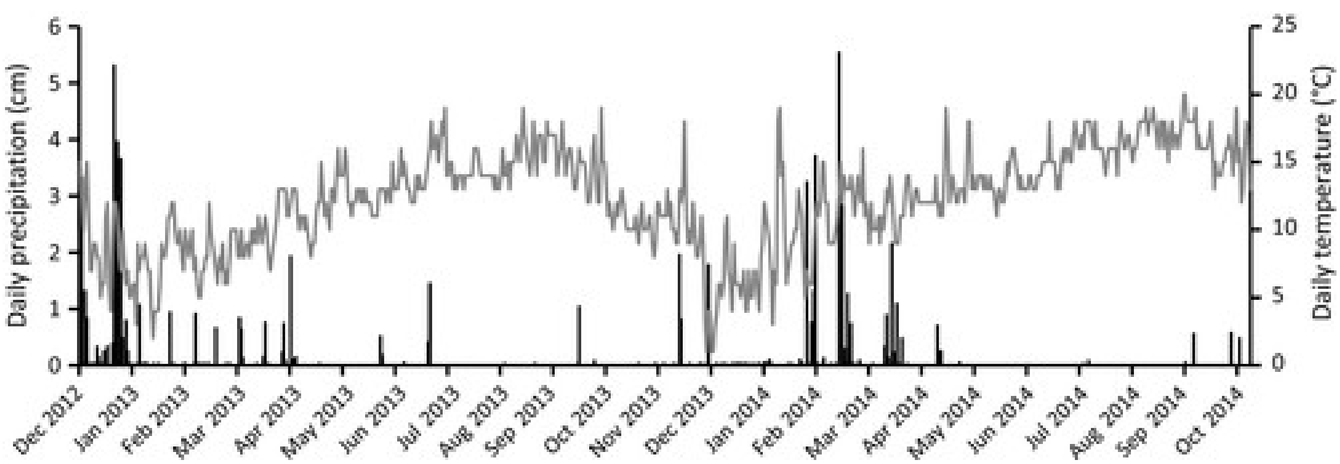

Figure 1

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Daily precipitation (bars) and mean temperature (line) during the sampling period.

Caption

The dairy had approximately 200–250 cows, evenly split between milk cows and heifers, which typically grazed on pasture for 9 months each year. Manure management practices were typical for dairies in the region. Manure accumulated in corrals that were used to temporarily hold milk cows prior to milking and to house perinatal cows (with a rotating population of 10–15 cows). Manure was collected from the corrals once or twice per year and stored in a pile. Bedding and manure from a freestall barn (used for several months in the rainy season) were also added to the pile. Pile solids were spread on pastures once a year or every other year, depending on the nutrient status of the soil. Liquid manure slurry derived from the milk parlor was applied to the silage field three times during the study, with an application rate of approximately 6 g mineral $N \cdot m^{-2}$ per application. The field was seeded with a mix of rye grasses annually.

Our sampling focused on the soil-based manure management practices likely to be the greatest GHG sources: the corral holding perinatal cows, the manure pile, and the silage field. Measurements were made from January 2013 through October 2014. Corral samples were collected from a 70 × 40 m area that contained the feed trough. The corral was scraped prior to the February 2013 and September 2014 measurements, and the trough was moved prior to the March 2013 and February 2014 measurements. The pile was 30 m long, 12 m wide, and 1–2 m tall at the start of the study; manure was added in 2–3 m wide rows to the north end of the pile in February 2013 and to the south end in November 2013. The pile was turned in May 2014 and was otherwise minimally disturbed except for weeds being scraped off using a bulldozer in June and September of 2013. The silage field was 1.67 ha and uniformly managed; sampling occurred in a 50 × 60 m plot. Silage was harvested in April, May, and July of 2013 and in April and July of 2014. The field was briefly grazed by heifers following the last harvest (<2 d in August 2013 and July 2014). Slurry was applied in August 2013, November 2013, and September 2014, and the field was aerated and seeded the following month.

Manure, soil, and plant sampling

To characterize the substrate, we sampled fresh feces in the corral, the corral surface, the manure pile, and the field soil. Five fresh feces samples (with freshness determined by the absence of a surface crust or observation of deposition) and five grab samples of the corral soil were collected from random locations monthly throughout 2013 and then once again in June 2014. At the same sampling interval, soil temperature, moisture, and depth were measured in 30 random locations in the corral using 10 cm long soil temperature probes, 20 cm long soil moisture probes, and a ruler, respectively. The corral soil was thin (generally <10 cm) and composed almost entirely of accumulated manure with the mineral soil thinned by long-term scraping, thus, the probes were inserted at an angle to completely cover the sensors.

The manure pile was sampled on the same schedule as the corral. Samples were collected volumetrically at three locations along the top of pile (to minimize disturbance to trace gas sample chambers) using metal corers at depth intervals of 0–10, 10–30, 30–50, and 50–100 cm (6.5 cm diameter for <30 cm, 5.5 cm diameter for 30–100 cm).

Silage field soil sampling and vegetation measurements occurred in January, April, August, and December of 2013, and June of 2014. Soil was volumetrically sampled from 0–5, 5–10, 10–20, 20–30, and 30–50 cm with metal corers (using a 6.5 cm diameter corer from 0–20 cm and a 5.5 cm diameter corer from 20 to 50 cm) at five locations along a 40 m long transect. The transect was located more than 30 m from the nearest fence line to avoid edge effects. The sampling transect intersected but did not disturb the gas sampling locations described below. At the soil sampling locations, biomass samples were clipped from three 10 cm diameter circles, collected into pre-dried and weighed paper bags, dried at 65°C, and weighed to determine aboveground biomass. Soil temperature, soil moisture, and maximum plant height were measured at 30 random locations within 20 m of the chambers and soil sampling transect.

Laboratory analyses

Subsamples of the fresh feces, corral surface, manure pile, and field soil were oven dried at 105°C to calculate water content and, in the case of the manure pile and field soil, bulk density. The remaining samples were air dried. Manure pile and field soil samples were passed through a 2 mm sieve and the separated rocks were weighed to calculate rock concentration. Subsamples of the air-dried feces, corral surface, fine manure pile, and fine soil were ground to a powder with a ball grinder (SPEX Sample Prep Mixer Mill 8000D, Metuchen, New Jersey, USA). These samples were analyzed for total C and N concentration on an elemental analyzer (Carlo Erba Elantech, Lakewood, New Jersey, USA) using atropine as a standard run every 10 samples. Volatile solids (VS) were determined for the fresh manure, corral, and pile samples by loss on ignition; samples were oven dried at 105°C, weighed, then combusted at 550°C for 2 h.

Soil pH was measured in a 1:1 volumetric slurry of sample (fresh feces; air-dried corral surface, pile material, and field soil) and deionized water using a pH electrode (Denver Instruments, Bohemia, New York, USA). Nitrate (NO_3^-) and ammonium (NH_4^+) were measured in fresh manure, corral soil, and field soil (samples from the upper 20 cm) using a KCl extraction (Mulvaney 1996). Pile samples were not analyzed for mineral N content. Within 24 h of collection, samples (5 g wet mass manure and corral soil, 15 g wet mass field soil) were shaken for 1 h in 75 mL of 2 mol/L KCl solution and then filtered through pre-rinsed filters. The samples were frozen until colorimetric analysis of NO_3^- and NH_4^+ using an autoanalyzer (Lachat

Quik Chem flow injection analyzer, Lachat Instruments, Milwaukee, Wisconsin, USA). Additional samples were incubated at room temperature in loosely covered, darkened jars for 7 d, after which NO_3^- and NH_4^+ were extracted with KCl as above. The difference in NO_3^- concentration between the two time points was used to calculate potential net nitrification, and the difference in the sum of NO_3^- and NH_4^+ concentrations between time points was used to calculate potential net mineralization, with soil mass corrected for moisture content (Hart et al. 1994).

Trace gas sampling

Carbon dioxide, CH_4, and N_2O fluxes from the corral, pile, and field were measured using static flux chambers, with 12 chambers deployed at each site on each sampling date. The chambers consisted of 30 cm diameter PVC rings that were topped with opaque, cylindrical, vented lids (approximately 4500 cm^3 in volume). Samples were collected monthly from January 2013 through October 2014. Previous work at nearby sites using the same methods measured the highest emissions during the fall wet-up events (Chou et al. 2008, Ryals and Silver 2013) so we also sampled in more intensive campaigns following the first rains of autumn and the first rain after a winter dry spell. Our intensive sampling periods included five dates in the week following a 3-cm rainfall November 2013, three dates following a 3-cm rainfall February 2014 (after a dry December and January), and three dates in the week following a 0.5-cm rainfall September 2014. Sampling usually occurred between 10:00 and 16:00, when fluxes may approximate the daily average (Parkin and Venterea 2010). Many approaches have been used to estimate greenhouse gas emissions on dairies including static flux chambers, dynamic chambers, tower-based measurements using eddy covariance or mass balance approaches in conjunction with lasers, Fourier transform infrared spectroscopy (FTIR), and isotope tracers, each with pros and cons (reviewed by Owen and Silver 2015). We chose to use static chambers as the most comparable across the treatments measured here. Static chambers measure a relatively small footprint, but can be highly replicated and, when vented and used for relatively short flux intervals (40 min in this study), result in minimal changes to the substrate and headspace environment (Davidson et al. 2002). Static chambers are particularly well suited to manured soils, where slight vibrations from dynamic chambers can stimulate gas release and result in an overestimate of fluxes. Other approaches may be more suitable for estimating fluxes from whole dairies (e.g. tower-based approaches), but this was not the goal of the current study.

In the corral, the chambers were arranged along two transects of six chambers each located on either side of the feed trough. For the pile, six chambers were deployed along each side and were evenly distributed between top, shoulder, and slope positions in order to best approximate an area-average of pile emissions without capturing only the highest emissions from the top (Sommer et al. 2004, Andersen et al. 2010). In the silage field, the chambers were arranged in two 20 m diameter circles of six chambers each located in approximately the same places each time. Grass within the chamber base was twisted and bunched together to fit inside the chamber. At all sites, soil temperature and moisture were measured next to each chamber.

Gas samples (30 mL) were collected from each chamber at 0, 5, 15, 25, and 40 min after the lid was put in place and stored in pre-evacuated gas vials until analysis (within 72 h) for CO_2, CH_4, and N_2O on a gas chromatograph (Shimadzu GC-14A, Pleasanton, California, USA) equipped with flame ionization detector and an electron capture detector. Samples were analyzed once the coefficient of variation of the peak heights for calibration standards were less than 2%, and standards were run every 10 samples. If the coefficient of variation between standards exceeded 2%, the samples were reanalyzed. Detection limits for the GC were estimated at 0.09 ppm CH_4, 0.49 ppb for N_2O, and 0.09 ppm for CO_2. Fluxes were calculated using an iterative exponential curve-fitting approach (Matthias et al. 1978); time series that were not statistically significant were reported as zero fluxes. The majority of statistically significant fitted curves had $R^2 > 0.99$. To compare across years, fluxes for 1 January through 20 October for 2013 and 2014 were summed using linear interpolations between the average fluxes of each sampling day. We also estimated a total annual flux for 2013 that allowed us to compare with literature values. Total CH_4 fluxes were calculated with and without outliers, with outliers defined as fluxes greater than 400 ng $C\cdot cm^{-2}\cdot h^{-1}$ or less than −100 ng $C\cdot cm^{-2}\cdot h^{-1}$. These criteria excluded 12 high CH_4 flux outliers and one low datum out of 312 measurements in the corral, and 31 high outliers out of 312 measurements in the pile; there were no outliers in the field. Fluxes of N_2O and CH_4 were converted to CO_2 equivalents using 100-year global warming potentials (GWP) of 298 and 34, respectively (Myhre et al. 2013).

Manure pile O_2 and temperature profile measurements

To better understand the relationship between temperature, O_2 availability, and GHG fluxes from the pile, three pre-calibrated (at room temperature, 100% relative humidity) continuous O_2 sensors equipped with thermistors (SO-111 series; Apogee Instruments, Logan, Utah, USA) were installed in the manure pile at depths of 30, 55, and 120 cm. Three holes within 50 cm of each other were excavated with an auger to the appropriate depth, the sensor was lowered in, and the hole was backfilled with the original material. The sensors were connected to a CR-1000 data logger (Campbell Scientific, Logan, Utah, USA) powered with a solar panel. Measurements were made every 30 s and saved as 5-min averages. Data were recorded from 13 April 2014 to 17 April 2014 to get pre-turning baseline characteristics. The pile was mixed with a bulldozer on 14

May, and the sensors were re-installed immediately after turning in approximately the same locations as before and allowed to equilibrate for 6 d. The sensors remained in place until 26 August 2014. Oxygen sensor output was converted from millivolts (mV) to O_2 concentration by multiplying the field mV value by 20.95% and dividing by the mV reading during calibration.

Statistical analyses

Statistical analyses were performed using JMPPro11 (SAS Institute, Cary, North Carolina, USA) and R (https://www.r-project.org/). Effects on GHG fluxes were tested using a full factorial, standard least squares model that included soil temperature, soil moisture, pH, mineral N concentration, soil C and N concentrations, bulk density (for pile and field fluxes), sampling month, and time of sampling. Characteristics of surface samples (fresh manure, corral, pile surface, and soil surface) and samples by depths within the pile and soil were analyzed with a one-way ANOVA for repeated measures and a post-hoc Tukey test with Bonferroni correction. Emissions of CO_2 were log-transformed (with 8 out of 1182 measurements excluded for being less than 0) and analyzed as above. Methane and N_2O fluxes were not normally distributed and could not be transformed to be so; thus, we analyzed them using a Friedman's test with a nonparametric post-hoc test. Statistical significance was determined as $P < 0.05$. Data in the text are presented as means and standard errors unless otherwise specified. Our study design did not include randomized replicate plots because there were no true replicate treatments, as is typical of working dairies. We used pseudo-replication to explore potential indications of treatment differences and urge appropriate caution in the extrapolation of this portion of our results.

Results

Manure and soil composition

Manure composition changed significantly as it decomposed and underwent different management practices. Carbon and N concentrations of fresh manure were 1.5–6 times higher than the corral, pile, and field (Table 1). The corral surface was mostly manure with some rock fragments incorporated from the underlying soil and bedrock. Depth to bedrock averaged 4.9 ± 0.5 cm, with the minimum average depth in February 2013 (2.6 ± 0.3 cm) after the corral was scraped. Manure accumulated most rapidly adjacent to the feed trough, often to more than 10 cm thick, and this area was also persistently moist from urine deposition. Except for the areas where soil was thinnest, a hard compacted layer composed of manure and mineral soil underlay uncompacted material. Total C and N concentrations in the corral surface varied seasonally, with the highest concentrations (nearly equal to fresh manure) occurring in September and October and the lowest in early spring (Fig. 2).

Table 1. Fresh manure, corral soil, manure pile, and field soil characteristics (mean ± standard error of all measurements)

Location	C (% DM)	N (% DM)	C:N	VS (% DM)	pH
Fresh manure	$37.85^{A} \pm 0.95$	$2.17^{A} \pm 0.10$	$17.61^{A} \pm 0.82$	$80.57^{A} \pm 1.68$	$6.70^{A} \pm 0.16$
Corral	$18.58^{B} \pm 1.98$	$1.51^{B} \pm 0.12$	$11.96^{B} \pm 0.36$	$36.62^{B} \pm 3.55$	$7.92^{B} \pm 0.16$
Pile					
0–10 cm	$12.52^{Ca} \pm 0.45$	$1.22^{Ca} \pm 0.04$	$10.15^{Ca} \pm 0.16$	$26.27^{Ca} \pm 1.86$	$7.53^{Ba} \pm 0.08$
10–30 cm	$11.93^{a} \pm 0.77$	$1.19^{a} \pm 0.07$	$9.91^{a} \pm 0.17$	$23.01^{a} \pm 1.30$	$7.77^{ab} \pm 0.10$
30–50 cm	$13.44^{a} \pm 0.70$	$1.26^{a} \pm 0.06$	$10.58^{a} \pm 0.15$	$23.04^{a} \pm 1.43$	$8.12^{bc} \pm 0.14$
50–100 cm	$12.10^{a} \pm 0.69$	$1.11^{a} \pm 0.05$	$10.77^{a} \pm 0.23$	$28.41^{a} \pm 1.95$	$8.32^{c} \pm 0.14$
Field soil					
0–5 cm	$6.29^{Db} \pm 0.19$	$0.57^{Db} \pm 0.02$	$10.96^{Db} \pm 0.10$	NM	$6.08^{Cd} \pm 0.07$
5–10 cm	$4.88^{c} \pm 0.12$	$0.45^{c} \pm 0.01$	$10.86^{b} \pm 0.07$	NM	$5.76^{d} \pm 0.08$
10–20 cm	$2.76^{d} \pm 0.05$	$0.25^{d} \pm 0.01$	$11.06^{b} \pm 0.04$	NM	$5.89^{d} \pm 0.12$

Location	C (% DM)	N (% DM)	C:N	VS (% DM)	pH
20–30 cm	$2.15^{e} \pm 0.05$	$0.19^{e} \pm 0.01$	$11.28^{c} \pm 0.06$	NM	$5.97^{d} \pm 0.07$
30–50 cm	$1.60^{f} \pm 0.09$	$0.14^{f} \pm 0.01$	$11.27^{c} \pm 0.13$	NM	$5.95^{d} \pm 0.09$

Notes:

- VS, volatile solid; DM, dry matter; NM, not measured. Different capital letters indicate significant differences ($P < 0.05$) between surface samples. Different lowercase letters indicate significant differences between depths.

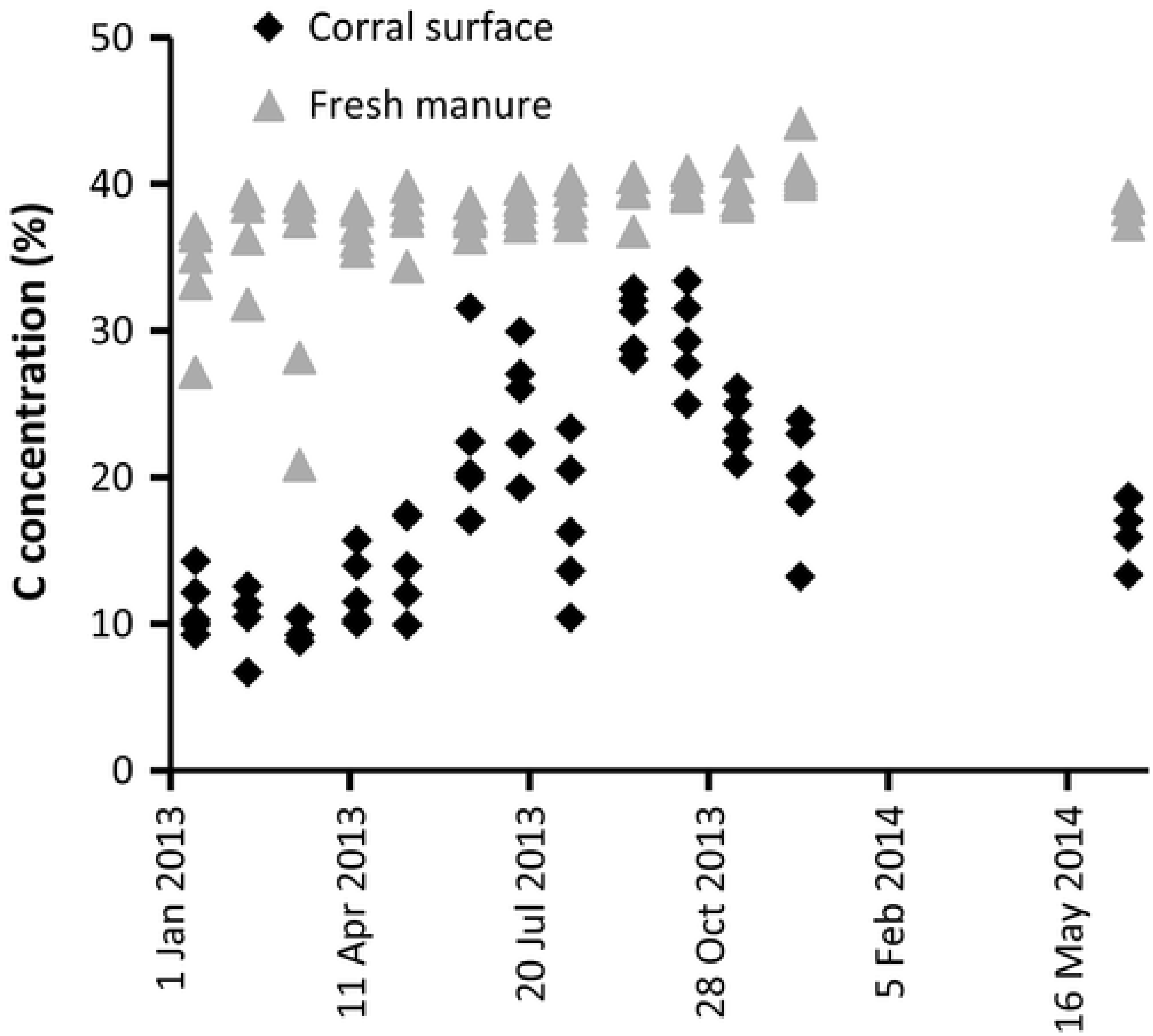

Figure 2

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Carbon concentration in the corral surface and fresh manure over the sampling period.

Caption

The pile had lower C and N concentrations than fresh manure and corral surface material (Table 1). Pile C and N concentrations were spatially heterogeneous, varying between the three replicate locations and with depth. For example, C and N concentrations increased with depth at one end of the pile and decreased with depth at the other. They did not vary significantly over time. The field soil had the lowest C and N concentrations (Table 1) and these also did not vary significantly over time. Soil C and N concentrations decreased with depth while bulk density increased, as is typical for the region (Owen et al. 2015).

Volatile solid concentrations also decreased from fresh manure to corral to pile (Table 1). The corral had less than half the VS measured in the fresh manure. Corral VS concentration was correlated with C concentration ($r^2 = 0.64$, $P < 0.0001$) and followed the same seasonal trend. Pile VS concentration was also correlated with C concentrations ($r^2 = 0.34$, $P < 0.0001$) but had no seasonal trend.

Inorganic N decreased from fresh manure to corral to field as a result of decreasing NH_4^+concentrations (Table 2). Fresh manure had the highest NH_4^+ concentrations (504 ± 107 μg/g) and the corral had the highest NO_3^- concentrations (93 ± 33 μg/g). The potential net nitrification rate in fresh manure was strongly negatively correlated with NO_3^- concentration ($r^2 = 0.99$, $P < 0.0001$) and potential net mineralization rate was strongly negatively correlated with NH_4^+ concentration ($r^2 = 0.84$, $P < 0.0001$). Corral NH_4^+ concentrations were relatively constant over time whereas NO_3^- concentrations varied, increasing slightly from January to December 2013 ($r^2 = 0.21$, $P = 0.0004$). The potential net nitrification rate in corral soil was negatively correlated with NO_3^- concentration ($r^2 = 0.71$, $P < 0.0001$) and potential net mineralization was negatively correlated with both NH_4^+ concentration ($r^2 = 0.30$, $P < 0.0001$) and NO_3^- concentration ($r^2 = 0.46$, $P < 0.0001$). In the field, NH_4^+ decreased with depth but did not vary significantly across seasons. Potential net nitrification and mineralization rates were positive, indicating increasing inorganic N concentrations during the incubations.

Table 2. Inorganic N, potential net nitrification, and potential net mineralization in fresh manure, corral soil, and field soil

Location	**NH_4^+(μg N/g soil)**	**NO_3^-(μg N/g soil)**	**Potential net nitrification (μg N·g soil^{-1}·d^{-1})**	**Potential net mineralization (μg N·g soil^{-1}·d^{-1})**
Fresh manure	$506^A \pm 107$	$38.2^A \pm 13.6$	$-4.15^A \pm 2.22$	$-54.2^A \pm 15.5$
Corral	$112^B \pm 32.1$	$93.0^B \pm 32.6$	$-4.98^A \pm 4.71$	$-12.2^B \pm 7.54$
Field soil				

Location	NH_4^+(μg N/g soil)	NO_3^-(μg N/g soil)	Potential net nitrification (μg N·g soil^{-1}·d^{-1})	Potential net mineralization (μg N·g soil^{-1}·d^{-1})
0–5 cm	$5.20^{Ba} \pm 1.00$	$30.0^{Aab} \pm 6.33$	$3.83^{Aa} \pm 3.29$	$3.85^{Ba} \pm 3.30$
5–10 cm	$3.46^{a} \pm 0.99$	41.1^{b}[a] ± 5.5	$0.38^{a} \pm 0.25$	$0.47^{a} \pm 0.22$
10–20 cm	$1.35^{b} \pm 0.34$	$22.2^{a} \pm 1.9$	$0.55^{a} \pm 0.25$	$0.66^{a} \pm 0.19$

- *Notes:* Different capital letters indicate significant differences ($P < 0.05$) between surface samples. Different lowercase letters indicate significant differences between depths.
- a Significantly different from 0 to 5 cm for $P < 0.09$.

Manure pH changed significantly as it passed through the management practices (Table 1). Fresh manure had a near neutral pH; manure pH became more basic as it accumulated on the corral surface. This was consistent with observations in the lab where air-dried manure had pH values 1.5–2 units higher than when it was measured immediately after collection. Pile samples also had relatively high pH that increased with depth, ranging from 7.53 ± 0.08 from 0–10 cm to 8.32 ± 0.14 from 50–100 cm (Table 1). Field pH did not vary with depth but did vary seasonally in the upper 20 cm where pH was lower in January and December and higher in August and June. This seasonal variation in pH was correlated with soil moisture. Soil pH decreased with moisture at 0–5 cm ($r^2 = 0.41$, $P < 0.0005$); however, pH increased with moisture at 20–30 cm ($r^2 = 0.30$, $P < 0.01$) and 30–50 cm ($r^2 = 0.37$, $P < 0.004$).

Substrate moisture and temperature

The seasonal cycles of precipitation and temperature were reflected in the moisture content and temperature of the corral, pile, and field (using moisture measured at the chambers, Fig. 3). Soil moisture and temperature showed strong seasonality in all sites and were significantly negatively correlated (Fig. 4). Pile surface moisture and temperature were the highest of the three sites

(Fig. 3). Moisture varied seasonally following precipitation patterns (Fig. 3a) and generally increased with depth. Pile temperatures also varied seasonally, following and usually exceeding maximum air temperatures (Fig. 3b). Corral moisture content and temperature also varied by season and year (Fig. 3). Corral temperatures were intermediate between pile and field temperatures. They had large seasonal variation, typically exceeding maximum air temperatures in the summer and being closer to mean daily air temperature in winter.

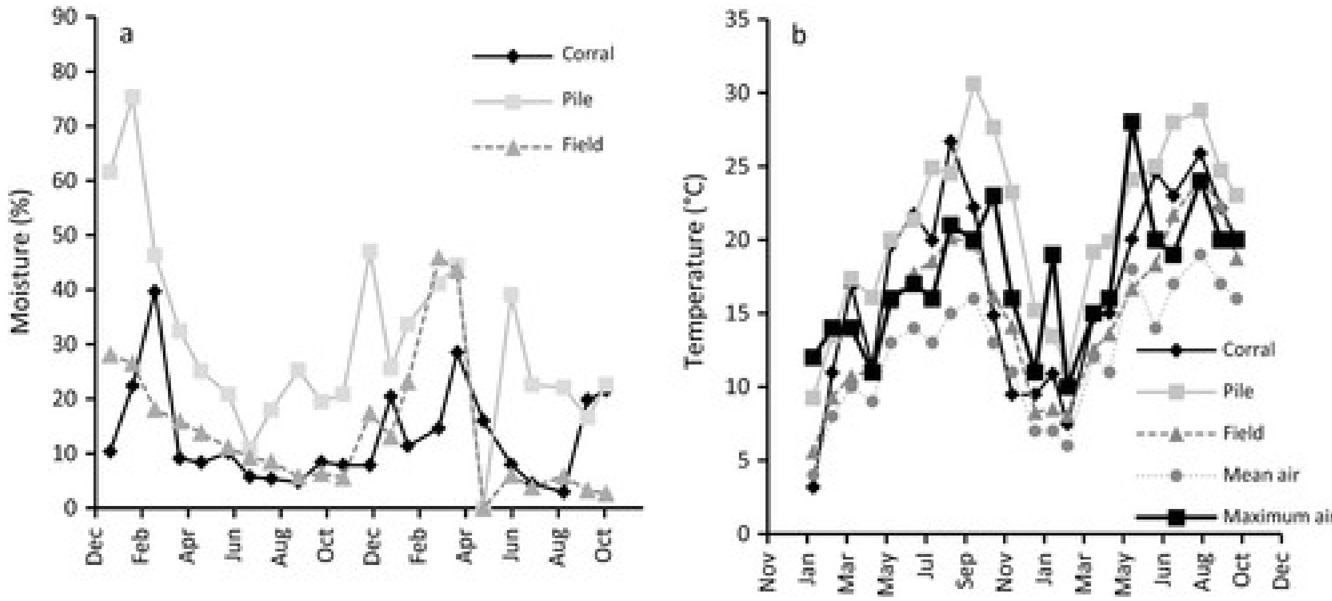

Figure 3

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(a) Soil moisture in the top 20 cm of the corral, pile, and field over the sampling period. (b) Mean and maximum air temperatures compared to corral, pile, and field surface (upper 10 cm) temperatures.

Caption

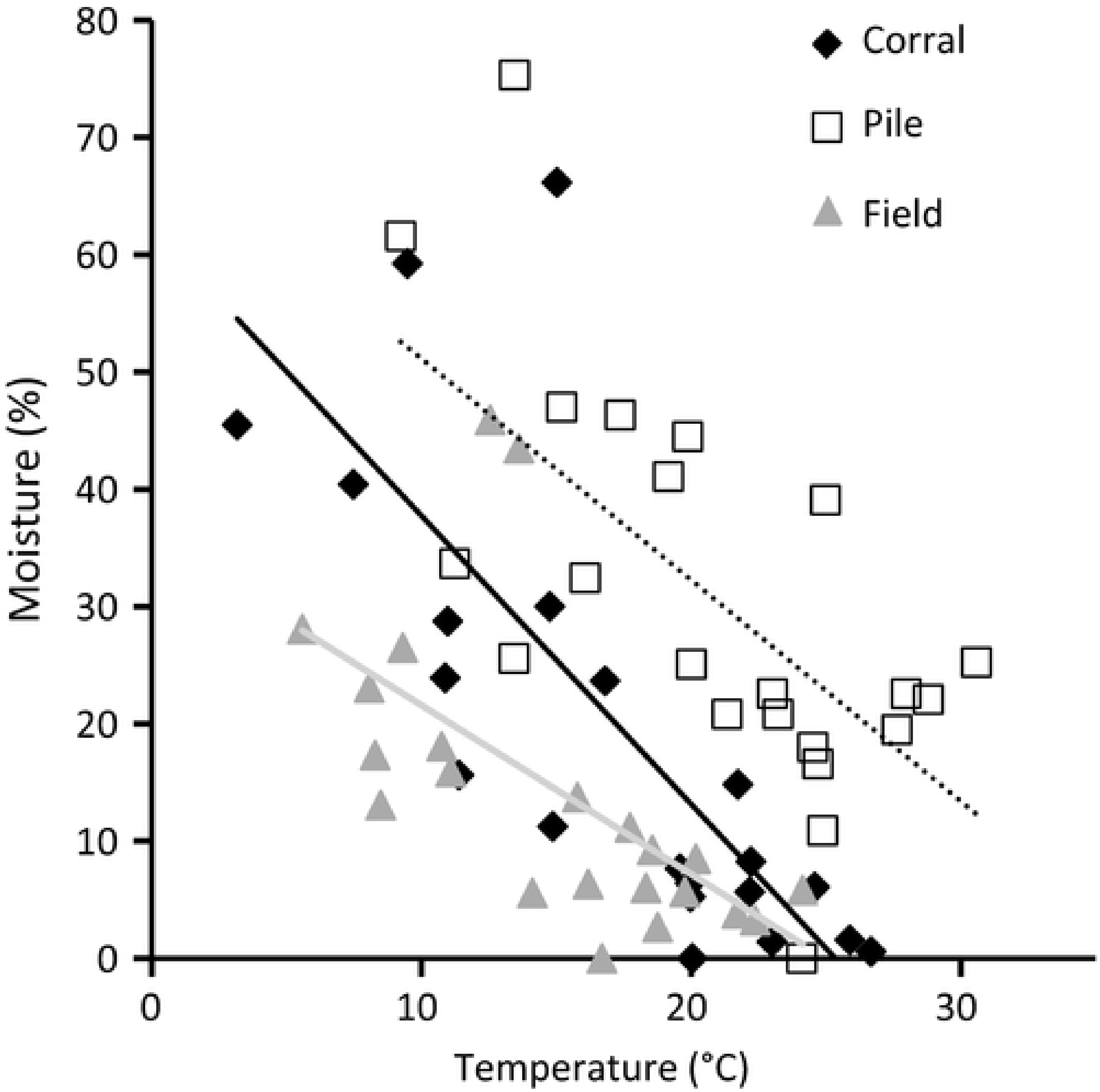

Figure 4
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Relationship between mean moisture and temperature at the chamber locations for the corral, pile, and field. Trendlines: solid black, corral, $r^2 = 0.59$, $P < 0.0001$; dotted, pile, $r^2 = 0.43$, $P < 0.0009$; gray, field, $r^2 = 0.37$, $P < 0.004$.
Caption

In the field, soil moisture varied over the course of the year with the greatest variation in the surface (more than 30%), but moisture was essentially constant from 10 to 50 cm in any sampling month. Soil texture was not measured for this field, but nearby fields on the same dairy had 9–18% clay in the top 10 cm (Owen et al. 2015; “dairy C” data) and all showed evidence of shrink-swell clays, with small cracks forming during the dry season. Cracks appeared to be bigger and deeper during the summer of 2014. Field soil temperatures closely tracked air temperature (Fig. 3b) and were usually between the mean and maximum air temperatures.

Greenhouse gas fluxes

Temporally weighted mean hourly CO_2 emissions in 2013 ranged from 11.4 ± 2.8 μg C·cm^{-2}·h^{-1}in the corral to 32.6 ± 14.1 μg C·cm^{-2}·h^{-1} in the pile (Fig. 5, Table 3). Average pile CO_2 fluxes ranged from 4.2 ± 3.3 μg C·cm^{-2}·h^{-1} (April 2013) to 102.7 ± 51.3 μg C·cm^{-2}·h^{-1} (November rain 2013) and had no seasonal trend. Average CO_2 fluxes from the corral ranged from 2.2 ± 0.9 μg C·cm^{-2}·h^{-1} (November 2013) to 39.7 ± 7.0 μg C·cm^{-2}·h^{-1} (March 2013), also with no seasonal trend. In the field, average CO_2 fluxes ranged from 1.3 ± 0.1 μg C·cm^{-2}·h^{-1} (November 2013) to 61.3 ± 3.9 μg C·cm^{-2}·h^{-1} (April 2013), with peak fluxes corresponding to peak plant biomass (described further in Management effects on GHG fluxes). Total field CO_2 fluxes from January through October for 2013 were significantly greater than the same period in 2014 ($P < 0.0001$, Table 3).

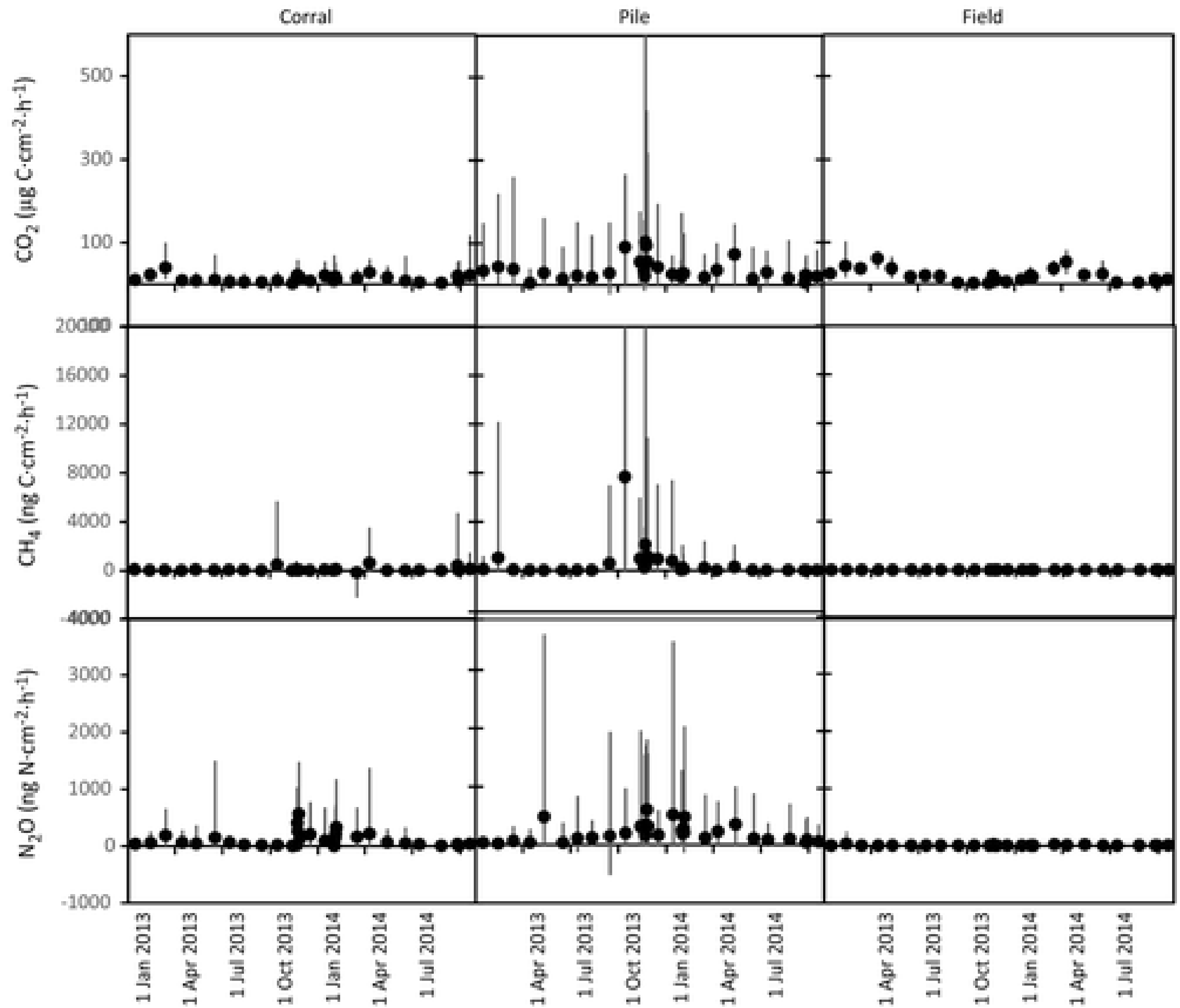

Figure 5

[**Open in figure viewerPowerPoint**]

Greenhouse gas fluxes (mean [circles] and range [bars]) for the corral, pile, and field over the sampling period. Note that the maximum CO_2 flux from the pile in November is 620 µg C·cm^{-2}·h^{-1} and cut off in the plot, as are maximum CH_4 fluxes from the pile in October and November of 2013 (85 000 and 22 000 ng C·cm^{-2}·h^{-1}, respectively).

Caption

Table 3. Interpolated mean measured greenhouse gas fluxes from each site for 2013 and for January through October of 2013 and 2014

Gas and time span	Corral	Pile	Field
CO_2 (µg C·cm^{-2}·h^{-1})	A	B	A (C for $P < 0.06$)

Gas and time span	Corral	Pile	Field
2013	11.35 ± 2.78	32.61 ± 14.09	22.38 ± 2.31
January–October 2013	12.19 ± 3.82	27.75 ± 12.89	28.79 ± 11.36
January–October 2014	13.94 ± 3.71	29.70 ± 9.55	18.92 ± 2.16
CH_4 (ng C·cm^{-2}·h^{-1})	A	A	B
2013	63.83 ± 57.58	875.56 ± 787.19	−0.09 ± 0.95
January–October 2013	54.85 ± 47.73	585.97 ± 548.32	−0.07 ± 0.98
January–October 2014	89.72 ± 93.24	164.25 ± 123.45	−1.17 ± 0.37
CH_4 no outliers[a] (ng C·cm^{-2}·h^{-1})			
2013	12.74 ± 9.35	12.37 ± 7.86	−0.09 ± 0.95
January–October 2013	21.13 ± 11.90	9.03 ± 5.90	−0.07 ± 0.98
January–October 2014	15.33 ± 10.17	13.98 ± 9.52	−1.17 ± 0.37

Gas and time span	Corral	Pile	Field
N_2O (ng N·cm^{-2}·h^{-1})	A	B	C
2013	68.33 ± 32.00	138.08 ± 80.06	3.71 ± 1.98
January–October 2013	62.34 ± 32.46	117.61 ± 75.99	25.36 ± 28.09
January–October 2014	99.13 ± 45.53	202.16 ± 88.14	6.62 ± 2.26

- *Note:* Significant differences among management types ($P < 0.05$) are indicated by different uppercase letters in the row for each gas.
- a "No outliers" values were calculated by excluding fluxes greater than 400 ng C·cm^{-2}·h^{-1} and less than −100 ng C·cm^{-2}·h^{-1}.

Methane fluxes were characterized by hotspots and hot moments. Temporally weighted average CH_4 fluxes were greatest from the pile, about an order of magnitude smaller in the corral, and negative in the field; the corral and pile were not significantly different from one another due to high temporal variability (Fig. 5, Table 3). Pile CH_4 fluxes ranged from −0.09 ± 0.14 ng C·cm^{-2}·h^{-1} (April 2013) to 7670 ± 7030 ng C·cm^{-2}·h^{-1} (October 2013, with one chamber measurement of 84 790 ng C·cm^{-2}·h^{-1}), and were generally highest in the winter of 2013–2014. Average corral CH_4 fluxes ranged from −149 ± 191 ng C·cm^{-2}·h^{-1} (March 2014) to 617 ± 323 ng C·cm^{-2}·h^{-1} (April 2014), with no seasonal trend. The field was a consistent net sink of CH_4 with net uptake in 29 of the 33 measurement periods. Monthly average CH_4 fluxes ranged from −2.23 ± 191 ng C·cm^{-2}·h^{-1} (November 2013) to 6.43 ± 7.05 ng C·cm^{-2}·h^{-1} (October 2013, with one chamber measurement of 83.9 ng C·cm^{-2}·h^{-1}), with no seasonal trend but significantly lower CH_4 fluxes in 2014 than 2013 ($P < 0.05$).

Nitrous oxide fluxes were also highly variable over space and time, with significant differences across management types (Table 3). The pile had the largest N_2O fluxes with monthly average fluxes ranging from 19.6 ± 10.7 ng N·cm^{-2}·h^{-1} (February 2013) to 598 ± 158 ng N·cm^{-2}·h^{-1}(November 2013), with no seasonal trends. Pile N_2O and CH_4 (with outliers removed) fluxes were weakly, positively correlated ($r^2 = 0.21$, $P = 0.03$). Average corral N_2O emissions ranged from 0.84 ± 0.50 ng N·cm^{-2}·h^{-1} (November 2013) to 563 ± 142 ng N·cm^{-2}·h^{-1} (June 2013) and were generally higher during the rainy season. They were positively correlated with monthly precipitation ($r^2 = 0.40$, $P < 0.002$), but not soil moisture. In the field, monthly average N_2O fluxes were low, ranging from -0.05 ± 0.10 ng N·cm^{-2}·h^{-1} (October 2013) to 31.7 ± 20.4 ng N·cm^{-2}·h^{-1} (February 2013) and were also generally higher during the rainy season.

Soil CO_2 fluxes were higher from the corral and field immediately following the first rains of the fall (Fig. 6). We did not detect CO_2 pulses associated with the rain at the end of January and beginning of February, which was the first large event in 2 months. When expressed in units of GWP, N_2O fluxes were nearly as large as CO_2 fluxes for the corral and pile (Fig. 7). The field had net CH_4 consumption, but the rates were low and insufficient to offset net emissions from the corral and pile (Fig. 7).

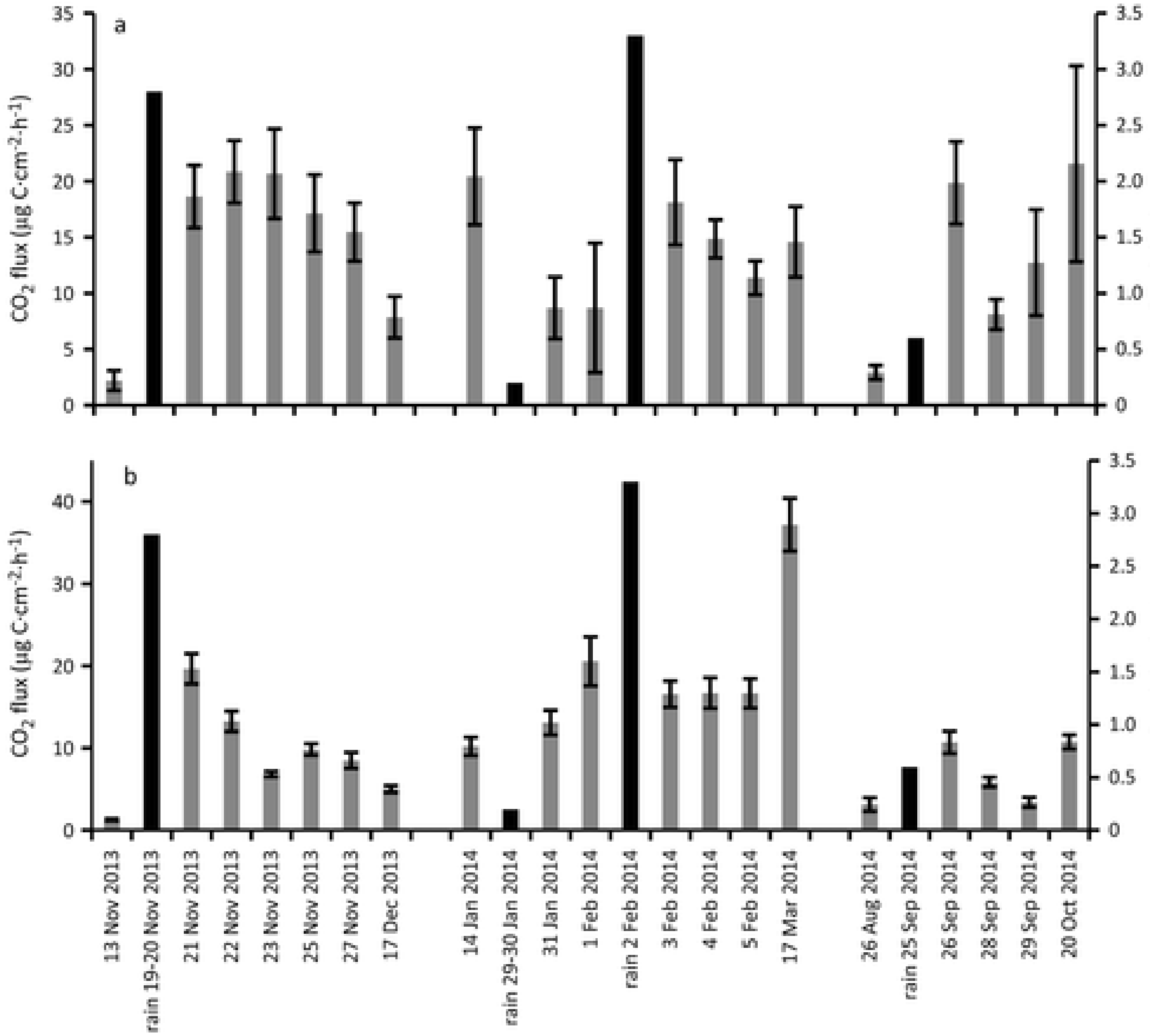

Figure 6

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Pulses of CO_2 from (a) the corral and (b) the field following rain events. Precipitation events are black bars, mean CO_2 fluxes are gray bars, error bars are standard error.

Caption

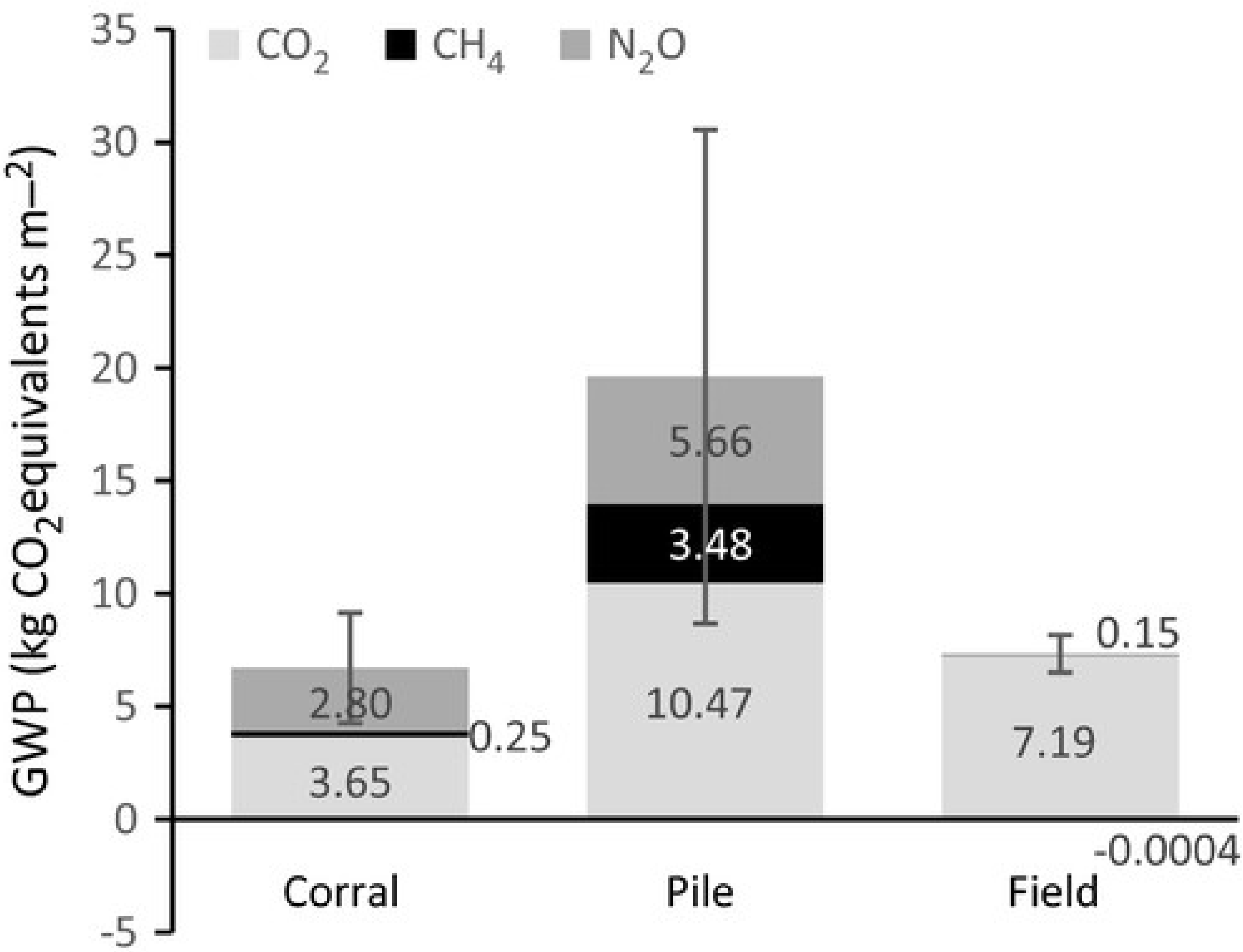

Figure 7

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Global warming potential (GWP) of cumulative fluxes in 2013. Error bars are the sum of the standard errors of the three gases. CO_2 equivalents are the amount of CO_2 that would have the equivalent global warming impact as the gas being measured.

Caption

Management effects on GHG fluxes

The corral, pile, and field were subject to different management practices that had the potential to impact GHG fluxes. In the corral, scraping and removing surface material did not affect GHG fluxes or C and N concentrations of the surface. The pile was turned on 14 May 2014 and turning was completed a few hours before that month's GHG measurements. Prior to turning, O_2 concentrations averaged 15.5% ± 1.9% (mean ± SD) at 30 cm and 19.4% ± 1.3% at 55 cm whereas they were 2.9% ± 3.7% at 120 cm. In the week after turning, O_2concentrations decreased to <1% at all depths, though O_2 oscillated as high as 10% at 30 cm and 5% at 55 cm in the first two days of measurements (Fig. 8a). Oxygen concentrations gradually increased in the two upper depths over the following 3 months, approaching pre-turning levels by the end of August. They also oscillated diurnally, with highs between 15:00 and 20:00 and lows between 0:00 and 08:00 (Fig. 9a). At 120 cm, O_2 remained low. Turning increased pile temperatures from pre-turning (20°–22°C at all depths) to peaks of 49.84°, 58.21°, and 52.02°C at 30, 55, and 120 cm, respectively (Fig. 8b). Temperatures decreased slowly at 30 and 55 cm over the next 3 months, whereas the temperature stayed between 45° and 50°C at 120 cm. Temperatures also oscillated diurnally, but on different schedules at the different depths, and the amplitude of oscillation decreased with depth (Fig. 9b). Fluxes of CO_2, CH_4, and N_2O measured in May 2014 were higher relative to the preceding 2 or 3 months, corresponding to low O_2 concentrations (Fig. 5). The effect had disappeared by the next measurements in June, though high temperatures and low O_2 concentrations persisted throughout the manure pile.

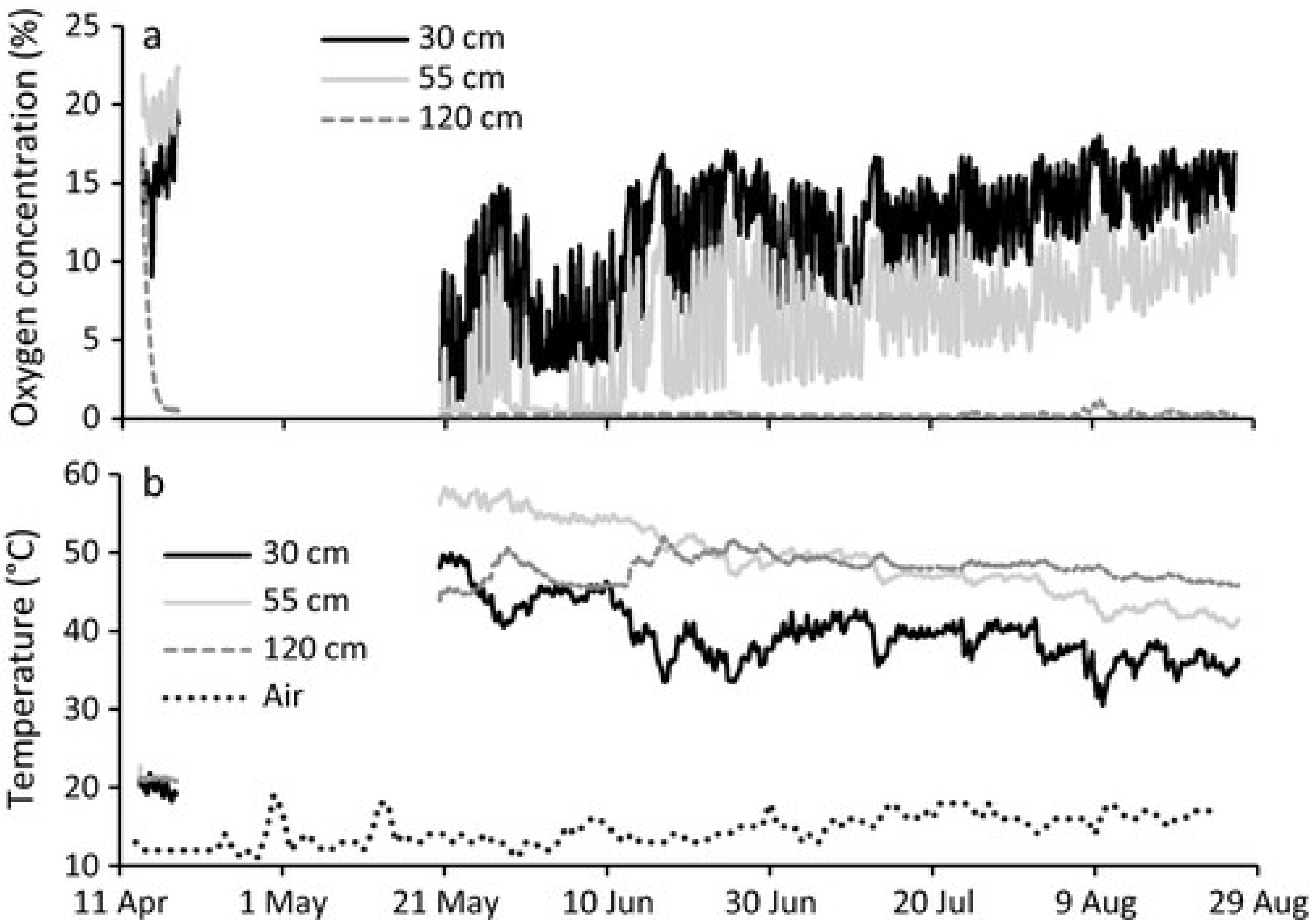

Figure 8

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(a) Oxygen concentrations and (b) temperature at three depths in the pile before and after turning (turning occurred 14 May, data collection 20 May).

Caption

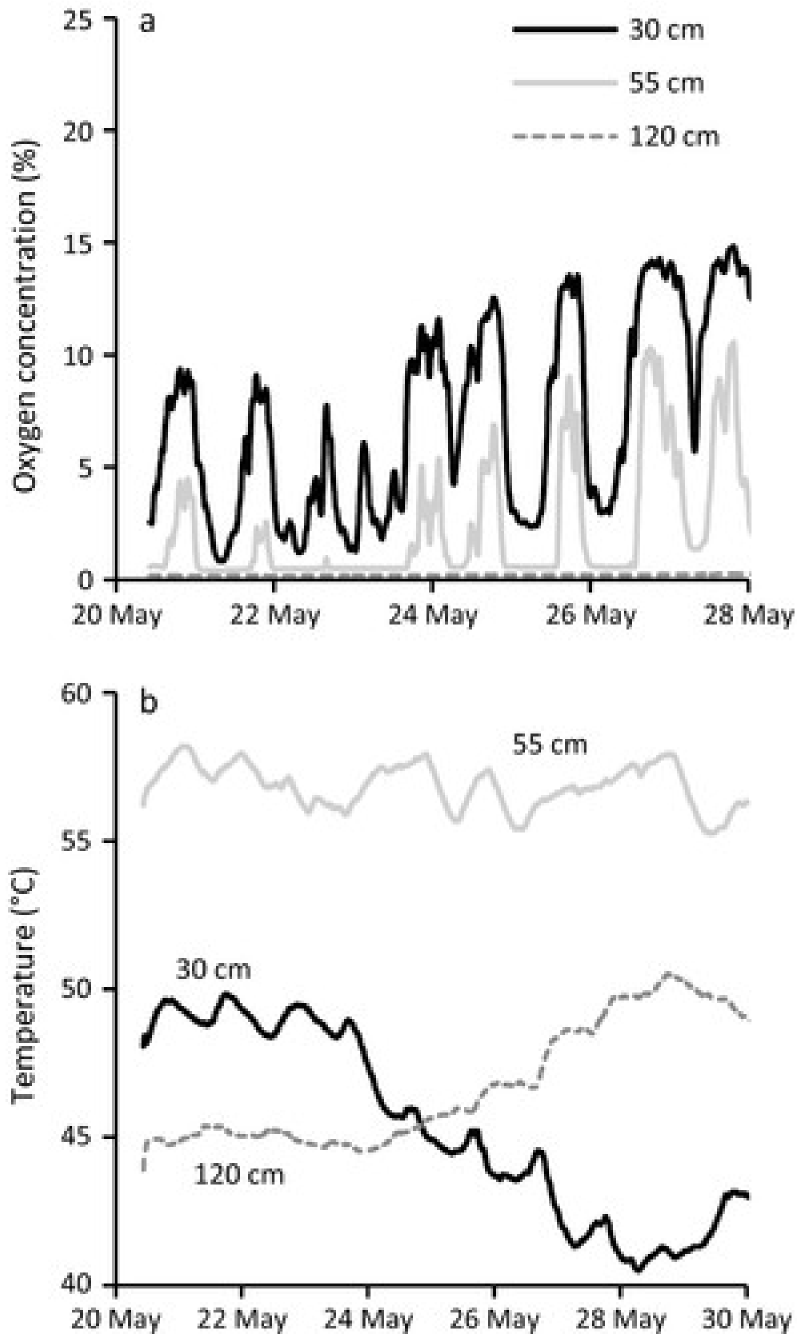

Figure 9

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Diurnal oscillations in pile (a) oxygen concentrations and (b) temperature at three depths.

Caption

In the field, CO_2 fluxes increased with vegetation height (r^2 = 0.86, P < 0.0001, Fig. 10a). Vegetation height was also a good proxy for biomass density (r^2 = 0.95, P < 0.03) and showed a typical trend of winter growth and late summer senescence (Fig. 10b). Harvest events (between the April and May, May and June, and July and August measurements in 2013) reduced vegetation height and CO_2 fluxes but both rapidly increased as the grass regrew. The only exception was July 2013 when vegetation height was high and CO_2 flux stayed low (Fig. 10b). Aeration and manure spreading did not increase N_2O fluxes, but may have set up conditions for greater emissions following the first rains. However, manure addition was not required to produce relatively higher N_2O fluxes, which were also observed in February 2013 and March and May 2014.

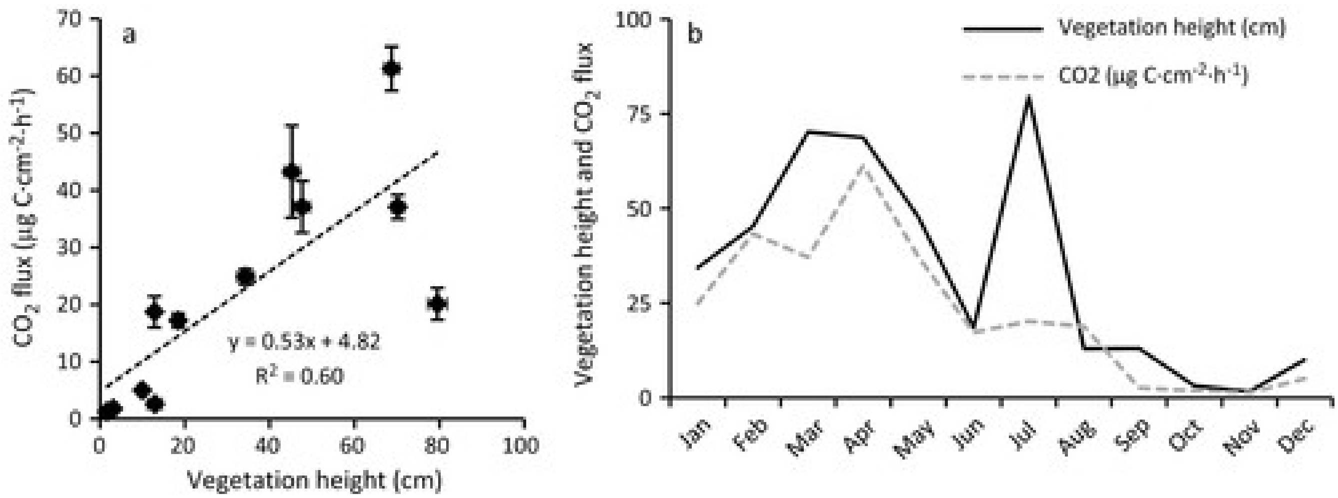

Figure 10

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Field vegetation height vs. (a) CO_2 flux and (b) their seasonal cycles.

Caption

Discussion

Manure and soil composition effects on GHG fluxes

Soil and manure total C and N concentrations were not correlated to patterns in GHG fluxes within or across management types. The corral had the highest C and N concentrations, both of which increased during the summer dry season (corresponding to low CO_2 fluxes as microbes were water limited) and decreased in the winter wet season, when the highest CO_2fluxes occurred. Greenhouse gas fluxes were higher in the pile than the corral despite lower C and N concentrations, likely due to greater overall moisture and temperature and lower O_2availability. The field had relatively low C and N concentrations compared to the other sites, which, when combined with seasonal drought, resulted in much lower CH_4 and N_2O fluxes.

Inorganic N concentrations in the fresh manure were comparable to the few other published measurements (Griffin et al. 2005, Pettygrove et al. 2009). Ammonium concentrations in the fresh manure were the highest of those measured and were five times greater than the corral surface. Inorganic N was also high in the corral surface; our results were similar to those of

Schuman and McCalla (1975) and within the range measured by Vaillant et al. (2004). Corral N_2O fluxes were not correlated with inorganic N concentrations, however. Nitrate gradually increased over the course of 2013, but did not increase N_2O fluxes. One possible explanation is that moisture and temperature were negatively correlated in the corral, thus, water or temperature limitations may have reduced the ability of microbes to respond to high substrate ability. This may also explain why total N_2O emissions for 2013 were relatively low compared to other measurements of dairy corrals (ranging from 0.001 to 0.22 kg $N_2O \cdot m^{-2} \cdot yr^{-1}$, summarized in Owen and Silver 2015).

Climate and GHG fluxes

Models of GHG fluxes from manure and soils generally assume that GHG fluxes are sensitive to soil moisture and temperature (Parton et al. 1998, Dong et al. 2006, Del Prado and Scholefield 2008, Rotz et al. 2011); however, several factors appeared to obscure or counteract these effects in the substrates measured here. First, moisture content was somewhat independent of precipitation in the pile and corral. Pile moisture followed a Long-term seasonal drying and wetting, yet through this seasonal cycle the core of the pile stayed moist and anaerobic throughout the year beneath a relatively dry surface crust. In the corral, urine inputs maintained moist conditions in areas with high cow presence throughout the year while the rest of the area fluctuated seasonally.

Second, moisture and temperature were inversely correlated, i.e., when the sites were moist in the winter, GHG production may have been inhibited by cold temperatures, whereas when the sites were consistently warm, they were also the driest. The majority of studies on GHG fluxes from piles, corrals, and fields have been in temperate regions where moisture availability and temperature are positively correlated (Owen and Silver 2015). This suggests that using mean annual temperature as the control on the range of emission factors for each manure management type, as the IPCC does (Dong et al. 2006), may overestimate emissions from systems with summertime water limitations such as those in Mediterranean and some semiarid climates.

Greenhouse gas fluxes, particularly CO_2 and N_2O, were also expected to be sensitive to individual wetting events. They have been shown to increase in crop fields following rainfall and irrigation (Ryden and Lund 1980, Burger et al. 2005) and in rangelands (Clayton et al. 1997, Chou et al. 2008; M. S. DeLonge and W. L. Silver, *unpublished data*). We observed pulses of $CO_2$from the corral and field following the first rain of autumn both years, but none from the pile and no pulses of N_2O. The center of the pile was persistently moist and the surface moisture did not change much with rain, so the insensitivity of pile emissions to rain events is not surprising.

Rainfall is highly variable in California and our study occurred during relatively dry and warm years. These conditions are similar to those expected in the future under climate change (Mastrandrea and Luers 2011). Drought typically decreases GHG production from soils by limiting water for microbial activity and increasing soil aeration, whereas increased temperatures stimulate microbial processes (Conrad 1996, Davidson et al. 2000, Davidson and Janssens 2006). Although 2014 was drier and warmer than 2013, the only significant difference in GHG fluxes at the sites between years was decreased CO_2 fluxes in the field. This appeared to be driven by decreased plant biomass likely reflecting greater drought stress on vegetation.

GHG fluxes from manure management

Emissions of all three GHGs were highest from the pile, consistent with predictions from other studies (Dong et al. 2006, Owen and Silver 2015). Manure piles have high organic matter content that helps retain moisture and provides substrate for microbial growth as well as an anaerobic core facilitating methanogenesis and denitrification. However, the pile measured here had lower GHG emissions compared to other studies, with fluxes comparable to the low end of the ranges summarized in Owen and Silver (2015). This could be due to the negatively correlated temperature and moisture regime of this region, and/or because the pile contained older material. Nitrous oxide fluxes from the pile in 2013 had a GWP equal to CO_2 fluxes, highlighting the importance of N_2O fluxes for understanding the total climate impacts of livestock agriculture as well as potential mitigation opportunities.

The corral had the second highest total GWP of CH_4 and N_2O (Table 4, Fig. 7). The corral soil was composed primarily of fresh and partially decomposed manure, providing ample substrate for microbial decomposition and denitrification. However, annual scraping and removal of the surface material decreased its thickness and allowed the surface to dry rapidly, except where cows congregated. This rapid drying likely drove the seasonal variation in C and N, where C and N increased in the summer as microbes became water limited. It also may have affected total CO_2 emission rates, which were lower than other measurements (3.6 ± 0.9 kg CO_2/m^2 summed for 2013 vs. a range of 12–365 kg $CO_2 \cdot m^{-2} \cdot yr^{-1}$, summarized in Owen and Silver 2015). Several studies of corral emissions observed net CH_4 consumption (Kaharabata et al. 2000, Bjorneberg et al. 2009; when corrected for cow emissions as in Owen and Silver 2015), a common process in many soils (Conrad 2007). Only during one measurement period in this study was there average CH_4 consumption (March 2014) and of the total 396 chamber measurements, less than 11% had negative CH_4 fluxes. With little to no underlying soil, the corral surface material may only rarely experience conditions conducive to CH_4 oxidation.

Table 4. Total greenhouse gas emissions for 2013 summed over the area of each site

Parameter	Corral	Pile	Field
Area (ha)	0.28	0.042	1.67
CO_2 (Mg CO_2e)	10.21 ± 2.50	4.40 ± 1.90	120.05 ± 12.41
CH_4 (Mg CO_2e)	0.71 ± 0.64	1.46 ± 1.31	−0.01 ± 0.06
N_2O (Mg CO_2e)	7.85 ± 3.68	2.38 ± 1.38	2.54 ± 1.36
Total (Mg CO_2e)	18.77 ± 6.82	8.24 ± 4.59	122.58 ± 13.83

- *Note:* CO_2e refers to CO_2 equivalents (the amount of CO_2 that would have the equivalent global warming impact as the gas being measured).

Field GHG emissions had the lowest GWP of the three sites (Table 4, Fig. 7). Measured N_2O fluxes were almost an order of magnitude larger than those measured at a nearby rangeland (Ryals and Silver 2013), but the manure amendment used here had much higher mineral N content and our measured N_2O fluxes were comparable to those measured from other fertilized rangelands globally (Clayton et al. 1997, Jones et al. 2005).

Unique among the sites considered here, the field was generally a small net sink for CH_4, which is consistent with observations from other rangelands locally (Ryals and Silver 2013) and globally (Conrad 2007). Given its larger surface area compared to the corral and pile, we thought it might offset some GHG emissions. We calculated the total GHG fluxes integrated over each management practice's area for 2013 using the interpolated fluxes (Table 4). Methane oxidation in the field was inadequate to offset other GHG emissions, including N_2O fluxes from the field, making the field a net source. The corral had the greatest non-CO_2-related GWP in 2013 due to its high N_2O emission rate and greater area compared to the pile (Table 4). Only for the pile was CH_4 a relatively important component of total GWP, and it was still exceeded by the GWP of

N_2O. Overall, mitigating N_2O fluxes would have the greatest potential to decrease the climate impact of manure management on this site.

Though we report CH_4 emissions with and without outliers, the averages that include all the data are likely more representative of the sites. Hot spots and hot moments, locations or periods of greatly increased biogeochemical process rates (McClain et al. 2003), are well documented in terrestrial ecosystems. By using 12 replicate chambers per site, we attempted to capture both background fluxes and hot spots. Monthly sampling may have been inadequate to capture hot moments, such as those of N_2O that likely followed the application of liquid manure to the field. Our daily measurements following the first fall rains attempted to capture hot moments driven by wet up, and succeeded only for CO_2.

Management and GHG fluxes

Few studies have measured environmental conditions within manure piles and related them to GHG emissions. Pile mixing is done to aerate the pile and redistribute moisture and materials for even decomposition. In a well-aerated pile, CO_2 should be the primary GHG emitted. High biological O_2 demand due to ample C and N availability, however, can create anaerobic zones in which CH_4 and N_2O are produced. The IPCC estimates that composting (aerobic decomposition) has a lower CH_4 conversion factor (MCF, the fraction of the maximum CH_4 production potential of the manure achieved by the management practice) of 0.5–1.5% compared to static manure piles, which have MCFs of 2.0–5.0%, depending on temperature (Dong et al. 2006). In contrast, the N_2O emission factor for composting (0.006–0.1 kg N_2O-N/kg N excreted, depending on the composting technique used) is slightly greater than the emission factor for static piles (0.005 kg N_2O-N/kg N excreted), though these emission factors are estimated to have an uncertainty of a factor of two (Dong et al. 2006). Limited field observations have demonstrated the variability of GHG fluxes from piles, with responses sometimes counter to model predictions. Piles that are rarely mixed can have high GHG emissions that are unevenly distributed across the pile (Sommer et al. 2004), and some turned piles have higher GHG fluxes than static ones (El Kader et al. 2007, Ahn et al. 2011). Resolving this variability is important for determining best management practices of solid manure piles.

In this study, turning increased mean GHG fluxes from the pile, but the effect was short-lived and undetectable by the next month. Pulses of GHG in response to turning typically last a few days to 2 weeks (Hao et al. 2001, Fukumoto et al. 2003, El Kader et al. 2007, Maeda et al. 2010, Ahn et al. 2011). The GHG pulses were driven by increased temperatures and decreased O_2 at all depths, to levels associated with much younger material. This was a surprise given that most of the pile had been constructed before January 2013, at least 17 months before the turning event, and the responses of pile temperature and O_2 to turning typically decrease over time and with subsequent turnings (Fukumoto et al. 2003, El Kader et al. 2007, Ahn et al. 2011). A large portion of the pile had likely been anaerobic since construction so the material may have remained relatively undecomposed due to anaerobic inhibition of microbial activity. Except at the bottom of the pile, O_2 and temperature followed diurnal patterns before and after turning. At the shallowest depth, temperature peaked in mid- to late afternoon and was lowest in the early morning, tracking air temperature. Oxygen concentrations at all depths followed the same pattern, which suggests that as the pile heated, more external air was being drawn in through convection, at a rate exceeding O_2 consumption by microbes. Thus, the diurnal variations in O_2 and temperature appear to have been produced by physical drivers rather than the microbial processes that drove the longerbased approaches), but this term trends in O_2 and temperature.

Management also affected GHG emissions from the field. Liquid manure application and harvest were the main management activities, but only biomass management had detectable effects. It is possible that we missed a short pulse of N_2O directly following manure application, but this pulse was likely short-lived (Venterea and Rolston 2000, Venterea et al. 2005, Sosulski et al. 2014). As a result, our calculation of total N_2O flux in 2013 may be an underestimate. A model estimate using the IPCC Tier 2 approach (Dong et al. 2006) and a maximum emission factor (0.03 kg N_2O–N/kg N applied) suggests an emission rate of 0.4 g N_2O–N/m^2. This is equal to 4.56 ng N·cm^{-2}·h^{-1}, or 123% of the mean flux calculated for 2013 and within the standard error. Therefore, our underestimate is likely small and does not change the interpretation of our results. Field CO_2 fluxes were increased by the presence of vegetation, reflecting plant and rhizosphere respiration; therefore, CO_2 fluxes were sensitive to any management that affected plant biomass. We observed decreased CO_2 fluxes following harvests, during plant senescence, and in the drier year.

Conclusions

Our data show that patterns in GHG fluxes differ across soil-based manure management practices. Nitrous oxide emissions were an important component of the total GWP for all sites, and were similar to or exceeded that of CH_4 for each management unit. The pile had the highest GHG emissions of the three sites, as expected. However, when the fluxes were integrated over their total areas then the corral was the greatest source of N_2O and the field the greatest source of CO_2. Greenhouse gas emissions were not significantly correlated with soil moisture and temperature; this may be due to the inverse correlation between moisture and temperature in the Mediterranean climate. This poses a challenge to modelers and requires further data gathering and meta-analyses to better represent emissions from these agriculturally productive zones. Our results suggest that the solid manure management components on dairies such as this one can produce substantial GHG, and that minimizing the amount of time manure is stored in corrals and static piles would help mitigate emissions.

Acknowledgments

This research was partially supported by grant #69-3S75-10-172 from the NRCS-USDA to the Environmental Defense Fund and W. Silver as part of the Conservation Innovation Program. Support was also generously provided by grants from the 11th Hour Project, the Lia Fund, and the Rathmann Family Foundation to W. Silver. W. Silver was also supported by the USDA National Institute of Food and Agriculture, McIntire Stennis project CA-B-ECO-7673-MS. We thank Ryan Salladay, Laura Southworth, Laralyn Yee, Rina Estera, Allegra Meyers, Heather Dang, and Marcia DeLonge for help with field work and the dairy farmer who made his farm available. In kind support was provided by John Wick, Jeff Creque, and other members of the Marin Carbon Project.

Note

- 1www.wunderground.com