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The impact of water management practices on subtropical pasture methane emissions and ecosystem service payments

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Abstract. Pastures are an extensive land cover type; however, patterns in pasture greenhouse gas (GHG) exchange vary widely depending on climate and land management. Understanding this variation is important, as pastures may be a net GHG source or sink depending on these factors. We quantified carbon dioxide (CO_2) and methane (CH_4) fluxes from subtropical pastures in south Florida for three wet-dry seasonal cycles using eddy covariance, and estimated two annual budgets of CO_2, CH_4, and GHG equivalent emissions. We also estimated the impact of water retention practices on pasture GHG emissions and assessed the impact of these emissions on stakeholder payments for water retention services in a carbon market framework. The pastures were net CO_2 sinks sequestering up to 163 ± 54 g CO_2-C·m^{-2}·yr^{-1} (mean ± 95% CI), but were also strong CH_4 sources emitting up to 23.5 ± 2.1 g CH_4-C·m^{-2}·yr^{-1}. Accounting for the increased global warming potential of CH_4, the pastures were strong net GHG sources emitting up to 584 ± 78 g CO_2-C eq.·m^{-2}·yr^{-1}, and all CO_2 uptake was offset by wet season CH_4 emissions from the flooded landscape. Our analysis suggests that CH_4 emissions due to increased flooding from water management practices is a small component of the pasture GHG budget, and water retention likely contributes 2–11% of net pasture GHG emissions. These emissions could reduce water retention payments by up to ~12% if stakeholders were required to pay for current GHG emissions in a carbon market. It would require at least 93.7 kg CH_4-C emissions per acre-foot water storage (1 acre-foot = 1233.48 m^3) for carbon market costs to exceed water retention payments, and this scenario is highly unlikely as we estimate current practices are responsible for 11.3 ± 7.2 kg CH_4-C emissions per acre-foot of water storage. Our results demonstrate that water retention practices aimed at reducing nutrient loading to the Everglades are likely only responsible for a minor increase in pasture GHG emissions and would have a small economic consequence in a carbon market.

Key words: *carbon dioxide; eddy covariance; Everglades; Florida ranchlands; greenhouse gas budget; water retention.*

Introduction

Pastures cover ~22% of Earth's ice-free surface (Ramankutty et al. 2008), and play an important role in global carbon (C) exchange by removing ~0.2–0.7 Pg C from the atmosphere annually (Follett and Schuman 2005). Policies are implemented internationally to promote C sequestration and reduce greenhouse gas (GHG) emissions from pastures (Follett and Reed 2010). However, pastures are developed on all inhabited continents across a diversity of biomes (Ramankutty et al. 2008), so their GHG budgets are likely variable across climate and soil types. Dominant controls of pasture GHG exchange include net ecosystem exchange (NEE) of carbon dioxide (CO_2) by grasslands and methane (CH_4) emissions from grazing livestock.

Pasture CO_2 and CH_4 fluxes vary widely depending on climate and the ecosystems upon which they are developed. For example, pastures developed on drained peatlands are major sources of CO_2 as highly organic soils oxidize when exposed to the atmosphere (Nieveen et al. 2005, Teh et al. 2011, Hatala et al. 2012, Knox et al. 2015); whereas upland pastures tend to be net sinks of CO_2 (Soussana et al. 2007, Dengel et al. 2011, Mudge et al. 2011). Methane emissions from grazing livestock often offsets grassland GHG sink strength (Allard et al. 2007, Soussana et al. 2007, Dengel et al. 2011), as CH_4 has a global warming potential (GWP) 28–84 times greater than CO_2 over a 100- and 20-yr time horizon, respectively (Intergovernmental Panel on Climate Change [IPCC]

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2014). Methane and nitrous oxide (N_2O, another potent GHG) emissions from the underlying landscape are particularly important in regions that experience flooding (Teh et al. 2011, Chamberlain et al. 2015; N. Gomez-Casanovas et al., *unpublished manuscript*), as is common throughout the tropics and subtropics. Due to the variety of factors influencing pasture GHG exchange, there is great uncertainty about the GHG forcing associated with pastures and how this forcing varies with climate and management.

Land management impacts ecosystem GHG fluxes; however, its effects can be complex and have received little research attention (Baldocchi 2014). For instance, increased grazing pressure and fertilization increase CH_4 and N_2O emissions, respectively, and can reduce pasture GHG sinks up to 89% (Allard et al. 2007). The restoration of peatland pastures to wetlands changes these systems from strong GHG sources to sinks due to the changing balance between CO_2 and CH_4 fluxes (Knox et al. 2015), and water management practices may exert a particularly large influence on CH_4 emissions, as soil microbes produce CH_4 under anoxic conditions (Conrad 2007). Water is often heavily managed in mesic regions, such as south Florida where this study was conducted, yet the influence of water management practices on GHG fluxes is rarely evaluated.

Runoff from cattle pastures is a major source of nutrient loading to the Everglades, an important wetland complex in the south Florida region, and water retention and storage programs have been implemented to reduce nutrient runoff by reducing drainage from pastures in the Northern Everglades watershed (Appendix S1: Fig. S1). These programs aim to reduce phosphorus (P) and nitrogen (N) loading to downstream ecosystems by installing riser board water control structures to increase water and nutrient retention on pastures in the Everglades watershed (Bohlen et al. 2009). Pasture water retention projects are assumed to reduce total P loading by 30% (Florida Ranchlands Environmental Services Project [FRESP] 2012). While these programs effectively reduce nutrient loads to downstream ecosystems and compensate ranchers for the ecosystem services provided (Bohlen et al. 2009, Bohlen and Villapando 2011), the impact of these practices on pasture GHG emissions is unknown. Understanding the influence of water retention practices on pasture GHG emissions is important, given the high CH_4 production and emission rates observed from south Florida pasture soils during flooding events (Chamberlain et al. 2016).

The objectives of this study were to (1) estimate annual pasture NEE, CH_4, and GHG budgets, (2) estimate the impact of water retention practices on pasture GHG budgets, and (3) determine how CH_4 emissions associated with water retention might impact ecosystem service payments in a carbon market framework. We measured ecosystem-scale CO_2 and CH_4 fluxes for 2.5 yr using eddy covariance, and used these data to estimate annual GHG budgets and determine environmental controls of fluxes. Annual N_2O emissions were also calculated using IPCC Tier 2 guidelines for manure deposition on pastures (IPCC 2006). We then explored the influence of water retention on pasture GHG budgets by combining water table data from pasture water retention treatments with CH_4 flux data from our eddy covariance tower. This work advances our understanding of pasture GHG emissions and helps to develop a context for evaluating trade-offs between water quality and climate regulating ecosystem services from a globally dominant land use.

Methods

Study site

We measured CO_2 and CH_4 fluxes from a fenced pasture (92.1 ha; 27.1632004° N, 81.187302° W) that was rotationally grazed throughout the measurement period at an average capacity of ~1.4 cows/ha. The pasture is located within a 4290-ha commercial cattle ranch, Buck Island Ranch, which also operates as the MacArthur Agro-ecology Research Center (MAERC), an ecological research station and division of Archbold Biological Station (Fig. 1). The pastures are planted with an introduced forage grass, *Paspalum notatum,* and have not received herbicide or fertilizer since 2006 and 2007, respectively. Roughly 30 ha of the pasture were intentionally burned in January 2013. The region experiences heavy rainfall and flooding during the summer wet season, and pastures are drained by a network of ditches and canals throughout the landscape. Mean annual precipitation (1980–2015) is 1310 mm, with two-thirds of total annual precipitation falling from June to September (DayMet database; Thornton et al. 2012). The study pasture is 74.2% *P. notatum* pasture, 10.9% depressional wetland, 9.2% *Sabal palmetto* hammock, 4.0% drainage ditch, and 1.7% drainage canal by area (Chamberlain et al. 2015), and is established on fine sand spodosol soils (Bohlen and Villapando 2011).

Eddy covariance measurements

From May 2013 to November 2015, we measured ecosystem-scale fluxes of CO_2, CH_4, H_2O, and sensible heat using an eddy covariance tower installed in the center of the pasture described above. At a height of 2.6 m and a 10 Hz interval, we measured three-dimensional wind speed and direction with a sonic anemometer (CSAT3; Campbell Scientific, Logan, Utah, USA) and CO_2, H_2O, and CH_4 concentrations with open-path infrared gas analyzers (LI-7500A and LI-7700; Licor, Lincoln, Nebraska, USA). All instruments interfaced with a datalogger (LI-7550; LI-COR Biogeosciences) and data were transmitted by modem for processing. We also made ancillary atmospheric and hydrologic measurements at 30 min averaging intervals and logged these data to an additional logger (CR3000; Campbell Scientific) synchronized to the eddy covariance logger. Ancillary

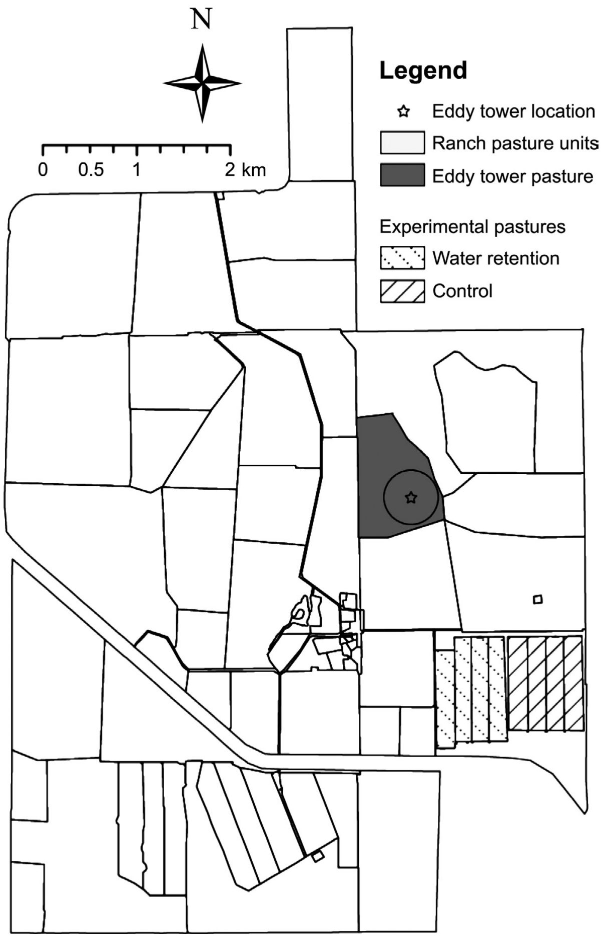

FIG. 1. Map of Buck Island Ranch, highlighting the eddy covariance tower location and pasture boundaries. Water retention treatment experiments occurred in the block of eight "experimental pastures" from 2005 to 2009. The circle surrounding the eddy tower location represents mean 90% daytime tower footprint across the entire measurement period (290 m).

measures at the tower site included air temperature and relative humidity (HMP115; Viasala, Helsinki, Finland), incoming radiation with a net radiometer (CNR4; Kipp & Zonen, Bohemia, New York, USA), water table depth with a pressure transducer (CS451; Campbell Scientific), and volumetric water content at 5, 10, and 20 cm depths with water content reflectometers (CS616; Campbell Scientific). Rainfall was measured at 30-min intervals with a tipping bucket gauge (TB4; Hydrologic Services America, Lake Worth, Florida, USA) located 1.7 km southwest of the tower (27.150475° N, 81.198568° W).

Fluxes were calculated from the covariance of vertical wind speed and scalars ([CO_2], [CH_4], [H_2O], heat) over 30-min intervals. Raw data were screened for spikes, drop-outs, amplitude resolution, absolute limits, skewness, and kurtosis as described in Vickers and Mahrt (1997) and designated default in EddyPro 4.2 (LI-COR Biogeosciences). We aligned the anemometer with mean wind flows using double-rotation corrections, used block averaging to calculate mean wind speeds and concentrations over the 30-min interval, and corrected for time lags between wind and gas concentration measurements by covariance maximization. We corrected for air density fluctuations using the Webb-Pearl-Leuning correction (Webb et al. 1980) and applied analytic spectral corrections according to Moncrieff et al. (1997, 2004). Quality of fluxes were then flagged according to Foken et al. (2004), where quality flags range from 1 (best) to 9 (worst) depending on atmospheric turbulence, flux stationarity, and flow distortions by the tower structure. All of the above processing and corrections were conducted using EddyPro 4.2 (LI-COR Biogeosciences) and a summary of these corrections can be found in Appendix S2: Table S1. Processed fluxes were rejected when atmospheric conditions were poor (quality flags > 4 for CO_2, quality flags > 6 for CH_4), and additional CH_4 fluxes were rejected when the CH_4 sensor path was blocked, CH_4 concentrations were below 1.74 ppm or above 5.00 ppm, and when fluxes were above 1500 $nmol \cdot m^{-2} \cdot s^{-1}$ or below −500 $nmol \cdot m^{-2} \cdot s^{-1}$ (Dengel et al. 2011, Baldocchi et al. 2012). We relaxed the quality flag criteria for CH_4 to reduce the number of CH_4 fluxes removed, and to include periods when cattle grazed the footprint that may induce non-stationary CH_4 fluxes over half-hourly intervals. Overall, 42% of all CH_4 fluxes and 36% of all CO_2 fluxes were removed from analysis. In this study, negative fluxes represent ecosystem uptake and positive fluxes represent ecosystem emissions. The daytime 90% tower footprint was calculated according to Hsieh et al. (2000) and was 290 m across the entire measurement period, which is within the boundaries of the fenced pasture (Fig. 1). The flux footprint measured an area that is largely grassland pasture with intersecting drainage ditches, and also included two depressional wetlands located at the outer reaches of the 90% flux footprint (Chamberlain et al. 2015). We did not remove fluxes from our analysis when the footprint extended into wetlands because these landscape features are characteristic of pastures in the region.

Gap filling, NEE partitioning, and annual budgets

We filled gaps in the NEE half-hourly time series using the Marginal Distribution Sampling method described in Reichstein et al. (2005). CO_2 fluxes were grouped according to incoming radiation, air temperature, and vapor pressure deficit conditions and mean values were used to fill missing data during similar conditions. When meteorological data were not available, CO_2 fluxes were filled based on measured fluxes occurring at similar times of day. The filled NEE time series was partitioned into gross ecosystem productivity (GEP) and ecosystem respiration (R_{eco}) using the method described in Reichstein et al. (2005). Here, the relationship between air temperature and nighttime NEE (when GEP is zero) was described using the Lloyd and Taylor (1994) model, and this relationship was then used to estimate R_{eco} in all periods using the modelled temperature relationship. GEP was then estimated as the difference between NEE and R_{eco}. A complete description of both methods can be found in Reichstein et al. (2005). All NEE gap filling and partitioning was conducted in R 3.2.0 (R Core Team, 2015) using the REddyProc package (Reichstein et al. 2016).

We filled gaps in the CH_4 time series by linear interpolation for gaps of up to 2.5 h and longer gaps were filled using the mean diurnal variation method (Dengel et al. 2011), where gaps were filled based on mean fluxes observed in the same half-hour period on adjacent days. We sampled adjacent half-hourly periods using a moving window of 7 d and calculated error estimates for gap-filled values based on the standard deviation of measured fluxes used in the moving window average. Separate moving window averages were used for periods with and without cattle present in the pasture.

Annual NEE, CH_4, and GHG budgets were calculated by integrating daily sums from the gap-filled time series over two annual cycles (1 April 2013 to 31 March 2014; 1 April 2014 to 31 March 2015). Both annual cycles begin near the onset of the pastures growing/wet season. We calculated the GHG budget by adding daily NEE and CH_4 budgets in terms of CO_2 equivalent emissions, where 1 g CH_4 is equivalent to 28 g CO_2 in the atmosphere based on the CH_4 GWP over the 100-yr time horizon (IPCC 2014). The GHG budgets presented here include measured fluxes only (NEE + CH_4). Uncertainty in annual NEE and CH_4 budgets were integrated from half-hourly flux random errors estimated as in Finkelstein and Sims (2001) and gap-filled estimate errors. Uncertainty in the GHG budget was estimated from the additive variance of the NEE and CH_4 budgets. All relationships between fluxes and environmental variables were described using regression analyses, and CH_4 fluxes were log-transformed to meet assumptions of normality. All regressions between half-hourly fluxes and environmental variables were conducted using high quality non-gap-filled half-hourly data for both CO_2 and CH_4 (quality flag 3 or less).

We estimated annual manure N_2O budgets using IPCC Tier 2 guidelines and daily stocking data for the pasture at Buck Island Ranch containing the eddy covariance tower. Daily N_2O emissions from deposited manure were calculated using the following equation (IPCC 2006):

$$N_2O_m = \frac{N \cdot N_{ex}}{365} \cdot EF_{N20}$$

where N_2O_m is the total amount of N_2O produced from manure on the pasture (kg N_2O-N/d), EF_{N2O} is the N_2O emission factor for "pasture range and paddock" deposited manure (mean 0.02, range 0.005–0.03 kg N_2O-N/kg N excreted), N_{ex} is the N excretion rate per non-dairy North American cattle (70 kg $N \cdot cow^{-1} \cdot yr^{-1}$), and N is the daily cattle population within the fenced pasture. EF_{N2O} and N_{ex} values are from the IPCC (2006). Total N_2O emissions from manure were then converted to comparable units (g $N_2O\text{-}N \cdot m^{-2} \cdot yr^{-1}$) by dividing annual emissions by the total pasture area (92.1 ha). We did not calculate fertilizer N_2O emissions because fertilizer has not been applied to these pastures since 2007.

Water retention analysis

To assess the influence of water retention on CH_4 fluxes, we analyzed existing water table data from eight side-by-side experimental pastures broken into two treatments (Fig. 1). These pastures were 1.5 km southeast (27.144434° N, 81.177001° W) of the eddy covariance tower, and within these pastures water flow was reduced on one block of four pastures (water retention treatment) and water flow was left unobstructed on the other block of four pastures (control). These data were initially published in Bohlen and Villapando (2011), and were used to track the efficacy of water control structures for increasing pasture water and nutrient retention. Water table depth was measured at 20-min intervals for 4 yr from all eight pastures (January 2005 to January 2009). Flooding dynamics in the experimental pastures are likely analogous to those in the eddy covariance tower pastures because both are established on the same soil type, planted with *P. notatum*, drained by ditch networks, and rotationally grazed by cattle.

We estimated the influence of water retention practices on net CH_4 emissions and annual GHG budgets by (1) estimating the influence of water retention practices on surface soil flooding and then (2) estimating the influence of extended surface soil flooding on CH_4 emissions according to the following equation:

$$F_{WR} = D \cdot F_{SH}$$

where F_{WR} is the total estimated CH_4 flux due to water retention, D is the duration of surface soil flooding (0–15 cm depth) due to water retention, and F_{SH} is the average CH_4 flux when the water table is within the surface soil horizon (0–15 cm depth). First, we estimated the influence of water retention on soil flooding (D) by calculating daily mean water table depths for retention and non-retention pastures (2005–2009), and D was calculated as the annual difference between the two treatments of the number of days when the water table was in the surface organic horizon (0–15 cm below surface). We were interested in how long this soil horizon is flooded because, in this region, flooding of the top 15 cm of soil controls ecosystem CH_4 emissions (Chamberlain et al. 2016). We then estimated total CH_4 emissions due to water retention (F_{WR}) by multiplying the annual difference in surface flooding duration (D) by the mean CH_4 emission rate when the water table was within the 0–15 cm surface soil horizon (F_{SH}), as estimated from our eddy covariance CH_4 flux data and accounting for flux uncertainty. We also estimated the regional impact of water retention on CH_4 emissions by extrapolating these findings across the areal extent of water retention ranchlands in the Northern Everglades region making the assumption that flooding dynamics, water retention efficacy, and CH_4 emission rates observed at Buck Island Ranch are characteristic of the region. The water retention structures used in the these treatments are the same used in regional water management projects, so our estimates of D and the responses observed in these pastures are likely representative of the region (Bohlen and Villapando 2011).

We also conducted a cost-benefit analysis to understand how these water-retention-driven CH_4 emissions might impact stakeholder ecosystem service payments if CH_4 emission incurred a cost in a carbon market. Here, we estimated the price of CO_2 eq. emissions from California Carbon Allowance Futures used in California's Cap and Trade program (US\$12.58/t CO_2 eq. [1 t = 1 Mg]; Climate Policy Initiative 2016) because California has an active carbon market whereas Florida does not. We then estimated carbon market costs associated with CH_4 emissions per acre-foot of water storage (1 acre-foot = 1233.48 m^3) for current water retention practices and CH_4 emissions associated with a simulated range of surface soil flooding durations up to 365 d/yr. We use the non-metric unit acre-feet because this is the unit used by both agencies and stakeholders (FRESP 2012, South Florida Water Management District [SFWMD] 2012). Regional pasture water retention data, including water retention pastures area (ha), average annual water retention (acre-feet storage/ha), and annual payment for retention services (US\$/acre-foot storage) were provided by MAERC. All dollar amounts referenced in this work are US dollars (US\$). All data processing, analysis, and visualization was conducted in R 3.2.0 using the dplyr (Wickham and Francois 2016), ggplot2 (Wickham 2009), and zoo packages (Zeileis and Grothendieck 2005).

Results

Climate and weather

Pasture meteorological and hydrological conditions were characteristic of the humid subtropics, where temperature (°C), incoming radiation ($MJ \cdot m^{-2} \cdot d^{-1}$), precipitation (mm/d), and water table depth (m below surface) all displayed strong seasonality (Fig. 2). Air temperature was highest in the wet season (May through September) and decreased in the dry season (October through April). In general, daily air temperature was more variable during the winter, occasionally dropping below 0°C during January and February both years (Fig. 2a). Daily incoming radiation also peaked in summer, but was highly variable on a day-to-day basis due to changing cloud cover (Fig. 2b). The frequency and intensity of

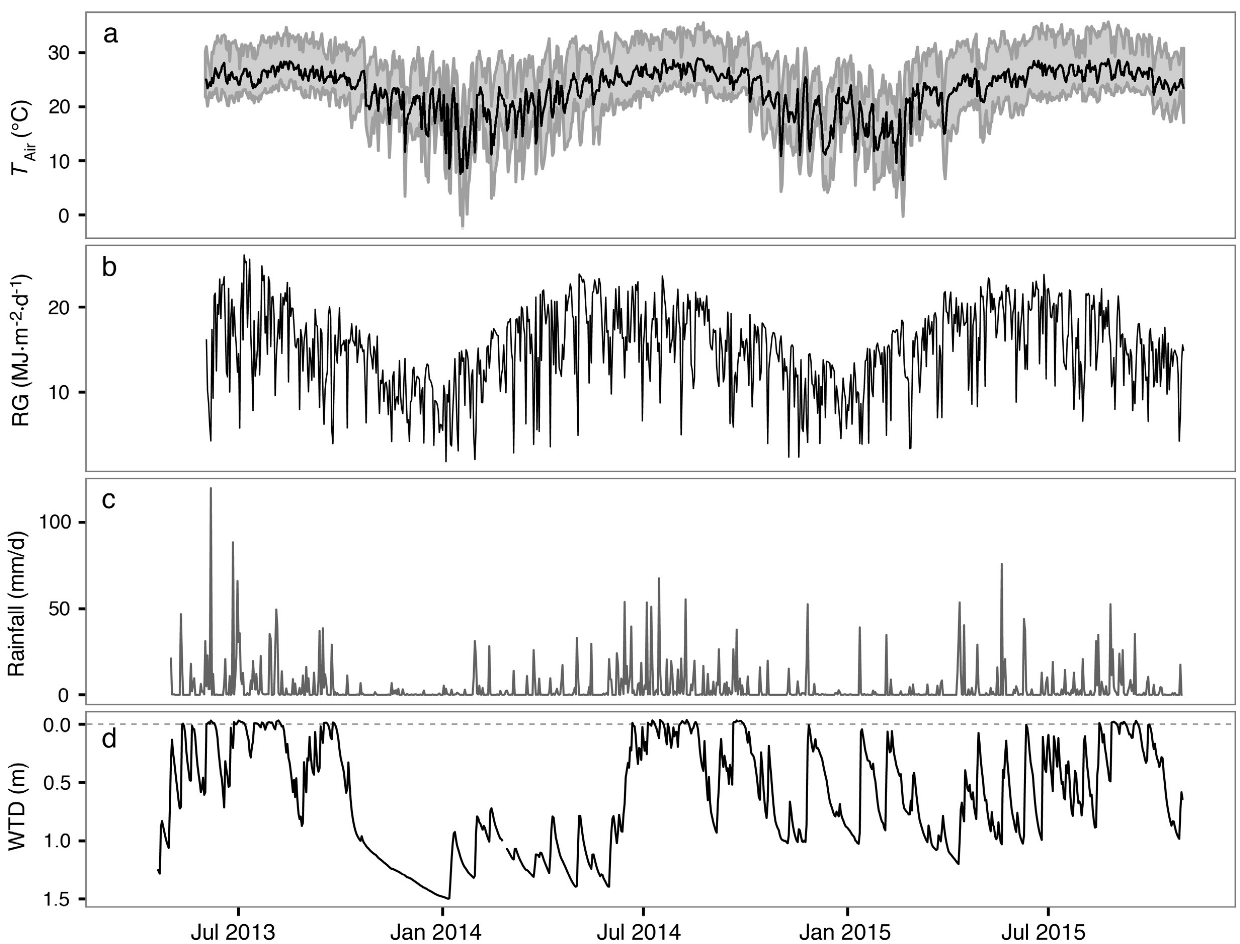

Fig. 2. Pasture (a) daily mean air temperature (T_{air}), (b) incoming radiation (Rg), (c) rainfall, and (d) water table depth (WTD). Shaded area on panel a bounds daily minimum and maximum air temperature. The dashed line on panel d represents land surface elevation and the solid line represents WTD.

rainfall events increased during summer months, with the most rainfall occurring between May and September each year. The summers of 2013 and 2014 exhibited clearly defined wet seasons where most rainfall events occurred during the summer and were separated by an extended dry period (Fig. 2c). Large rainfall events were more dispersed throughout 2015, causing a less clear delineation between dry and wet seasons (Fig. 2c). The frequency and duration of summer rain events had a clear influence on pasture water table dynamics. In both the 2013 and 2014 wet seasons, extended periods of surface flooding occurred during heavy rainfall periods and both seasons were separated by a period of pasture dry down (Fig. 2d). In contrast, in 2015, independent rain events drove many water table recharge events throughout the year but did not cause extended flooding until September 2015 when rainfall frequency increased (Fig. 2).

Seasonal cycles of CH_4, NEE, partitioned fluxes (GEP, R_{eco}), and annual budgets

Pasture CH_4 emissions were driven by landscape flooding and were primarily correlated to water table fluctuations as described in detail in Chamberlain et al. (2015, 2016). Ecosystem CH_4 emissions peaked in the wet season during periods of extended pasture flooding, and appreciable emissions were not observed when the water table reached the surface for 1 d or less (Fig. 3a). We observed a weak correlation between GEP and CH_4 fluxes at the daily time scale when the water table was within 5 cm of the land surface (r^2 = 0.13, P = 0.0012; Appendix S2: Fig. S1); however, these relationships were not observed when pasture water table was lower than 5 cm depth or in the half-hourly time series. Across both years, the pastures were sources of CH_4, emitting 23.4 ± 1.5 g CH_4-C·m^{-2}·yr^{-1} (mean ± 95% CI) in 2013–2014 and 23.5 ± 2.1 g CH_4-C·m^{-2}·yr^{-1} in 2014–2015 (Table 1).

Net ecosystem exchange followed similar seasonal cycles, where daily net CO_2 uptake was observed during the wet season and net CO_2 loss was observed during the dry season (Fig. 3b). Daily NEE was highly variable, and many periods of CO_2 emission were observed during the growing season, often adjacent to periods of CO_2 uptake (Fig. 3b). Much of this daily variability was driven by variability in pasture GEP, which exhibited high variability compared to R_{eco} (Fig. 3c). Variability in GEP was likely driven by the variability in pasture incoming radiation, as we observed a strong correlation between daily GEP estimates and incoming radiation (r^2 = 0.47, P < 0.0001). We observed a strong correlation between daily NEE and incoming radiation (r^2 = 0.41, P < 0.0001; Appendix S2: Fig. S2), and daily NEE correlated weakly

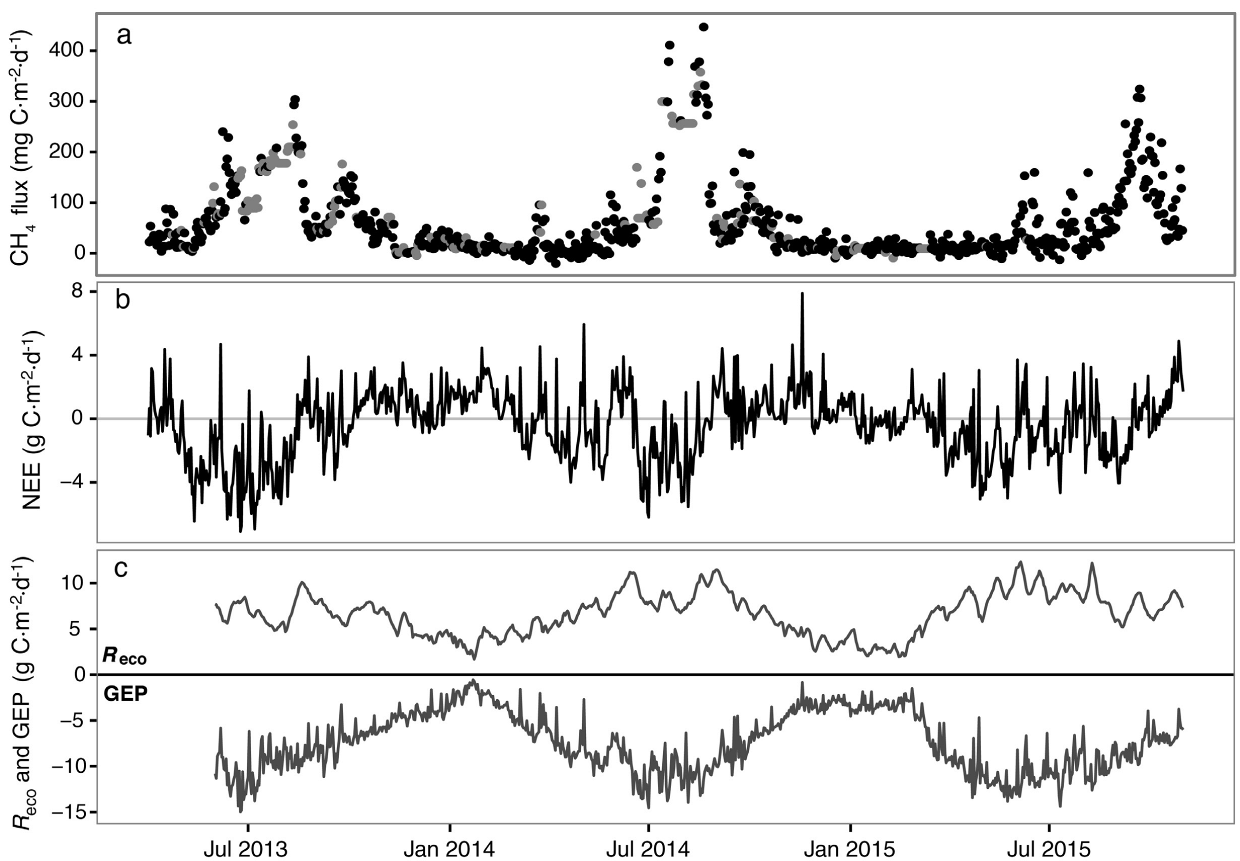

Fig. 3. Daily (a) pasture CH_4 fluxes, (b) net ecosystem exchange (NEE), and (c) partitioned NEE fluxes. In panel c, NEE is partitioned into ecosystem respiration (R_{eco}) and gross ecosystem productivity (GEP). Gray dots on panel a represent days when more than 66% of the half-hourly fluxes were gap filled.

TABLE 1. Pasture net ecosystem exchange (NEE), CH_4, and greenhouse gas (GHG) equivalent emissions (NEE + CH_4) for 2 yr.

Year (April–March)	NEE (g CO_2-C·m^{-2}·yr^{-1})	CH_4 (g CH_4-C·m^{-2}·yr^{-1})	GHG (g CO_2-C eq.·m^{-2}·yr^{-1})	Water retention (percentage of budget)
2013–2014	−163 ± 54	23.4 ± 1.5	491 ± 68	6.7 ± 4.3
2014–2015	−75 ± 51	23.5 ± 2.1	584 ± 78	5.7 ± 3.6

Notes: Water retention impact is estimated CH_4 emissions due to water retention practices. Measured fluxes only are included in GHG budget (N_2O estimates omitted). Values are mean ± 95% CI.

with water table depth ($r^2 = 0.17$, $P < 0.0001$; Appendix S2: Fig. S3) and daily temperature ($r^2 = 0.07$, $P < 0.0001$). Pastures were a CO_2 sink, sequestering 163 ± 54 g CO_2-C·m^{-2}·yr^{-1} in 2013 and 75 ± 51 g CO_2-C·m^{-2}·yr^{-1} in 2014 (Table 1). The reduced CO_2 sink strength in 2014 was likely driven by an extended period of CO_2 loss in June 2014 when GEP decreased and R_{eco} increased (Fig. 3c). This event occurred during a particularly dry early summer when the water table was not recharged until late June 2014 (Fig. 2), at which point CO_2 uptake resumed (Fig. 3b).

When we express pasture CH_4 emissions in terms of CO_2 equivalents, the pastures become a strong net source of GHGs (Fig. 4). After accounting for the GWP of CH_4, the pastures produced 491 ± 68 g CO_2-C eq.·m^{-2}·yr^{-1} in 2013 and 584 ± 78 g CO_2-C eq.·m^{-2} yr^{-1} in 2014 (Table 1). Here, the differences in annual GHG budgets were due to differences in annual NEE, as the pastures produced similar amounts of CH_4 each year (Table 1 and Fig. 4). In both years, the pastures were initial GHG sinks until CH_4 emissions began at the onset of the wet season, and once CH_4 emissions began, the pastures remained strong GHG sources for the rest of the year (Fig. 4).

Calculations based on the number of cows depositing manure within the pastures indicate that the pastures were net sources of N_2O both years, producing an estimated 0.17 (range 0.04–0.26) g N_2O-N m^{-2} yr^{-1} in 2013, and 0.15 (0.04–0.23) g N_2O-N m^{-2} yr^{-1} in 2014. These manure-based emission estimates correspond to 45 (5–30) g CO_2-C eq. m^{-2} yr^{-1} in 2013, and 17 (5–26) g CO_2-C eq. m^{-2} yr^{-1} in 2014, where 1 g N_2O is equivalent to 265 g CO_2 in the atmosphere based on the N_2O GWP over the 100-yr time horizon (IPCC 2014).

Water retention impacts

On average, water retention structures caused the water table to remain within the surface soil horizon (0–15 cm) for an additional 10.75 d/yr based on 4 yr of pasture water table data presented in Bohlen and Villapando (2011). There was large interannual variability in this retention impact; water retention structures caused an additional 37 d/yr of surface soil flooding in 2005, an additional 6 d/yr in 2006, 19 fewer d/yr in 2007, and an additional 19 d/yr in 2008. The reduced flooding observed in 2007 was driven by an extended drought, which prevented flow of water from nearby canals to the pastures. A more detailed description of flooding dynamics in these experimental pastures can be found in Bohlen and Villapando (2011). Surface soil flooding dynamics were very similar between the water retention treatment pastures and the nearby eddy covariance tower pasture (where water retention is applied). The surface soil horizon within the eddy covariance tower pasture was flooded for 79.0 d/yr across 2.6 yr, and the surface horizon within the water retention treatment pastures was flooded for 79.3 d/yr across 3 yr excluding the 2007 drought year. We excluded this drought year in

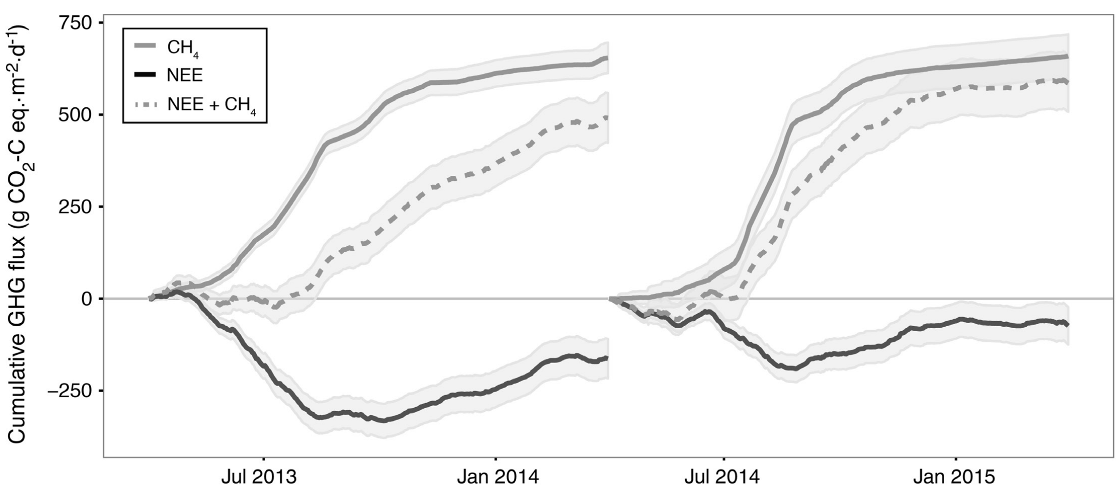

FIG. 4. Cumulative greenhouse gas (GHG) fluxes for net ecosystem exchange (NEE, black line), CH_4 (gray line), and CO_2 equivalent (eq) emissions (dotted line, NEE + CH_4) using the 100-yr CH_4 global warming potential. Gray shaded areas surrounding each line represents the 95% confidence interval of cumulative flux and gap-filled value uncertainty.

this comparison because drought conditions were not observed during the eddy covariance study.

From 2013 to 2015, eddy covariance-measured CH_4 fluxes were 109.82 ± 70.10 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ (median ± interquartile range, $n = 119$ d) when the water table was within the surface soil horizon (0–15 cm) and 21.50 ± 21.12 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ ($n = 615$ d) when the water table was below this horizon (Mann–Whitney, $P < 0.0001$). There was no difference in ecosystem CH_4 fluxes when the water table was within 5 cm of surface (131.25 ± 60.61 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$, $n = 69$ d) compared to the entire 0–15 cm soil horizon (Mann–Whitney, $P = 0.30$). We observed a weak correlation between daily NEE and water table depth ($r^2 = 0.17$, $P < 0.0001$; Appendix S2: Fig. S3) and, for this reason, we did not calculate the influence of water retention on CO_2 fluxes.

If we assume that water retention within the tower footprint creates 10.75 d/yr of additional surface soil flooding (water retention structures are currently installed within the pasture), then water retention accounts for 1.18 ± 0.75 g CH_4-C m^{-2} yr^{-1} of net CH_4 emissions. This is attributable to 6.7% ± 4.3% and 5.7% ± 3.6% of the total pasture GHG budget in 2013 and 2014, respectively (Table 1). Within the context of ecosystem services, water retention practices at Buck Island Ranch are estimated to store 1573 acre-feet of water annually across 1517 ha of land, at an average rate of 1.04 acre-feet of water per hectare of pasture (SFWMD 2012). Given our estimate that 1.18 ± 0.75 g CH_4-C m^{-2} yr^{-1} are emitted due to water retention practices at Buck Island Ranch, we therefore estimate that 11.35 ± 7.21 kg CH_4-C are emitted per acre-foot of water stored annually.

Cost–benefit analysis: water retention payments vs. carbon market costs

The Northern Everglades Payment for Environmental Services (NE-PES) project pays Buck Island Ranch \$99/acre-foot water storage annually, and compensation rates vary from \$74 to \$158/acre-foot water storage (Appendix S1: Table S1). If Buck Island Ranch were required to pay for CH_4 emissions associated with water retention, this compensation would decrease, and in the following calculations we use the \$12.58/t CO_2 eq. carbon price in California's cap and trade market (Climate Policy Initiative 2016). Using our estimate that water retention increases surface flooding by 10.75 d/yr and this results in 11.35 ± 7.21 kg CH_4-C per acre-foot of annual water storage, the resulting CH_4 emissions would cost stakeholders \$4.00 to \$11.99/acre-foot water storage using the 100-yr and 20-yr GWP of CH_4, respectively. Using the 20-yr GWP of CH_4 in cost calculations, 93.69 kg CH_4-C/acre-foot water storage would need to be emitted for carbon costs to exceed water retention payments, which would require 89 d/yr of surface soil flooding due to water retention practices (Fig. 5a). Using the 100-yr GWP of CH_4 in cost calculations, 281.06 kg CH_4-C/acre-foot water storage would need to be emitted for carbon costs to exceed water retention payments, which would require 267 d/yr of surface soil flooding due to water retention practices (Fig. 5b).

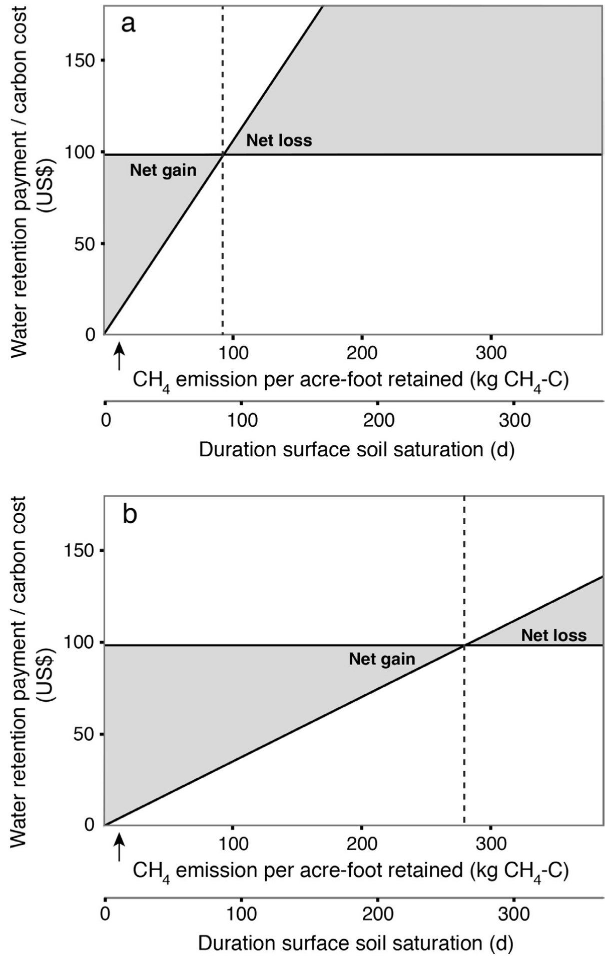

Fig. 5. Balance between water retention payments and carbon emission costs (US\$) across a range of CH_4 emissions per acre-foot of water stored (1 acre-foot = 1233.48 m^3). Emissions per acre-foot were calculated from a range of surface flooding durations (d/yr) due to water retention assuming the 1.04 acre-feet water stored per hectare of land documented for Buck Island Ranch. Panel a calculates CH_4 emission costs using the 20-yr CH_4 global warming potential (GWP), and panel b calculates emission costs using the 100-yr CH_4 GWP. On both panels, the horizontal line marks the current payment per acre-foot of water stored at Buck Island Ranch (US\$99), the diagonal line marks costs of CH_4 emissions, the vertical dashed line marks the point at which CH_4 emission costs exceed water retention payments, and the black arrow below both *x*-axes marks the current estimated magnitude of CH_4 emissions per acre-foot of water retained at Buck Island Ranch.

Discussion

Our analyses suggest that water retention interventions to improve water quality do not have a large impact on pasture GHG emissions. Methane emissions associated with these interventions account for only 2–11% of annual pasture GHG emissions. Total GHG emissions appear to be relatively insensitive to water retention practices because the associated emissions are small compared to the natural landscape and cattle CH_4 emissions (Chamberlain et al. 2015). It is likely that our findings are generalizable to the region because the water retention structures used at Buck Island Ranch are the same used in regional water management projects (Bohlen and Villapando 2011); however, further studies are warranted to evaluate our assumptions of similar CH_4 emission and flooding dynamics across regional water retention pastures. For example, CH_4 emission rates from depressional wetlands imbedded within the pasture landscape are over two times higher than emissions from pasture (Chamberlain et al. 2015), and similarly large emissions are observed from depressional wetlands overlying karst in the region (C. R. Hinkle, *personal communication*). Pastures with a higher density of imbedded weltands may have increased ecosystem CH_4 emissions and increased water-retention-related emissions if stored water is concentrated within these imbedded wetlands. Additionally, including our estimates of N_2O emissions from manure deposition would increase pasture GHG budgets by 3.9% (1.0–6.1%) in 2013–2014, and 2.9% (0.9–4.5%) in 2014–2015. The inclusion of these N_2O fluxes would in turn reduce the mean contribution of water retention CH_4 emissions to pasture GHG budgets to 6.5% in 2013–2014 and 5.5% in 2014–2015. Direct studies of N_2O emissions from these pastures are warranted, particularly since our IPCC-based estimates do not account for potentially large N_2O fluxes from flooded land. It may also be important to account for fertilizer-based N_2O emissions in some cases, but these pastures have not received fertilizer applications since 2007. Regardless, the true contribution of water retention-related emissions to pasture GHG budgets is likely lower than presented here due to potential N_2O emissions that were not directly quantified in this study.

It would require a substantial increase in surface soil flooding duration to make water retention unprofitable in a carbon market framework. Given that observed water retention practices increase surface soil flooding an average of 10.75 d/yr to a maximum of 37 d/yr, it appears unlikely that these practices could induce the minimum flooding increase of 89 d/yr required for carbon costs to exceed retention payments using the 20-yr GWP of CH_4. Using the more conservative 100-yr GWP of CH_4, would require 267 d/yr of surface soil flooding for costs to exceed payments, which is considerably higher than the total 59.75 d/yr of surface soil flooding observed in these pastures, which also includes all natural flooding events. Based on these analyses, water retention practices will likely remain profitable for stakeholders even if carbon market payments for CH_4 emissions are required.

Water retention is implemented across 1517 ha of improved pasture at Buck Island Ranch, so these practices could therefore be responsible for 17.91 ± 11.43 t CH_4-C/yr of total emissions (assuming the 10.75 d/yr flooding increase). Seventeen ranches are currently implementing water retention through the NE-PES program overseen by the South Florida Water Management District (Appendix S1: Fig. S1). Of these 17 ranches, three main management styles are used; 10 implement pasture water retention similar to Buck Island Ranch, two use wetland-only water storage, four use reservoir systems, and one uses both pasture retention and reservoirs (Appendix S1: Table S1). In total, these 17 ranches are projected to retain water on 24755 ha of ranchlands (Appendix S1: Table S1), which could be responsible for 292.11 ± 185.66 t CH_4-C/yr of emissions, assuming the 10.75 d/yr flooding increase. To put these values in perspective, the average cow in the region emits 58.9 kg CH_4-C/yr (0.0598 t CH_4-C/yr; Chamberlain et al. 2015). Current water retention practices are therefore equivalent to ranging an additional 1807–8112 cattle in the Northern Everglades region. Buck Island Ranch, where this study took place, grazes ~3000 cattle at any given time, and Highlands county alone, where the ranch is located, has a population of 125000 cattle (United States Department of Agriculture: National Agricultural Statistics Survey [USDA NASS] 2015). This suggests that any impact of current water retention practices on CH_4 emissions is small compared to the CH_4 emitted by the regional cattle population. This calculation of regional emissions assumes that CH_4 emission rates in reservoir and wetland storage areas are similar to pasture retention areas; however, emission rates are likely variable across management type and further research is warranted to quantify fluxes from reservoir and wetland storage areas. Current water retention areas comprise 13.5% of ranchlands in the Northern Everglades watershed, and the total area of improved pastures covers 183778 ha of land in the region (Hiscock et al. 2003). Regional water retention CH_4 emissions could be as high as 542.40 ± 346.22 t CH_4-C/yr or 1084.80 ± 692.43 t CH_4-C/yr if 25% or 50% of total pasture area in the Northern Everglades watershed were used for water retention, respectively.

The NE-PES program continues to compensate participating ranches and additional ranches are being added to the program through $46 million in funding (FRESP 2012). Pasture water retention is the most frequently implemented practice in the NE-PES program, but wetland restoration and reservoir systems are also widely implemented throughout the region (Appendix S1: Fig. S1 and Table S1). On average, wetland and reservoir systems store more water per hectare of land (2.9 acre-feet water/ha) and likely exhibit different GHG dynamics than discussed in this work. Further research is needed to assess GHG emissions relative to ecosystem services for these additional water management areas located throughout south Florida. Regardless, this study demonstrates that subtropical improved pastures are strong GHG sources when accounting for CH_4 fluxes, and water retention practices do not explain the majority of pasture CH_4 emissions. Additionally, current water retention practices would likely remain profitable for stakeholders in a carbon market framework, and these practices would only become a net loss with a substantial, and unrealistic, increase in surface soil flooding due to water retention. This suggests that water retention practices provide the ecosystem services of reduced nutrient loading at a minor climate and economic consequence.

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Literature Cited

Allard, V., et al. 2007. The role of grazing management for the net biome productivity and greenhouse gas budget (CO_2, N_2O and CH_4) of semi-natural grassland. Agriculture, Ecosystems and Environment 121:47–58.

Baldocchi, D. 2014. Biogeochemistry: managing land and climate. Nature Climate Change 4:330–331.

Baldocchi, D., M. Detto, O. Sonnentag, J. Verfaillie, Y. A. Teh, W. Silver, and N. M. Kelly. 2012. The challenges of measuring methane fluxes and concentrations over a peatland pasture. Agricultural and Forest Meteorology 153:177–187.

Bohlen, P. J., S. Lynch, L. Shabman, M. Clark, S. Shukla, and H. Swain. 2009. Paying for environmental services from agricultural lands: an example from the northern Everglades. Frontiers in Ecology and the Environment 7:46–55.

Bohlen, P. J., and O. R. Villapando. 2011. Controlling runoff from subtropical pastures has differential effects on nitrogen and phosphorus loads. Journal of Environmental Quality 40:989–998.

Chamberlain, S. D., E. H. Boughton, and J. P. Sparks. 2015. Underlying ecosystem emissions exceed cattle-emitted methane from subtropical lowland pastures. Ecosystems 18:933–945.

Chamberlain, S. D., N. Gomez-Casanovas, M. T. Walter, E. H. Boughton, C. J. Bernacchi, E. H. DeLucia, P. M. Groffman, E. W. Keel, and J. P. Sparks. 2016. Influence of transient flooding on methane fluxes from subtropical pastures. Journal of Geophysical Research—Biogeosciences 121:965–977.

Climate Policy Initiative. 2016. California carbon dashboard. http://calcarbondash.org/

Conrad, R. 2007. Microbial ecology of methanogens and methanotrophs. Pages 1–63 *in* D. L. Sparks, editor. Advances in agronomy. Elsevier, Amsterdam, The Netherlands.

Dengel, S., P. E. Levy, J. Grace, S. K. Jones, and U. M. Skiba. 2011. Methane emissions from sheep pasture, measured with an open-path eddy covariance system. Global Change Biology 17:3524–3533.

Finkelstein, P. L., and P. F. Sims. 2001. Sampling error in eddy correlation flux measurements. Journal of Geophysical Research—Atmospheres 106:3502–3509.

Florida Ranchlands Environmental Services Project (FRESP). 2012. Frequently asked questions about the Northern Everglades Payment for Environmental Services Program. http://www.fresp.org/pdfs/NE-PES%20FAQs%20SFWMD%20Final.pdf

Foken, T. M., M. Gockede, L. Mauder, B. Amiro. Mahrt, and W. Munger. 2004. Post-field data quality control. Pages 181–208 *in* X. Lee, W. J. Massman, and B. E. Law, editors. Handbook of micrometeorology. Kulwer Academic Publishers, Dordrecht, The Netherlands.

Follett, R. F., and D. A. Reed. 2010. Soil carbon sequestration in grazing lands: societal benefits and policy implications. Rangeland Ecology & Management 63:4–15.

Follett, R. F., and G. E. Schuman. 2005. Grazing lands contributions to carbon sequestration. Pages 266–277 *in* D. A. McGilloway, editor. Grassland: a global resource. Wageningen Academic Publishers, Wageningen, The Netherlands.

Hatala, J. A., M. Detto, O. Sonnentag, S. J. Deverel, J. Verfaillie, and D. D. Baldocchi. 2012. Greenhouse gas (CO_2, CH_4, H_2O) fluxes from drained and flooded agricultural peatlands in the Sacramento-San Joaquin Delta. Agriculture, Ecosystems and Environment 150:1–18.

Hiscock, J. G., C. S. Thourot, and J. Zhang. 2003. Phosphorus budget—land use relationships for the northern Lake Okeechobee watershed, Florida. Ecological Engineering 21:63–74.

Hsieh, C. I., G. Katul, and T. W. Chi. 2000. An approximate analytical model for footprint estimation of scalar fluxes in thermally stratified atmospheric flows. Advances in Water Resources 23:765–772.

Intergovernmental Panel on Climate Change (IPCC). 2006. Emissions from livestock and manure management. Pages 10.1–10.87 *in* H. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe, editors. IPCC guidelines for national greenhouse gas inventories, Prepared by the National Greenhouse Gas Inventories Programme. IGES, Kamiyamaguchi, Japan.

Intergovernmental Panel on Climate Change (IPCC). 2014. Climate change 2014: synthesis report. Contribution of Working Groups I, II and III to the Panel on Climate Change. *In* R. K. Pachauri, and L. Meyer, editors. Fifth Assessment Report to the Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Knox, S. H., C. Sturtevant, J. H. Matthes, L. Koteen, J. Verfaillie, and D. Baldocchi. 2015. Agricultural peatland restoration: effects of land-use change on greenhouse gas (CO_2 and CH_4) fluxes in the Sacramento-San Joaquin Delta. Global Change Biology 21:750–765.

Lloyd, J. J., and J. A. Taylor. 1994. On the temperature dependence of soil respiration. Functional Ecology 8:315–323.

Moncrieff, J. B., R. Clement, J. J. Finnigan, and T. Meyers. 2004. Averaging, detrending, and filtering of eddy covariance time series. Pages 7–31 *in* X. Lee, W. J. Massman, and B. E. Law, editors. Handbook of micrometeorology. Kulwer Academic Publishers, Dordrecht, The Netherlands.

Moncrieff, J. B., J. M. Massheder, H. De Bruin, J. Elbers, T. Friborg, B. Heusinkveld, P. Kabat, S. Scott, H. Soegaard, and A. Verhoef. 1997. A system to measure surface fluxes of momentum, sensible heat, water vapour and carbon dioxide. Journal of Hydrology 188–189:589–611.

Mudge, P. L., D. F. Wallace, S. Rutledge, D. I. Campbell, L. A. Schipper, and C. L. Hosking. 2011. Carbon balance of an intensively grazed temperate pasture in two climatically contrasting years. Agriculture, Ecosystems and Environment 144:271–280.

Nieveen, J. P., D. I. Campell, L. A. Schipper, and I. J. Blair. 2005. Carbon exchange of grazed pasture on drained peat soil. Global Change Biology 11:607–618.

R Core Team. 2015. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Ramankutty, N., A. T. Evan, C. Monfreda, and J. A. Foley. 2008. Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. Global Biogeochemical Cycles 22:1–19.

Reichstein, M., et al. 2005. On the separation of net ecosystem exchange into assimilation and ecosystem respiration: review and improved algorithm. Global Change Biology 11: 1424–1439.

Reichstein, M., A. M. Moffat, T. Wutzler, and K. Sickel. 2016. REddyProc: Data processing and plotting utilities of (half-) hourly eddy-covariance measurements. R package version 0.8-2/r15. https://R-Forge.R-project.org/projects/reddyproc/

Soussana, J. F., et al. 2007. Full accounting of the greenhouse gas (CO_2, N_2O, CH_4) budget of nine European grassland sites. Agriculture, Ecosystems and Environment 121:121–134.

South Florida Water Management District (SFWMD). 2012. FY2012 Projects: Northern everglades—payment for environmental services. sfwmd.gov/portal/page/portal/xrepository/sfwmd_repository_pdf/pay_enviro_svs_approved_contracts.pdf

Teh, Y. A., W. L. Silver, O. Sonnentag, M. Detto, M. Kelly, and D. D. Baldocchi. 2011. Large greenhouse gas emissions from a temperate peatland pasture. Ecosystems 14:311–325.

Thornton, P., M. M. Thornton, B. W. Mayer, N. Wilhelmi, Y. Wei, and R. B. Cook. 2012. Daymet: daily surface weather on a 1 km grid for North America, 1980–2008. Oak Ridge National Laboratory Distributed Active Archive Center, Oak Ridge, Tennessee, USA. http://daymet.ornl.gov/

United States Department of Agriculture: National Agricultural Statistics Survey (USDA NASS). 2015. Southern region news release cattle inventory. https://www.nass.usda.gov/Statistics_by_State/Regional_Office/Southern/includes/Publications/Livestock_Releases/Cattle_Inventory/Cattle_15.pdf

Vickers, D., and L. Mahrt. 1997. Quality control and flux sampling problems for tower and aircraft data. Journal of Atmospheric and Oceanic Technology 14:512–526.

Webb, E. K., G. I. Pearman, and R. Leuning. 1980. Correction of flux measurements for density effects due to heat and water vapour transfer. Quarterly Journal of the Royal Meteorological Society 106:85–100.

Wickham, H. 2009. ggplot2: elegant graphics for data analysis. Springer-Verlag, New York, New York, USA.

Wickham, H., and R. Francois. 2016. dplyr: A grammar of data manipulation. R package version 0.5.0. https://CRAN.R-project.org/package=dplyr

Zeileis, A., and G. Grothendieck. 2005. zoo: S3 infrastructure for regular and irregular time series. Journal of Statistical Software 14:1–27.

Supporting Information

Additional supporting information may be found in the online version of this article at http://onlinelibrary.wiley.com/doi/10.1002/eap.1514/full

Data Availability

Data associated with this paper have been deposited in figshare.
Daily integrated eddy flux data: https://doi.org/10.6084/m9.figshare.4579690
Raw eddy flux data: https://doi.org/10.6084/m9.figshare.3569415
Pasture water retention data: https://doi.org/10.6084/m9.figshare.3569412