

Agricultural peatland restoration: effects of land-use change on greenhouse gas (CO₂ and CH₄) fluxes in the Sacramento-San Joaquin Delta

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Abstract

Agricultural drainage of organic soils has resulted in vast soil subsidence and contributed to increased atmospheric carbon dioxide (CO₂) concentrations. The Sacramento-San Joaquin Delta in California was drained over a century ago for agriculture and human settlement and has since experienced subsidence rates that are among the highest in the world. It is recognized that drained agriculture in the Delta is unsustainable in the long-term, and to help reverse subsidence and capture carbon (C) there is an interest in restoring drained agricultural land-use types to flooded conditions. However, flooding may increase methane (CH₄) emissions. We conducted a full year of simultaneous eddy covariance measurements at two conventional drained agricultural peatlands (a pasture and a corn field) and three flooded land-use types (a rice paddy and two restored wetlands) to assess the impact of drained to flooded land-use change on CO₂ and CH₄ fluxes in the Delta. We found that the drained sites were net C and greenhouse gas (GHG) sources, releasing up to 341 g C m⁻² yr⁻¹ as CO₂ and 11.4 g C m⁻² yr⁻¹ as CH₄. Conversely, the restored wetlands were net sinks of atmospheric CO₂, sequestering up to 397 g C m⁻² yr⁻¹. However, they were large sources of CH₄, with emissions ranging from 39 to 53 g C m⁻² yr⁻¹. In terms of the full GHG budget, the restored wetlands could be either GHG sources or sinks. Although the rice paddy was a small atmospheric CO₂ sink, when considering harvest and CH₄ emissions, it acted as both a C and GHG source. Annual photosynthesis was similar between sites, but flooding at the restored sites inhibited ecosystem respiration, making them net CO₂ sinks. This study suggests that converting drained agricultural peat soils to flooded land-use types can help reduce or reverse soil subsidence and reduce GHG emissions.

Keywords: carbon flux, drained, eddy covariance, greenhouse gas balance, land-use change, methane, peatland, restored, rice, wetland

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Introduction

The worldwide drainage of peatlands for agricultural or forestry purposes has resulted in vast soil subsidence, due to changes in physical conditions and enhanced rates of microbial decomposition (Stephens *et al.*, 1984; Rojstaczer & Deverel, 1993; Syvitski *et al.*, 2009; Hirano *et al.*, 2012). In addition to degrading peat soils and associated habitat, these ecosystems have become globally significant sources of carbon dioxide (CO₂) to the atmosphere, as large amounts of carbon (C) are lost to the atmosphere through oxidation (Armentano, 1980; Drösler *et al.*, 2008; Couwenberg *et al.*, 2010). Nonetheless, this practice is widespread due to the high economic benefit of the fertile soil (Kramer & Shabman, 1993).

The Sacramento-San Joaquin Delta (referred to hereafter as the Delta) in California was drained over a century ago for agriculture and human settlement and has since experienced subsidence rates that are among the highest in the world (Stephens *et al.*, 1984; Rojstaczer & Deverel, 1995). Prior to drainage, the Delta consisted of a network of tidal marshes at the confluence of the Sacramento and San Joaquin Rivers that covered an area of approximately 1400 km² (Shlemon & Begg, 1975; Drexler *et al.*, 2009b). These highly productive ecosystems with low rates of decomposition accumulated up to 18 m of peat as the marsh surface kept pace with gradual sea-level rise over several thousand years (Shlemon & Begg, 1975; Atwater & Belknap, 1980). Since drainage, agricultural cultivation has caused high rates of peat soil oxidation and surface elevations have subsided to more than 8 m below sea level in some regions (Deverel & Rojstaczer, 1996; Drexler *et al.*, 2009a; Deverel & Leighton, 2010). Today the Delta exists as a

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network of islands that maintain an artificially low water table through an extensive levee network and continual pumping (Mount & Twiss, 2005). As the land continues to subside, the risk of levee failure and subsequent flooding also increases. This would cause saltwater intrusion into the Delta, threatening a critical supply of water for California, as the Delta is a transfer point for agricultural and municipal water for more than 22 million people (Miller *et al.*, 2000).

Slowing subsidence and converting drained Delta ecosystems from net C sources to C sinks is key to the long-term sustainability of the Delta and protection of California's water transfer infrastructure. Restoring degraded agriculture systems to flooded land-use types such as rice paddies and restored wetlands has been recognized as a potential management option for the Delta that can prevent further peat oxidation (Miller *et al.*, 2000, 2008; Hatala *et al.*, 2012b). Furthermore, it creates additional benefits such as providing habitat for wildlife. The flooded status of these environments decreases the rate of peat oxidation by physically impeding the transport of oxygen required for most microbial metabolisms into the soil. Consequently, rates of ecosystem respiration (ER) are lower in these flooded

environments than in traditional agricultural systems, and the reduction in CO₂ production can help reverse net C loss (Miller *et al.*, 2000; Hendriks *et al.*, 2007; Eugster *et al.*, 2010; Hatala *et al.*, 2012b; Herbst *et al.*, 2013). Earlier investigations in the Delta have confirmed this result through comparison of conventional drained agricultural crops and pastures, which were large C sources, with flooded agricultural systems (i.e. rice) (Deverel & Rojstaczer, 1996; Hatala *et al.*, 2012b). Wetlands are among the most effective terrestrial ecosystems at building soil organic matter and sequestering C (Rocha & Goulden, 2009; Mcleod *et al.*, 2011; Mitsch *et al.*, 2013). Consequently, wetland restoration has been broadly proposed as a way to mitigate fossil fuel emissions (Zedler & Kercher, 2005; Maljanen *et al.*, 2010; Poffenbarger *et al.*, 2011; Bernal & Mitsch, 2013). High C sequestration rates are expected from restored marshes in the Delta since the region has a long and warm growing season with abundant water and sunlight (Brinson *et al.*, 1981; Miller & Fujii, 2010). Miller *et al.* (2008) investigated the subsidence reversal potential of two restored wetlands in the Delta and found that land-surface elevations increased by an average of 4 cm yr⁻¹ in both wetlands over a 9 year period.

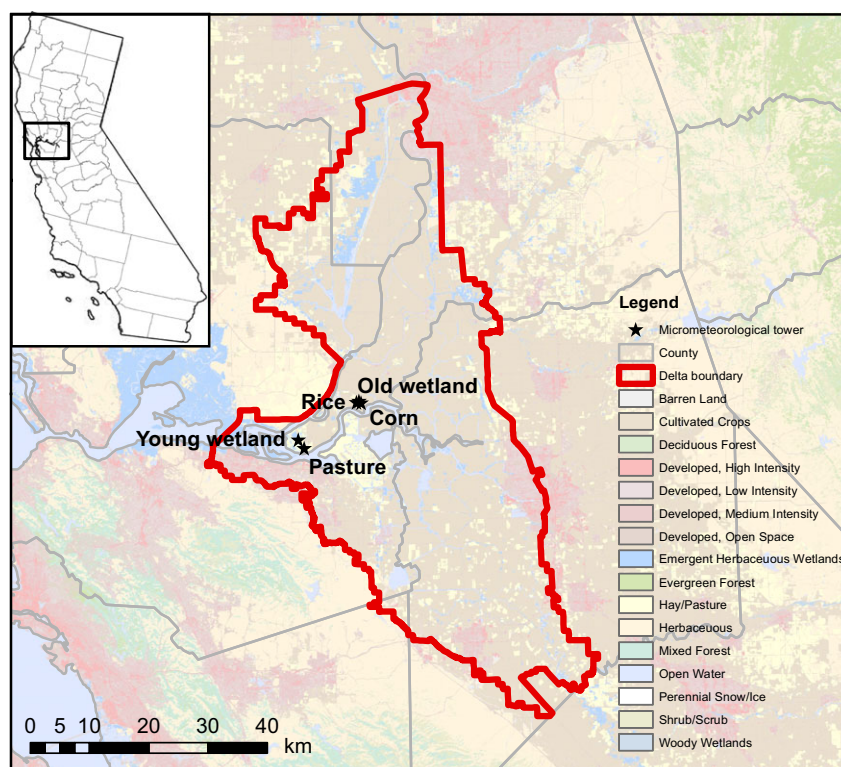


Fig. 1 Location of the five Delta sites. All field sites are located in the Sacramento-San Joaquin Delta, inland of San Francisco Bay. Since these sites are all within ~16 km of each other, they share the same basic meteorology, enabling a direct comparison of differences in the carbon and greenhouse gas budgets between sites. Pasture and corn together cover over 60% of the primary Delta, while rice and wetlands each currently cover less than 1% of the Delta.

However, managing the Delta for enhanced CO₂ sequestration is expected to alter the fluxes of other greenhouse gases (GHGs). While the conversion from drained to flooded land-use types will help stop the net emissions of CO₂ due to peat oxidation, flooding is expected to increase the emissions of methane (CH₄) (Miller, 2011; Teh *et al.*, 2011; Hatala *et al.*, 2012b; Herbst *et al.*, 2013), a GHG with a global warming potential (GWP) 25 times greater than CO₂ over a 100 year time scale (Forster *et al.*, 2007). Therefore, even relatively low rates of CH₄ emissions could offset the benefit of CO₂ sequestration in terms of the net GHG effect. This is often observed in natural wetlands,

particularly in more northern wetlands (Whiting & Chanton, 2001; Blais *et al.*, 2005; Bridgham *et al.*, 2006), however, overall the CH₄ budgets of wetland ecosystems remain highly uncertain (Bridgham *et al.*, 2006). Despite a growing interest in peatland restoration for C sequestration (Drösler *et al.*, 2008; Maljanen *et al.*, 2010), few studies have measured integrated, near-continuous CO₂ and CH₄ fluxes from restored wetlands (e.g. Hendriks *et al.*, 2007; Waddington & Day, 2007; Waddington *et al.*, 2010; Miller, 2011; Herbst *et al.*, 2013; Reid *et al.*, 2013).

In this study, we measured year-round fluxes of CO₂ and CH₄ from three land-use types in the Delta

Table 1 Site characteristics. These sites capture a range of hydrologic conditions within the Delta: the conventional drained agricultural sites are drained year-round, the agricultural wetland is flooded for over half the year but is drained for field preparation and planting, herbicide and fertilizer application and harvest, and the restored wetlands are permanently flooded

| | Conventional drained agricultural sites | | Agriculture wetland site | Restored wetland sites | |
|---|--|--------------------------|---------------------------|---|---|
| | Pasture | Corn | Rice | Young wetland | Old wetland |
| Location | 38.0366°N, 121.7540°W | 38.1047°N, 121.6433°W | 38.10875°N, 121.6530°W | 38.0498°N, 121.7650°W | 38.1074°N, 121.6469°W |
| Elevation (m) | −7 | −5 | −5 | −3 | −9 |
| Measurement period considered in this study | 1 March 2012–1 March 2013 | 9 May 2012–9 May 2013† | 1 March 2012–1 March 2013 | 1 March 2012–1 March 2013 | 1 August 2012–1 August 2013 |
| Average annual air temperature* (°C) | 14.9 | 15.3 | 14.5 | 15.0 | 15.6 |
| Average annual soil temperature* (°C) | 16.0 | 16.4 | 15.7 | 15.6 | 12.5 |
| Average peak growing season air temperature‡ (°C) | 22.2 | 21.9 | 20.3 | 19.6 | 16.6 |
| Annual precipitation* (mm) | 263 | 290 | 390§ | 390§ | 278§ |
| Total incoming radiation (MJ m ^{−2} yr ^{−1}) | 7001 | 7139 | 7001 | 7001 | 7135 |
| Mean (min, max) peak growing season water table depth‡ (cm) | −65 (−92, −37) | −82 (−86, −76) | 9 (−30, +17) | +107 (+102, +109) | +26 (+6, +32) |
| Peak PAI | 2.2 | 3.0 | 4.1 | N/A | N/A |
| Peak aboveground biomass (g DM m ^{−2}) | N/A | 2201 | 2050 | 2303¶ | 1357¶ |
| Typical vegetation | <i>Hordeum murinum</i> L. <i>Lepidium latifolium</i> L. | <i>Zea mays</i> | <i>Oryza sativa</i> | <i>Schoenoplectus acutus</i> <i>Typha</i> spp. | <i>Schoenoplectus acutus</i> <i>Typha</i> spp. |

*Corresponds to the measurement period considered in this study.

†Measurements only began on May 22nd, therefore to complete a full year of measurements, fluxes from May 9th–May 22nd were extrapolated based on the first 13 days of measurement and meteorological variables were either extrapolated or estimated based on measurements from other sites.

‡Corresponds to August and September 2012.

§Values estimated from a nearby California Irrigation Management Information System (CIMIS) station on Twitchell Island since measurements weren't available at these sites.

¶From Byrd *et al.* (2014).

N/A, not applicable.

spanning a range of inundated conditions using the eddy covariance (EC) technique to assess the short-term consequences and tradeoffs of the conversion from drained to flooded land-uses on C capture and GHG emissions. Our study investigated a total of five field sites (Fig. 1) comprising three representative land-use classes: (i) *Conventional drained agricultural sites*: a drained and grazed degraded peatland pasture and a corn field; (ii) *Agricultural wetland site*: a rice paddy; and (iii) *Restored wetland sites*: a newly restored wetland (2010) and a long-term restored wetland (1997). The first goal of this study was to compare the annual C balance of the three different land-use types. We hypothesized that the conventional drained agricultural sites would be large net sources of C while the rice paddy and restored wetlands would be net C sinks and thus viable land-use types for stopping or reversing soil subsidence in the Delta. Our second objective was to quantify the impacts of land-use conversion from drained to flooded ecosystems on CH₄ fluxes due the greater GWP of CH₄ relative to CO₂. This has implications for the overall GHG budgets of these sites and is important for GHG accounting protocols and verification. We hypothesized the higher CH₄ emissions from the flooded sites could result in these ecosystems being net GHG sources despite their potential for C sequestration.

Materials and methods

Study sites

The locations and overall characteristics of the five sites in this study are described in Table 1. All sites are located in the Sacramento-San Joaquin Delta of California, which is roughly 100 km inland from the Pacific Ocean (Fig. 1). The region experiences a Mediterranean climate, with hot, dry summers and cool, wet winters. The growing season typically extends from February to November. The 30-year mean air temperature (1981–2010) recorded at a nearby climate station (Antioch, CA, USA) is 16.4 °C and mean annual precipitation is 335 mm.

Conventional drained agricultural sites: The two business-as-usual land-use types are a grazed degraded peatland pasture on Sherman Island (Pasture) and a cornfield on Twitchell Island (Corn). Flux measurements at the Pasture began in April 2007. Two invasive plants make up the dominant cover types in the pasture: from December–April the canopy is dominated by mouse barley (*Hordeum murinum* L.), a long-naturalized C₃ grass in this region, and from April–October the canopy is dominated by pepperweed (*Lepidium latifolium* L.), a perennial forb. This site (~0.9 km × ~0.4 km), which has been a pasture for over 20 years, is fenced and grazed year-round

by ~100 cattle. The cattle tend to congregate in the far end of the field opposite the flux tower during the day in the summer months, however, they commonly pass by the tower in the evening and during the winter months. Their presence in the flux footprint notably impacts CH₄ fluxes and to a much lesser extent CO₂ fluxes (Detto *et al.*, 2010; Baldocchi *et al.*, 2012). The site is located on degraded peat soil, where the upper 0.6 m of soil is silt loam that overlays a deep peat layer (Hatala *et al.*, 2012b). The water table is largely maintained below the soil surface throughout the year by continual pumping. While the drained portion of the Pasture is a small CH₄ source (Baldocchi *et al.*, 2012), strong sources of CH₄ exist from cattle, flooded depressions in the field and drainage ditches (Teh *et al.*, 2011; Baldocchi *et al.*, 2012). The typical daytime flux footprint is confined to the well-drained portions of the field, but the elongated nighttime flux footprint crosses drainage ditches and wetter portions of the pasture that are hot-spots for CH₄ production (Detto *et al.*, 2010; Baldocchi *et al.*, 2012).

EC measurements at the cornfield were made from May 2012 to May 2013. The site (~1 km × ~1.2 km) is also on degraded peat soil. The water table is kept below the crop-rooting zone by an actively managed drainage network. During this study, the field was planted May 20–21, 2012 and was harvested November 1–8, 2012. The field remained fallow during winter. The variety of corn planted was ES-7477 hybrid corn commercialized by Eureka seeds, and the field was fertilized once at seeding with 118 kg N ha⁻¹ in the form of urea ammonium nitration solution (UAN 32%).

Agricultural wetland site: The rice paddy on Twitchell Island (Rice) represents the agricultural wetland land-use class. Micrometeorological measurements at this site began in April 2009 upon conversion from traditional corn and alfalfa agriculture. The rice site is a pilot project managed by the California Department of Water Resources (CADWR) to assess the potential of growing rice in the Delta. Prior to 1990, rice was not farmed in the Delta due to cool nighttime growing season temperatures. However, with the development of new varieties capable of withstanding these conditions, currently there are ~20 km² of rice farmed in the Delta. The field where this site is located is ~0.55 km × 0.7 km. The field was flooded for more than half the year, however, it was drained several times during the year for cultivation and planting, fertilizer and herbicide application, and harvest. Due to late precipitation in winter 2012 the field was plowed, harrowed, and leveled for planting in early May, which is about a month later than previous years (Hatala *et al.*, 2012b). The rice variety M104, a cold weather cultivar, was planted on May 17, 2012. The field was fertilized with 11-52-0 mono-ammonium phosphate fertilizer at a rate of 68 kg acre⁻¹ during planting and then again with 30-0-20 ammonium sulfate fertilizer on June 16, 2012 at a rate of 68 kg acre⁻¹. The rice field was first treated with herbicide in mid-June (3.65 g acre⁻¹ Regiment, 324 g acre⁻¹ Prowl, 32 g acre⁻¹ SYL-TAC, 324 g acre⁻¹ UN-32, and 4.1 g acre⁻¹ Sandea), and then again with Propanil Flowable Herbicide (i.e. SuperWham!) in mid-July to control a weed infestation. The crop was harvested November 13–16, 2012. Following harvest of the rice grains, the remaining plant residue was left on

top of the soil, and the field was re-flooded for the following winter to provide habitat for migrating birds.

Restored wetland sites: CO₂ and CH₄ fluxes were also measured at a long-term restored wetland (Old wetland) and newly restored wetland (Young wetland). In 1997, the Old wetland (0.028 km²) was constructed in the central part of Twitchell Island (Miller *et al.*, 2008; Miller, 2011). We began GHG measurements at this site in July 2012. The wetland was built on a former agricultural field by excavating surface soil, which was used to construct berms around the area excavated for the wetland. *Schoenoplectus acutus* (tule) shoots and rhizomes were planted in the eastern portion of the site prior to flooding, and cattails (*Typha latifolia*, *T. domingensis*, and *T. angustifolia*) from adjacent waterways were allowed to disperse naturally to the pond via windborne seeds. In October 1997, the sites was flooded to a constant water depth of ~25 cm. Following flooding, the site was rapidly colonized by cattails, and presently approximately 100% of the pond is filled with emergent macrophytes. Several floating aquatic plants (*Ludwigia peploides* and *Lemna* sp.) and submerged aquatics are also present at the site.

A considerably larger (1.21 km²) restored wetland was constructed in 2010 on a drained peatland pasture on Sherman Island (Young wetland), with flux measurements initiated shortly after (October, 2010). During wetland construction, a heterogeneous bathymetry was excavated to preserve existing wetland vegetation and generate regions of shallow water (a few centimeters) and adjoining areas of deeper water (up to 2 m). Consequently, this site is spatially heterogeneous, consisting of a mix of open water and vegetation patches, where the abundance and spatial aggregation of the vegetation strongly affect CH₄ fluxes at this site (Matthes *et al.*, 2014). Today, this mixed configuration presents a contrast with the Old wetland, where the site and flux footprint is much more homogenous and dominated entirely by emergent vegetation. Both wetlands support both *Schoenoplectus acutus* and *Typha* spp., but differ in relative dominance between them.

Eddy covariance measurements

We employed the EC method to measure 30-min fluxes of CO₂ (NEE; $\mu\text{mol m}^{-2} \text{s}^{-1}$), CH₄ ($\text{nmol m}^{-2} \text{s}^{-1}$), latent heat (LE; W m^{-2}), and sensible heat (H; W m^{-2}). A similar set of EC instrumentation was deployed at each site (Table 2). At each tower, a sonic anemometer measured high frequency wind velocity in three coordinates (u , v , w ; m s^{-1}) and temperature (T_{sonic}) (Table 2). Fluctuations in CO₂ and H₂O molar density (ρ_{CO_2} and $\rho_{\text{H}_2\text{O}}$) were measured with open-path infrared gas analyzers (Table 2). Open-path CH₄ sensors, based on wavelength modulation spectroscopy, were used to measure fluctuations in CH₄ molar density (ρ_{CH_4}), with the exception of the Pasture, where CH₄ mixing ratio (χ_{CH_4}) was measured with a closed-path tunable diode laser fast methane analyzer (FMA). AC power was available at the Pasture, therefore we used a scroll pump (BOC ESDP 30A, Edwards, Tewksbury, MA, USA) which requires 770 W of power and provides a flow rate of ~40 l min⁻¹ at the FMA cell pressure (19 kPa). Extensive field testing was conducted to evaluate the performance of the FMA sensor at this site (Detto *et al.*, 2010) and comparisons between the magnitude of 30-min fluxes showed good agreement between open-path and closed-path flux systems (Detto *et al.*, 2011). CH₄ fluxes were not measured at the Corn as CH₄ emissions were assumed to be negligible from this site since the water table was well below the soil surface throughout the study. We used either digital dataloggers systems (LI-7550A; LI-COR Biogeosciences, Lincoln NE, USA) or Campbell CR1000 dataloggers (Campbell Scientific, Logan, UT, USA) to record raw turbulence data at 10 Hz.

Fluxes were calculated using the 30-min covariance of vertical wind speed (w) and the appropriate scalar after applying a series of standard corrections using in-house software (Detto *et al.*, 2010; Hatala *et al.*, 2012b). First, this software removed artificial spikes in the 10 Hz data and diagnostic instrument values that corresponded to poor readings, which were primarily associated with precipitation or fog events. Next, coordinate rotations were used to align the streamlines with the surface of each site resulting in zero mean w and v within each

Table 2 Description of the eddy covariance flux measurement systems and analysis

| | Conventional drained agricultural sites | | Agriculture wetland site | Restored wetland sites | |
|---|---|--------------------|--------------------------|------------------------|-------------|
| | Pasture | Corn | | Young wetland | Old wetland |
| Eddy covariance height (m) | 3.2 | 5.15 | 3.25 | 3.7 | 4.64 |
| Sonic anemometer, | WMP 1352 | WMP 1352 | WMP 1352 | WM 1590 | WM 1590 |
| CO ₂ analyzer | LI-7500 | LI-7500 | LI-7500 | LI-7500 | LI-7500 |
| and CH ₄ analyzer | FMA | N/A | LI-7700 | LI-7700 | LI-7700 |
| Threshold friction velocity (m s^{-1}) | 0.15–0.20 | 0.15 | 0.15 | 0.15–0.185 | 0.20 |
| Filtered wind directions | N/A | 180–200° & 355–10° | 0–180° | 115–120° | 290–250° |
| Percentage of observations rejected* | 26–31% | 36–42% | 40–48% | 20–34% | 54–55% |
| Energy balance closure | 0.91 | 0.76 | 0.81 | 1.14 | 0.72 |

*The % of observation rejected varies depending on which flux measurement is considered.

N/A, not applicable.

30-min block. Where open-path sensors were used, the Webb-Pearman-Leuning correction was applied to account for the effects of air density fluctuations (Webb *et al.*, 1980; Detto & Katul, 2007), and the relevant additional spectroscopic corrections for ρ_{CH_4} fluctuations measurements with the LI-7700 instrumentation were also applied (Mcdermitt *et al.*, 2011). For the closed-path sensor, CH₄ measurements were adjusted to eliminate air density variations due only to the effects of water vapor since it is assumed that high frequency temperature fluctuations were dampened when sampling through the tube (Detto & Katul, 2007; Detto *et al.*, 2010). Fluctuations in T_{sonic} were calculated from fluctuations in the speed of sound following crosswind and humidity corrections (Schotanus *et al.*, 1983; Kaimal & Gaynor, 1991). After calculating the fluxes, we filtered flux values with low friction velocity (u_*) to constrain our analysis to turbulent conditions. Friction velocity thresholds (Table 2), which varied seasonally, were identified as values above which nighttime NEE no longer varied with increasing u_* . Fluxes were further filtered for spikes in 30-min mean densities, variances and covariances with thresholds varying seasonally and between sites. Lastly, fluxes from wind directions outside the footprint of each site were filtered from the data set and omitted from this analysis (Table 2). The percentage of 30-min fluxes excluded from this analysis is given in Table 2. Additionally, for the closed-path CH₄ measurements, we used a procedure similar to the one outlined in Aubinet *et al.* (1999) and Humphreys (2004) to correct for the high frequency flux losses resulting from sensor separation and attenuation of fluctuations of χ_{CH_4} down the sampling tube. No high frequency corrections were applied to fluxes measured using open-path sensors since flux losses amounted to less than 5%, which is well within the accuracy of an individual flux measurement (Aubinet *et al.*, 1999).

Energy balance closure at these sites, defined as the energy balance ratio (Wilson *et al.*, 2002), ranged between 0.72 and 1.24 (Table 2). These values fall within the range generally observed at sites within the FLUXNET network (Wilson *et al.*, 2002; Stoy *et al.*, 2013). At all sites we accounted for radiant energy absorbed in photosynthesis as in Leuning *et al.* (2012), and we approximated heat storage in the water column at the Rice and Old wetland as in Drexler *et al.* (2004) since water depth was relatively uniform across these sites. At the Young wetland, an EB closure greater than 100% is likely related to the difficulty in adequately estimating net radiation (R_{net}) and storage terms at this site where the distribution of water and vegetation is spatially heterogeneous.

Gap-filling, NEE partitioning, and annual budget computation

The artificial neural network (ANN) technique was used to gap-fill half-hourly fluxes with meteorological variables (Papale *et al.*, 2006; Moffat *et al.*, 2007). The ANN routine was optimized for both representativeness and generalizability. To avoid the ANN being biased toward environmental conditions that typically have better data coverage such as summertime and daytime measurements, the explanatory data were divided into natural data clusters using a k-means clustering

algorithm (Mathworks Inc., 2012). Data used to train, test, and validate the ANN were proportionately sampled from these clusters. For generalizability, the simplest ANN architecture with good performance (<5% gain in model accuracy for additional increases in architecture complexity) was chosen for 20 extractions of the training, test, and validation data. Within each extraction, each tested ANN architecture was reinitialized 10 times, and the initialization with the lowest root-mean-square-error was selected to avoid local minima. When the optimum ANN architecture for each extraction was determined, the resultant prediction was saved. The median of the 20 predictions was used to fill each gap and the spread of the predictions was used to provide a measure of uncertainty resulting for the ANN gap-filling procedure. The only variable that was not gap-filled was CH₄ flux at the Pasture, as extra precaution is needed to interpret eddy flux CH₄ measurements in intensively grazed pastures (Baldocchi *et al.*, 2012). CH₄ fluxes at this site were strongly influenced by the combined effects of CH₄ emissions from cattle, and by the collapsed nocturnal boundary layer and elongation of the flux footprint over CH₄ hot spots (Baldocchi *et al.*, 2012). Rather than integrating all CH₄ flux measurements we used the method described in Baldocchi *et al.* (2012) to estimate conditional averages, which we used to bound the annual CH₄ budgets. This resulted in a lower bound that is representative of CH₄ emitted from the drained portion of the pasture with less influence from cows, and an upper bound that is representative of CH₄ emissions from the drained and wet portions of the field and includes CH₄ emitted by cows.

The EC method measures the net exchange of CO₂ between the land surface and the atmosphere, which represents the sum of ecosystem photosynthesis (gross ecosystem production; GEP) and ecosystem respiration (ER). NEE can be partitioned into GEP and ER by separately considering the day and night observations, as photosynthesis only occurs during daylight hours. Winds in the Delta are strong even during the night (Hatala *et al.*, 2012b), thereby minimizing the need to account for the uncertainties related to nighttime EC measurements due to atmospheric stratification and stability (Massman & Lee, 2002). Consequently, NEE was partitioned into GEP and ER using a method similar to Reichstein *et al.* (2005). Briefly, an Arrhenius-type model after Lloyd & Taylor (1994) is used to describe the temperature dependence of nighttime NEE (i.e. ER since GEP is assumed to be zero at night), and this model is then extrapolated to daytime periods. GEP was then calculated as the difference between NEE and ER.

We computed annual budgets by integrating the gap-filled and partitioned fluxes over the course of a full year. The dates over which the annual sums were calculated are given in Table 1 as they are not the same for all sites due to differences in the timing of the deployment of the sites. The net C balance was estimated as the annual sum of NEE after accounting for the loss of C from the system via CH₄ emissions and harvest. As such, the C balance reflects the net ecosystem C balance as opposed to the C balance from an atmospheric perspective as measured by the EC method (Chapin *et al.*, 2006). The net GHG budget was derived from annual sums of CO₂ and CH₄, assuming that 1 g CH₄ is equivalent to 25 g CO₂ with respect

to the greenhouse effect over a time horizon of 100 years, and we assumed that C removed from the sites through harvest would eventually be released to the atmosphere as CO₂. The GHG budgets did not account for secondary emissions at the sites such as emissions related to pumping water off the islands or from farming machinery due to the difficulties in constraining these values. To estimate the amount of uncertainty in the ANN gap-filling procedure for the annual budget of each scalar, we calculated the integrated annual budget using the full range of predictions used to fill each gap (i.e. based on the 20 extractions from the ANN), and calculated the 95% confidence interval from the distribution of the 20 annual budgets. In this study, fluxes toward the surface are negative and fluxes away from the surface are positive, therefore negative NEE represents net CO₂ uptake and positive NEE indicates a net CO₂ source.

Supporting measurements

Micrometeorological instrumentation was deployed at each site to accompany EC measurements. Air temperature (T_{air}) and relative humidity were measured with an aspirated and shielded thermistor and capacitance sensor (HMP45C or HMP60; Vaisala, Vantaa, Finland). Precipitation was measured at the Pasture and Corn with a tipping bucket rain gauge (TR-525I or TR-525M; Texas Electronics Inc., Dallas, TX, USA) and water table depth was measured using pressure transducers (CS450 or CS451; Campbell Scientific, Logan, UT, USA or

PDCR 1830; GE Druck, Billerica, MA, USA) at all of the sites except the Pasture where measurements were made manually from a well during field visits. R_{net} was measured with a four-component net radiometer (CNR1; Kipp and Zonen, Delft, Netherlands or NR01; Hukseflux, Delft, The Netherlands), except at the Rice and Young wetland where it was measured with a double-sided net radiometer (NR Lite; Kipp and Zonen, Delft, The Netherlands). Quantum sensors were used to measure incoming and outgoing photosynthetically active radiation (PAR) (PAR-LITE or PQS 1; Kipp and Zonen). Ground heat flux at the Pasture, Corn, and Rice was measured as the average of three replicate ground heat flux plates (HFP01 or HFP01SC; Hukseflux Thermal Sensors, Delft, The Netherlands) buried just below the soil surface at each of these sites. At all sites we measured soil temperature (T_s) at soil depths of -0.02 , -0.04 , -0.08 , -0.16 , and -0.32 m with copper constant thermocouples. We report the mean of three sample replicates at each depth. Similarly, water temperatures were measured just above the soil surface at the Rice, 0.02 and 0.04 m at the Young wetland, and 0.04 , 0.8 , and 0.16 m at the Old wetland. All supporting measurements were sampled every 10 s, and the 30 min average values were stored on CR10X, CR23X, or CR1000 dataloggers (Campbell Scientific, Logan, UT, USA).

Vegetation sampling and monitoring

Plant area index (PAI) was measured every 1–2 weeks at the Pasture, Corn, and Rice sites during the growing season using

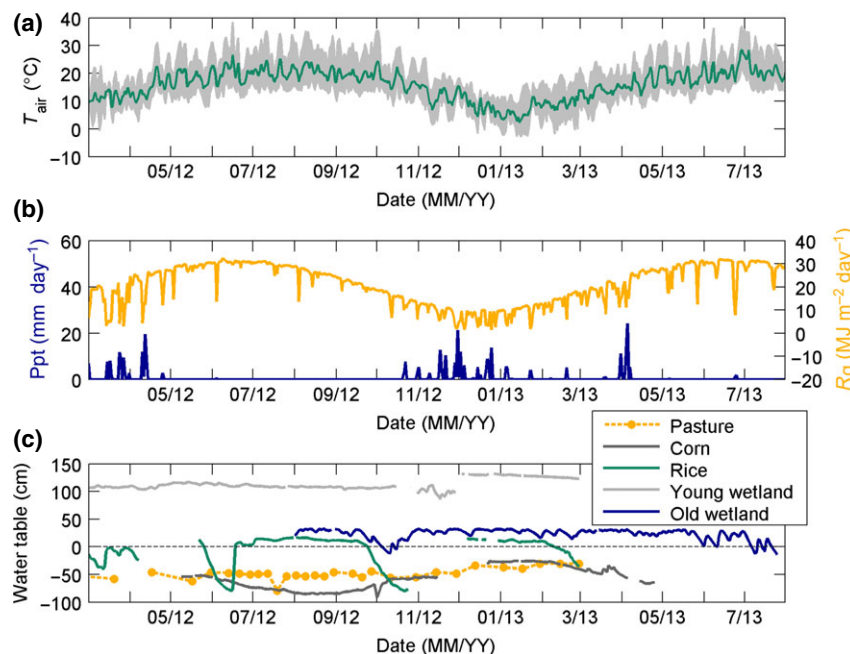


Fig. 2 Typical pattern of the Mediterranean climate experienced at the sites, which is characterized by warm summers and cool winters (T_{air}) (a), and high incoming radiation (R_g) and low precipitation (Ppt) during the summer months and wet winters (b), and differences in water table management across sites (c). Measurements in (a) and (b) are only plotted for the peatland pasture since the values for the other sites are almost identical. The gray shaded area in (a) bounds the minimum and maximum daily air temperature and the dash-dot line in (c) reflects the fact that measurements were made manually during weekly or biweekly field visits.

an LAI-2000 Plant Canopy Analyzer (LI-COR Biogeosciences, Lincoln, NE, USA). Measurements were made every 10 m along a 100 m transect. At the Corn and Rice, destructive measurements of aboveground biomass were also made by clipping all vegetation within five randomly sampled 400 cm² plots. At all sites, canopy phenology was monitored using digital camera images. Digital cameras were mounted near the top of each flux tower to record images year-round. JPEG images with red, green, and blue channels were recorded every 30 minutes. An analysis of a predefined 'region of interest' (ROI) in the foreground of each image was conducted using the PhenoCam GUI application available as a precompiled MATLAB® program (PhenocamGUI v1.1; <http://phenocam.sr.unh.edu/webcam/tools/>) to calculate a normalized green channel brightness (% Green) for the ROIs as in Richardson *et al.* (2007). The % Green index was selected to monitor canopy phenology since it reveals clear seasonal patterns that can be associated with canopy development and senescence and is a good descriptor of NDVI (Richardson *et al.*, 2007; Migliavacca *et al.*, 2011; Westergaard-Nielsen *et al.*, 2013).

Results

Weather conditions, water table management, and phenology

As a result of the Delta's Mediterranean climate, trends in meteorological variables at the sites followed a strong seasonal cycle (Fig. 2a and b). Mean

annual T_{air} was similar across sites despite somewhat different measurement periods (Table 1), and values were just below the 30-year (1981–2010) mean of 16.4 °C from the Antioch Climate Station located 10 km from the Pasture site. Annual precipitation at the sites ranged from 263 to 390 mm (Table 1), with a mean of 322 mm across sites that is comparable to the 30-year average of 335 mm. Total incoming radiation ranged between 7002 and 7139 MJ m⁻² yr⁻¹ with differences between sites attributable to differences in measurement periods. T_s at the flooded sites were lower than at the drained sites, particularly during the peak growing season ($T_{s,\text{grow}}$) when the rice and wetland canopies were closed (Table 1). T_s and $T_{s,\text{grow}}$ were lowest at the Old wetland which was due to the tall, dense, and closed canopy at this site.

Water table depth at the Pasture and Corn was maintained below the soil surface throughout the year by continual pumping (Fig. 2c). Nonetheless, at the Pasture CH₄ is produced within flooded drainage ditches and in zones of saturation in the soil profile (Teh *et al.*, 2011). Water levels at the Rice were regulated to ~0.05 to 0.10 m above the soil surface for just over half the year, but the field was drained for about 80 days (March 1–May 19, 2012) for cultivation and planting, approximately 20 days

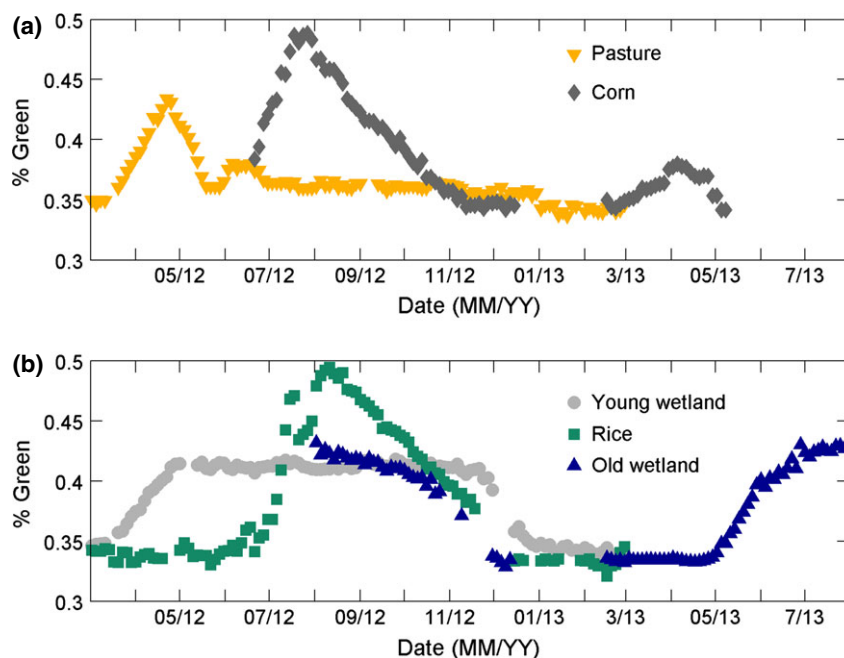


Fig. 3 Time series of relative green brightness (% Green) for the conventional drained agricultural sites (a), and the agricultural and restored wetland sites (b). The pattern of % Green generally reaches a maximum at all sites during the summer months, except at the Pasture. This index does not appropriately reflect vegetation cover at this site as the greenness index decreases during this time despite high cover, as the pepperweed's small white flowers cause a decrease in site greenness. It is also clear from the seasonality of greenness index that while the crops are highly productive, their growing season is much shorter than that of the restored wetlands.

(May 28–June 19, 2012) for fertilizer and herbicide application, about 50 days (September 30–November 21, 2012) for harvest, and then again for another 20 days when the field was prepared for planting the following spring (February 10–March 1, 2013) (Fig. 2c). Water table depth at Young wetland was always well above the soil surface, and while the Old wetland was generally flooded to a depth of ~25 cm, accidental drops in the water table to or below the soil surface occurred periodically due to mechanical problems with the pumps.

As indicated from the % Green index derived from the digital camera images, the restored wetlands had a much longer growing season than the crops (Fig. 3); the wetlands began greening up as early as mid-March and canopy senescence did not begin until November, whereas the growing season at the Rice and Corn sites only extended from June to October. The secondary peak in % Green at the Corn site in April was due to an invasion by grasses in spring. Peak % Green occurred in the summer months, with the exception of the Pasture where the index decreased during this time as the pepperweed's small white flowers masked the true 'greenness' of the site (Sonnentag *et al.*, 2011).

Temporal variability in CO₂ fluxes and annual budgets

The general pattern of NEE at all sites followed a similar seasonal cycle (Fig. 4) with most photosynthesis

occurring in spring and summer when incoming solar radiation was greatest (Fig. 5). The highest rate of net CO₂ uptake at the Pasture (~–6 g C m^{–2} d^{–1}) occurred in late spring, corresponding with pepperweed growth. Despite the lack of precipitation and low soil moisture during this period, plant growth was possible since the pepperweed can tap the shallow water table. Low rates of photosynthesis during the rainy season were due to the presence of winter grasses at this site (Fig. 5), which grow slowly over this time period due to low temperatures. Although peak GEP was greatest at the Corn due to canopy architecture and C₄-type photosynthesis, peak net CO₂ uptake (~–5 g C m^{–2} d^{–1}) was lower than at the Pasture as a result of higher rates of ER (Fig. 5). On an annual timescale, GEP at the Pasture was greater than at the Corn, since although the Corn had higher rates of growing season photosynthesis, year-round growth at the Pasture compensated for the lower canopy photosynthetic capacity. The highest rates of net CO₂ release at the drained sites corresponded to the return of the winter rains, and reached values up to 10 g C m^{–2} d^{–1} during this period. The increase in ER following the start of the rainy season occurs as moisture at the surface reactivates microbial activity resulting in large pulses of CO₂ emissions following the first rains (Huxman *et al.*, 2004; Ma *et al.*, 2012). Although budgets of GEP at the Corn and Pasture were largely comparable to those at the flooded sites, the drained land-use types were net sources of

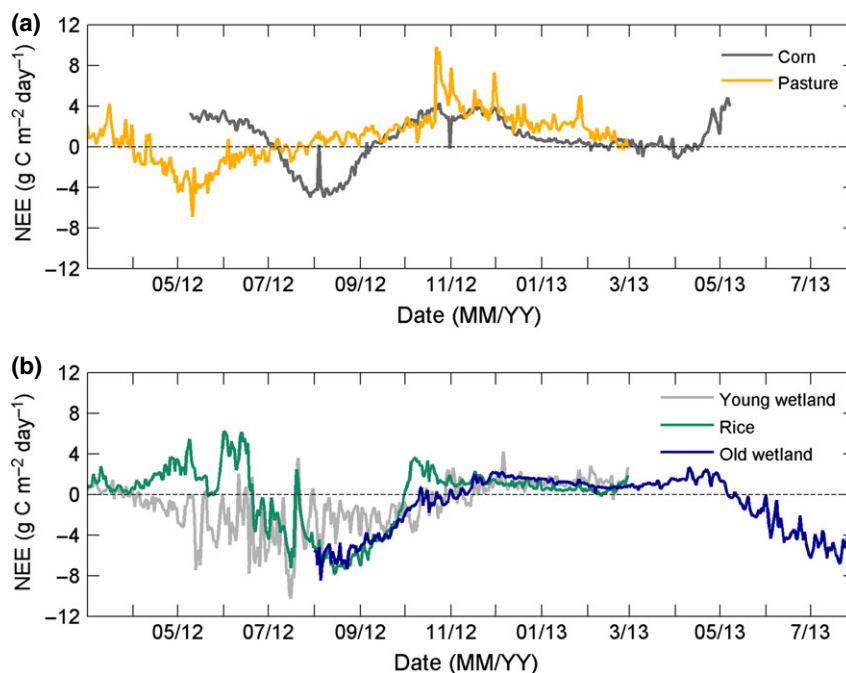


Fig. 4 Seasonal variability in net ecosystem CO₂ exchange (NEE) for the conventional agricultural sites (a), and the agricultural and restored wetland sites (b). Net ecosystem exchange at all sites follows a similar seasonal cycle with peak carbon uptake in the spring and summer and net carbon emissions in the wintertime.

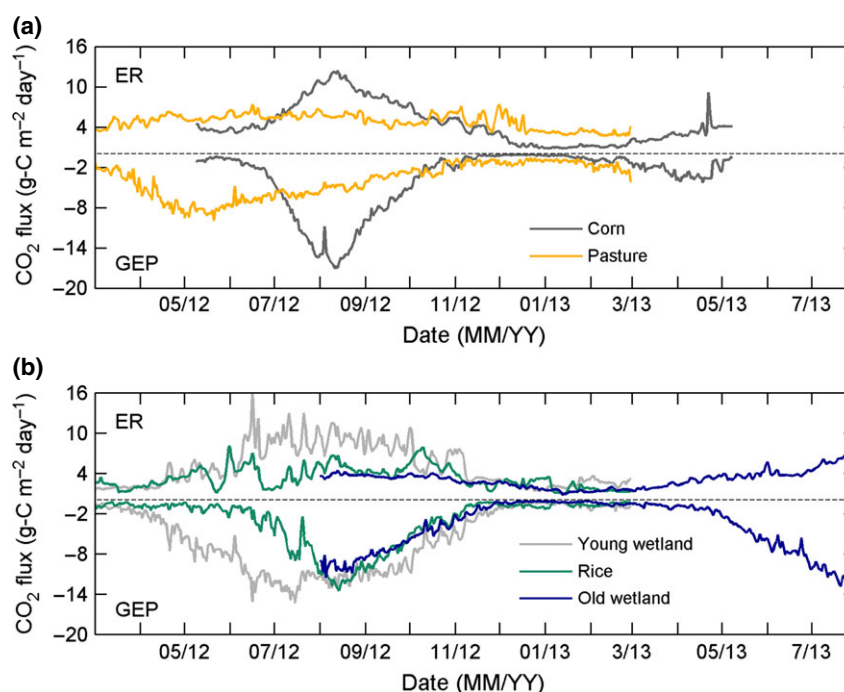


Fig. 5 Daily partitioned net ecosystem exchange for the conventional agricultural sites (a), and the agricultural and restored wetland sites (b). Peak gross ecosystem production (GEP) occurs during the summer growing season. Flooding and cool temperatures inhibit wintertime ecosystem respiration (ER) at the rice paddy and restored wetlands, whereas autumn rains at the drained sites stimulate ER and drainage at the Rice for planting and harvest cause large pulses of CO₂ to the atmosphere.

CO₂ to the atmosphere on an annual basis due to high rates of ER (Table 3). The Pasture was the largest net source of CO₂ to the atmosphere, releasing 341 g C m⁻² when integrated over a year, while the Corn was a slightly smaller net CO₂ source (278 g C m⁻² yr⁻¹).

Growing season NEE at the flooded sites was quite similar with maximum net CO₂ uptake ranging between -8 and -10 g C m⁻² d⁻¹ (Fig. 4). Wintertime NEE at these sites rarely exceeded 2 g C m⁻² d⁻¹ since flooding and cool temperatures inhibited ER (Fig. 5). The periods

of increased net CO₂ emissions at the Rice in spring and fall coincided with drainage events that resulted in large pulses of CO₂ to the atmosphere, which are attributable a combination of degassing due to reduced hydrostatic pressure and increased ER. On an annual basis the flooded sites were strong to weak net CO₂ sinks due largely to lower rates of ER (Table 3). Flooding resulted in comparably low annual sums of ER at the Rice and Old wetland, but despite similar rates of growing season photosynthesis (Fig. 5), yearly GEP was considerably

Table 3 Annual sums of net and partitioned CO₂ fluxes, CH₄ fluxes, harvest, and total ecosystem carbon and greenhouse gas budgets. Error bounds reflect the 95% confidence interval for the gap-filling procedure. Note that there are no error bounds for ecosystem respiration since it is modeled based on the relationship between nighttime net CO₂ exchange and air temperature and is independent of the gap-filling procedure

| Site | NEE g C m ⁻² | GEP g C m ⁻² | ER g C m ⁻² | CH ₄ g C m ⁻² | g CO ₂ eq m ⁻² | Harvest g C m ⁻² | C budget g C m ⁻² | GHG budget g CO ₂ eq m ⁻² |
|---------------|----------------------------|----------------------------|---------------------------|--|--------------------------------------|--------------------------------|---------------------------------|--|
| Pasture* | 341±73 | -1438±10 | 1762 | 5.84±1.51 to 11.4±2.66 | 194±51 to 381±88 | N/A | 347±75 to 352±76 | 1444±319 to 1631±356 |
| Corn | 278±24 | -1356±9 | 1619 | N/A | N/A | 293 | 571±24 | 2094±88 |
| Rice | -50±76 | -1159±14 | 1203 | 5.30±0.80 | 177±27 | 162 | 117±77 | 588±306 |
| Young wetland | -368±46 | -2106±16 | 1834 | 53.0±0.78 | 1769±26 | N/A | -315±47 | 420±194 |
| Old wetland | -397±20 | -1506±7 | 1108 | 38.7±1.10 | 1293±36 | N/A | -358±21 | -162±109 |

*The upper and lower bounds for the CH₄ budget at the Pasture are representative of different field conditions. Additional details are given in the body of the article.

N/A, not applicable.

greater at the Old wetland due to its longer growing season (Fig. 3). Consequently, the Old wetland was a much larger annual net CO₂ sink, capturing 397 g C m⁻² yr⁻¹, whereas the rice paddy only captured 50 g C m⁻² yr⁻¹ as CO₂ from the atmosphere. While NEE budgets at the Old and Young wetlands differed by less than 40 g C m⁻², the partitioning of NEE into ER and GEP differed notably between sites (Table 3); although the Young wetland captured more CO₂ through photosynthesis than the Old wetland, this was offset much higher rates of ER, making the net CO₂ budget between these sites comparable. The higher productivity at the Young wetland compared to the Old wetland was likely due to the rapid expansion of new vegetation during the 2012 growing season as the Young wetland continues to fill in over time, and high rates of respiration may be attributable to higher GEP as ecosystem respiration scales with productivity (Janssens *et al.*, 2001).

Seasonal course of CH₄ fluxes and annual budgets

Large differences in both the magnitude and seasonal pattern of CH₄ fluxes were observed across sites (Fig. 6). CH₄ emissions at the Pasture were a mixture of fluxes from lower CH₄-emitting upland

soils, high CH₄-emitting drainage ditches, and CH₄ emitted by cattle (Teh *et al.*, 2011; Baldocchi *et al.*, 2012). CH₄ fluxes during the dry season generally ranged between 0 and 50 mg C m⁻² d⁻¹. Greater CH₄ effluxes were observed during the rainy period, when daily CH₄ fluxes ranged between 10 and 330 mg C m⁻² d⁻¹. We constrained the annual CH₄ budget at the Pasture to range between 5.84 and 11.4 g C m⁻² yr⁻¹, with the lower end representative of the drained portion of the field with little influence from the cows and the upper end comprising both the drained and wet portions of the field and greater influence from the cattle.

The CH₄ fluxes at the Rice were comparable in magnitude to those at the Pasture, but followed a different seasonal pattern (Fig. 6a): CH₄ emissions at the Rice largely tracked GEP (Fig. 5), and large pulses of CH₄ to the atmosphere occurred during drainage of the field in late September 2012 and in mid-February 2013 (Fig. 2c). The release of CH₄ upon drainage can be attributed to a combination of degassing due to reduced hydrostatic pressure and to decreased CH₄ oxidation due to the more rapid transport of CH₄ through the soil profile (Han *et al.*, 2005a; Hatala *et al.*, 2012b). Once the field was well-drained, CH₄ fluxes were quite low, rarely exceed-

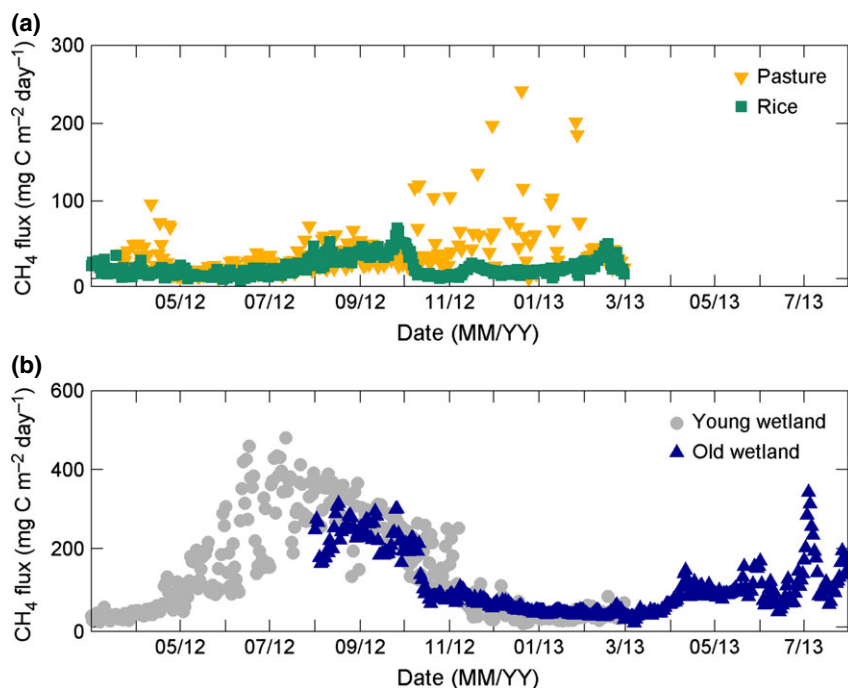


Fig. 6 Seasonal course of daily integrated CH₄ flux for the drained and flooded agricultural sites (a) and restored wetland sites (b). CH₄ fluxes were generally low at the rice paddy and pasture sites, while fluxes at the restored wetlands were up to an order of magnitude greater [note the difference in the vertical axis in (a) and (b)]. With the exception of the pasture, CH₄ emissions largely followed a pattern that was closely related to that of gross ecosystem production. CH₄ emissions were also strongly influenced by water table dynamics. Note that CH₄ fluxes at all sites were gap-filled with the exception of the Pasture, therefore integrated daily CH₄ emissions at this site were only estimated for days when there was a measured flux for at least half of the possible 48 half-hour intervals.

ing 15 mg C m⁻² d⁻¹. Over the course of a year, the rice paddy released 5.3 g C m⁻² as CH₄ to the atmosphere.

Growing season CH₄ emissions at the restored wetlands were an order of magnitude greater than those at the Rice or Pasture, with peak emissions of 480 mg C m⁻² d⁻¹ at the Young wetland and 350 mg C m⁻² d⁻¹ at the Old wetland. Again CH₄ emissions followed a seasonal pattern that largely paralleled GEP, with the exception of the pronounced peak in daily CH₄ flux at the Old wetland in early July 2013, which was the result of a sudden water table drawdown (Fig. 2c). On a yearly basis, the Young wetland released the largest amount of CH₄ to the atmosphere (53 g C m⁻²), while the Old wetland released somewhat less CH₄ (38.7 g C m⁻²) (Table 3).

Annual C and GHG budgets

As expected from the high rates of subsidence on Delta islands, conventional agricultural sites were both large net C and GHG sources (Table 3). While the Pasture was the largest annual CO₂ source to the atmosphere (Table 3), if we account for the fact that 293 g C m⁻² was removed from the cornfield through harvest, total C loss was greatest at the Corn site (571 g C m⁻² yr⁻¹). The drained sites were GHG sources of 1444 and 2094 g CO₂eq m⁻² yr⁻¹, which is considerably greater than GHG emissions from the flooded land-use types.

Although the Rice was a small atmospheric sink for CO₂ (Table 3), if we consider the amount of C removed from the field through harvest (162 g C m⁻²) and CH₄ emissions, the rice paddy acted as both a net C and GHG source (117 g C m⁻² yr⁻¹ and 588 g CO₂eq m⁻² yr⁻¹ respectively). The restored wetlands could be either a GHG source or sink depending on CH₄ emissions: while NEE at the wetlands was comparable, higher CH₄ fluxes at the Young wetland relative to the Old wetland resulted in the former site being a net GHG source (Table 3). Regardless of the differences in GHG flux totals, the restored wetlands were both strong C sinks, sequestering between 315 and 358 g C m⁻² yr⁻¹.

Discussion

CO₂ fluxes

Due to high rates of soil subsidence, the practice of drained agriculture in the Delta is unsustainable in the long-term. Like other drained and degraded peatlands (e.g. Nieveen *et al.*, 2005; Veenendaal *et al.*, 2007; Hirano *et al.*, 2012), the Pasture was a large net source of CO₂ to the atmosphere (Table 3), as reported for previous years (Hatala *et al.*, 2012b). Corn is generally a C

sink or C neutral, even when considering harvest (Suyker *et al.*, 2004; Bernacchi *et al.*, 2005; Hollinger *et al.*, 2005), however, high rates of peat oxidation resulted in the Corn being a large net CO₂ source.

The results of this study showed that converting drained Delta landscapes back to flooded conditions offers a promising intervention to halt C loss and associated subsidence. Rice agriculture is one possible flooded land-use type that can slow subsidence by limiting ER, as the rice paddy was an atmospheric sink for CO₂ (Table 3), in agreement with earlier studies at this site (Hatala *et al.*, 2012b). The rice paddy was a weaker annual CO₂ sink than generally reported in other studies (e.g. Campbell *et al.*, 2001; Saito *et al.*, 2005; Mcmillan *et al.*, 2007), which could be due to warmer temperatures at sites that are more traditionally suited to rice agriculture.

Wetland restoration is the most promising management option for reversing subsidence in the Delta, as the restored marshes were the largest C sinks (Table 3). We found that NEE budgets at the Young and Old wetlands were over an order of magnitude greater than values reported for temperate and northern peatlands (e.g. Rinne *et al.*, 2007; Roulet *et al.*, 2007; Saarnio *et al.*, 2007; Nilsson *et al.*, 2008; Drewer *et al.*, 2010; Olson *et al.*, 2013), and nearly twice as large as values reported for other temperate *Typha* marshes (Bonneville *et al.*, 2008; Rocha & Goulden, 2008; Bernal & Mitsch, 2013). In contrast, Whiting & Chanton (2001) reported NEE budgets ranging from -896 to -1139 g C m⁻² yr⁻¹ for *Typha* marshes in the Southeast USA, which exceeds values at the Delta wetlands. However, these rates were estimated from monthly or bimonthly chamber measurements, and while the chamber method is ideal for assessing the spatial variability in fluxes, EC is more suitable for assessing temporal variability and up-scaling both spatially and temporally (Hendriks *et al.*, 2010). C sequestration rates in restored Delta marshes were also in the upper range of values reported in the literature for other wetland types (cf. Table 5 in Bernal & Mitsch, 2012; cf. Table 2 in Mitsch *et al.*, 2013). High rates of net CO₂ uptake at our sites are attributable to the Delta's long, warm growing season with abundant water and sunlight (Brinson *et al.*, 1981). A handful of studies have measured C budgets at other created or restored wetlands and reported net CO₂ uptake rates lower (Waddington *et al.*, 2010; Bernal & Mitsch, 2013; Herbst *et al.*, 2013) or within the range (Hendriks *et al.*, 2007; Badiou *et al.*, 2011) observed in this study.

CH₄ fluxes

While flooding is an effective means of sequestering C, it also has secondary effects on GHG budgets through

increased CH₄ production. CH₄ emissions at the Pasture were low relative to CH₄ fluxes at the restored wetlands (Table 3), but comparable to fluxes measured at other drained peatlands (Kroon *et al.*, 2010; Schrier-Uijl *et al.*, 2010). Despite the rice paddy being flooded for more than half the year, less CH₄ was emitted annually than at the Pasture (Table 3). Furthermore, CH₄ fluxes at the Rice were considerably lower than values measured from rice paddies elsewhere in California (Cicerone & Shetter, 1981; Cicerone *et al.*, 1983; Mcmillan *et al.*, 2007), and by other studies of CH₄ emissions from rice agriculture around the world (Seiler *et al.*, 1983; Holzapfel-Pschorn & Seiler, 1986; Huang *et al.*, 1997; Ding *et al.*, 1999; Meijide *et al.*, 2011; Bhattacharyya *et al.*, 2014). However, a study measuring CH₄ fluxes from irrigated rice fields in the Philippines using the EC technique observed similar daily mean CH₄ emissions as our study with values averaging between 25.4 mg C m⁻² d⁻¹ during the vegetative stage and 34.1 mg C m⁻² d⁻¹ during the reproductive stage, and seasonal CH₄ emissions of only 3.26 g C m⁻² which was attributed to intermittent irrigation during the vegetative stage (Alberto *et al.*, 2014). Other studies where rice paddies were intermittently drained during the growing season also showed CH₄ fluxes approaching the values reported in this study (Miyata *et al.*, 2000; Nishimura *et al.*, 2004; Han *et al.*, 2005b), as brief aerobic periods may re-oxidize reduced alternative electron acceptors, making methanogenesis energetically unfavorable upon reflooding until the sequence of more favorable electron acceptors for anaerobic metabolism have been depleted (Magonigal *et al.*, 2003). Lower CH₄ emissions in our study could be related to the lengthy drainage period prior to planting and drainage early in the vegetative stage for fertilizer and herbicide application (Alberto *et al.*, 2014), the presence of oxidizing agents such as iron, manganese or sulfate in the soil, lower productivity relative to other sites (Minoda & Kimura, 1994; Hatala *et al.*, 2012a), cooler soil temperatures (Schütz *et al.*, 1990; Conrad, 2002), differences in rice cultivar (Huang *et al.*, 1997), and the lack of labile organic substrate within the soil due to past land-use history (Hatala *et al.*, 2012b).

Numerous studies have shown a strong relationship between NEE or plant productivity and CH₄ flux, as vegetation is the primary source of C substrate for methanogenic metabolism (Chanton *et al.*, 1993; Whiting & Chanton, 1993; Whalen, 2005; Hatala *et al.*, 2012a). Since annual NEE at the restored wetlands was in the upper range of values reported in the literature, it follows that CH₄ emissions from these sites were also higher than values reported in other studies, particularly for more northern wetlands (e.g. Shurpali & Verma, 1998; Rinne *et al.*, 2007; Roulet *et al.*, 2007; Wille

et al., 2008; Drewer *et al.*, 2010; Jackowicz-Korczyński *et al.*, 2010; Olson *et al.*, 2013; Sun *et al.*, 2013; Turetsky *et al.*, 2014). Annual CH₄ budgets at a number of temperate and tropical wetlands exceeded values reported in this study, including emissions from marshes in the Midwestern USA (~60 g C m⁻² yr⁻¹) (Kim *et al.*, 1999; Nahlik & Mitsch, 2010), marshes in the Southwestern USA (up to 130 g C m⁻² yr⁻¹) (Whiting & Chanton, 2001), and tropical wetlands in Costa Rica (220–263 g C m⁻² yr⁻¹) (Nahlik & Mitsch, 2010). However, with the exception of the study by Kim *et al.* (1999), these emissions were estimated from chamber measurements taken at most once per month. Few studies have quantified CH₄ emissions from restored wetlands: mean annual CH₄ emissions ranged between 68 and 17 g C m⁻² yr⁻¹ for two long-term created marshes in Ohio, USA (Nahlik & Mitsch, 2010), between 31.3 and 32.3 g C m⁻² yr⁻¹ for a restored peatland in the Netherlands (Hendriks *et al.*, 2007), between 9 and 13 g C m⁻² yr⁻¹ from a restored wetland in Denmark (Herbst *et al.*, 2013).

Impacts of land-use change on C sequestration and GHG budgets

While the restoration of drained wetlands is often suggested as a means to sequester C (Drösler *et al.*, 2008; Maljanen *et al.*, 2010), there are few comprehensive studies that quantify the effects of restoration activities on C and GHG budgets. This study corroborates the findings of the few recent investigations that also found that while managed peatlands are large sources of C, rewetting can convert these ecosystems back to C sinks (Waddington *et al.*, 2010; Schrier-Uijl *et al.*, 2013). Although the Rice was a net C source from an ecosystem perspective, it lost 454–230 g C m⁻² yr⁻¹ less than the Corn and Pasture respectively. Therefore, while rice agriculture in the Delta does not reverse subsidence, it does experience subsidence rates up to an order of magnitude lower than rates for conventional drained agriculture (Hatala *et al.*, 2012b). Restored wetlands are the land-use type with the greatest potential to capture C (Table 3) and reverse subsidence (Miller *et al.*, 2008). Therefore, restoring drained and degraded peatlands to natural wetlands may be critical to ensuring the long-term sustainability of the Delta and other heavily subsided regions throughout the world (Armentano, 1980; Syvitski *et al.*, 2009).

In addition to stopping the degradation of peat soils and reversing subsidence, another goal of drained to flooded land-use change in the Delta is to reduce GHG emissions. This is part of a growing interest in California and elsewhere in wetland restoration and manage-

ment for C sequestration and possible inclusion in C finance markets (Emmert-Mattox *et al.*, 2010; Murray *et al.*, 2011). In addition to being large sources of C, the drained agricultural sites were significant GHG sources (Table 3). GHG balances for these sites represent conservative estimates as the budget of nitrogen GHGs was unaccounted for, and N₂O emissions at the Corn have been found exceed 400 g CO₂eq m⁻² yr⁻¹ (Morris, 2014) while even larger N₂O fluxes have been measured at the Pasture (Teh *et al.*, 2011). In all cases, flooding reduced GHG emissions (Table 3), even when factoring in N₂O emissions from the Rice, which are comparable to those at the Corn (Morris, 2014). Wetland restoration provides the most benefit from both a subsidence and GHG standpoint, with the potential of converting drained peatlands from GHG sources to GHG sinks (Table 3). Furthermore, N₂O emissions from natural wetlands are generally low compared with terrestrial soil environments due to the low rates of N₂O production (Page & Dalal, 2011). Additional benefits from wetland restoration include flood protection and the provision of key habitat for many species. While the EC method is a valuable tool to monitor trace gas fluxes from a range of land-use types (Baldocchi *et al.*, 2001), this study is unique in assessing the impacts of restoring drained and subsided agricultural peatlands to flooded ecosystems on CO₂ and CH₄ fluxes. Although this study only focused on observations from 2012 to 2013 as this was the only period where we had a full year of measurements from all sites, our conclusions are robust as the sign and relative magnitude of the C and GHG budgets are largely consistent between years (Baldocchi *et al.*, 2012; Hatala *et al.*, 2012b; Teh *et al.*, 2011; S.H. Knox, C. Sturtevant, L. Koteen, P. Oikawa, J. Verfaillie, D. Baldocchi, unpublished data). The only site that shows significant inter-annual variability in both the sign and magnitude of annual sums is the Young wetland since it is in the early stages of ecosystem development; nonetheless it is always more beneficial from a C and GHG perspective than the conventional drained agriculture sites (S.H. Knox, C. Sturtevant, L. Koteen, P. Oikawa, J. Verfaillie, D. Baldocchi, unpublished data). With a growing global interest in peatland rewetting for C sequestration (Couwenberg *et al.*, 2010; Maljanen *et al.*, 2010), there is a strong need for further research on the short and long-term effects of restoration activities on C and GHG exchange.

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