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Key Points:
- Wetland energy and carbon fluxes varied seasonally and between study periods
- Standing dead plant material may have reduced overall wetland productivity
- Large measured methane emissions offset the radiative cooling from photosynthetic carbon uptake

Supporting Information:
- Supporting Information S1

Correspondence to:
F. E. Anderson,
fanders@usgs.gov

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Variation of energy and carbon fluxes from a restored temperate freshwater wetland and implications for carbon market verification protocols

Frank E. Anderson[1], Brian Bergamaschi[1], Cove Sturtevant[2], Sara Knox[2], Lauren Hastings[3], Lisamarie Windham-Myers[4], Matteo Detto[5], Erin L. Hestir[6], Judith Drexler[1], Robin L. Miller[1], Jaclyn Hatala Matthes[2], Joseph Verfaillie[2], Dennis Baldocchi[2], Richard L. Snyder[7], and Roger Fujii[1]

[1]U.S. Geological Survey, Sacramento, California, USA, [2]Ecosystem Science Division, Department of Environmental Science, Policy and Management, University of California, Berkeley, California, USA, [3]Delta Stewardship Council, Sacramento, California, USA, [4]U.S. Geological Survey, Menlo Park, California, USA, [5]Smithsonian Tropical Research Institute, Panama City, Panama, [6]Department of Marine, Earth and Atmospheric Sciences, North Carolina State University, Raleigh, North Carolina, USA, [7]Department of Land, Air and Water Resources, University of California, Davis, California, USA

Abstract Temperate freshwater wetlands are among the most productive terrestrial ecosystems, stimulating interest in using restored wetlands as biological carbon sequestration projects for greenhouse gas reduction programs. In this study, we used the eddy covariance technique to measure surface energy carbon fluxes from a constructed, impounded freshwater wetland during two annual periods that were 8 years apart: 2002–2003 and 2010–2011. During 2010–2011, we measured methane (CH_4) fluxes to quantify the annual atmospheric carbon mass balance and its concomitant influence on global warming potential (GWP). Peak growing season fluxes of latent heat and carbon dioxide (CO_2) were greater in 2002–2003 compared to 2010–2011. In 2002, the daily net ecosystem exchange reached as low as $-10.6\,g\,C\,m^{-2}\,d^{-1}$, which was greater than 3 times the magnitude observed in 2010 ($-2.9\,g\,C\,m^{-2}\,d^{-1}$). CH_4 fluxes during 2010–2011 were positive throughout the year and followed a strong seasonal pattern, ranging from $38.1\,mg\,C\,m^{-2}\,d^{-1}$ in the winter to $375.9\,mg\,C\,m^{-2}\,d^{-1}$ during the summer. The results of this study suggest that the wetland had reduced gross ecosystem productivity in 2010–2011, likely due to the increase in dead plant biomass (standing litter) that inhibited the generation of new vegetation growth. In 2010–2011, there was a net positive GWP ($675.3\,g\,C\,m^{-2}\,yr^{-1}$), and when these values are evaluated as a sustained flux, the wetland will not reach radiative balance even after 500 years.

1. Introduction

Wetlands can be a more effective carbon sink than upland terrestrial ecosystems, as marsh vegetation typically has high gross primary productivity [*Rocha and Goulden*, 2009] and flooded environments create anaerobic soil conditions that prevent the decomposition of dead biomass and preserve sequestered carbon within the soil profile [*Mitra et al*., 2005; *Savage and Davidson*, 2001]. Recently, *Mitsch et al*. [2013] calculated that wetlands globally have the potential to sequester $1280\,Tg\,C\,yr^{-1}$ of carbon dioxide (CO_2) from the atmosphere. The preservation and reestablishment of wetlands in subtropical and temperate latitudes could support a large portion of this potential [*Aselmann and Crutzen*, 1989], as longer growing seasons lead to greater CO_2 uptake [*Whiting and Chanton*, 2001] and suitable conditions exist to favor high phytomass production [*Rocha and Goulden*, 2009]. However, as much as $448\,Tg\,C\,yr^{-1}$ of the CO_2 sequestered may return to the atmosphere as methane (CH_4) [*Mitsch et al*., 2013]. The anaerobic conditions that promote CO_2 sequestration also make wetlands the largest natural source of CH_4 emissions to the atmosphere [*Bloom et al*., 2010].

While there is good evidence for long-term net carbon accumulation in wetlands over hundreds to thousands of years [*Drexler et al*., 2009a; *Richardson*, 2010; *Rippke et al*., 2010], more research is needed to quantify the short-term carbon balance. Observed interannual variability in wetland net ecosystem exchange (NEE) of CO_2 has been frequently associated with changes in environmental conditions such as hydro-period, water table depth, the timing and magnitude of precipitation, and air and water temperatures [*Bubier et al*., 2003; *Chu et al*., 2014; *Lafleur et al*., 2003; *Petrescu et al*., 2015; *Roulet et al*., 2007]. Although interannual NEE variability may be high, restored wetlands can successfully sequester carbon [*Badiou et al*., 2011; *Knox et al*., 2015; *Mitsch et al*., 2013; *Waddington et al*., 2010]. As an example, converting agricultural land use to prairie wetlands in the central United States stored 378 Tg of organic carbon in the soil and plant biomass,

as well as through sediment accumulation, within 10 years of establishment [*Euliss et al*., 2006]. However, there are only a few examples in the literature that document long-term changes in developing freshwater wetland NEE at the decadal time scale [cf. *Drexler et al*., 2009a; *Rocha and Goulden*, 2008; *Roulet et al*., 2007].

Wetland restoration can also provide many other ecosystem services including the reduction of CO_2 respiration from historic soil carbon pools, land surface accretion, and relieving hydrostatic pressure on flood control infrastructure thus improving flood protection, particularly in coastal regions facing accelerated rates of sea level rise [*Hatala et al*., 2012; *Knox et al*., 2015; *Miller et al*., 2008]. Additionally, there is a growing need to better characterize the carbon and global warming potential (GWP) balance of restored wetlands as they may qualify as credited projects in carbon offset programs [e.g., *Emmet-Maddox et al*., 2010]. To support acceptance of wetland restoration as a creditable means of carbon sequestration, protocols for verification have been adopted or are under consideration by major carbon registries, e.g., Climate Action Reserve, American Carbon Registry, Verified Carbon Standard, and the European Union Emissions Trading System. However, due to the high, short-term variability in wetland NEE versus long-term sequestration, considerable uncertainty exists regarding the efficacy and permanence of such restoration projects and the ability to document any GWP reduction using verification protocols over the 50–100 year period of carbon offset trading programs [*Bonn et al*., 2014].

Verification protocols also need to consider the effects that CH_4 emissions have on the GWP balance in freshwater wetlands [e.g., *Bridgham et al*., 2006], as high CH_4 emissions can be relatively consistent despite changes in concurrent CO_2 uptake [*Chu et al*., 2015]. Modeled estimates of global CH_4 emissions from wetlands range from 60 to 210 Tg C yr^{-1} [*Bloom et al*., 2010; *Bridgham et al*., 2013], corresponding to a range from 2.0 to 7.1 Pg C yr^{-1} (CO_2-equivalent), given the GWP of CH_4 is 34 times CO_2 for a 100 year time horizon [*Myhre et al*., 2013]. Even though CH_4 emitted by wetlands may be only a small fraction (1.3 Pg C yr^{-1}, globally) of the CO_2 sequestered [*Mitsch et al*., 2013], the CO_2 equivalent of CH_4 emission rates has shown to be greater than CO_2 storage rates in wetland systems over short time horizons [*Petrescu et al*., 2015; *Whiting and Chanton*, 2001]. For restoration projects this can become significant as it may take decades to centuries for restored wetland radiative forcing to become negative [*Neubauer*, 2014; *Petrescu et al*., 2015], time in which changes in CO_2 uptake could alter the creditable sequestration benefit. Furthermore, *Neubauer and Megonigal* [2015] make a compelling argument that wetlands should be treated as a continuous source of methane, suggesting that the creditable benefit would be lower at market-relevant time scales.

In this study, we present the ecosystem exchange of atmospheric CO_2 and CH_4 data from different successional stages of a restored wetland, thus illustrating an 8 year interval of change for consideration in the development of monitoring and verification protocols within carbon emissions offset programs. Our study site is situated in the temperate climate of the western Sacramento-San Joaquin Delta, California, USA. Once an expansive, freshwater tidal marsh system, it was reclaimed for agriculture beginning in 1869 [*Thompson*, 1957]. In 1997, 7 ha of freshwater wetlands were restored on historical peat soils to test the potential rate for subsidence reversal through permanent flooding of the land surface. This larger-scale experimental study has been monitored for land surface elevation, vegetation dynamics, water quality, and carbon flux dynamics in comparison with neighboring agricultural production [*Miller*, 2011; *Miller et al*., 2008]. Our research questions focus on the magnitude and direction of atmospheric carbon fluxes through concurrent measurements of plant photosynthesis, ecosystem respiration, and CH_4 production and emission over seasonal to annual time scales. We sought to compare these fluxes from early (2002–2003) and later successional stages (2010–2011) of marsh development and considered changes over this period by coordinating with additional metrics tracked during study. Through our monitoring efforts, we show that (1) daily net CO_2 uptake was considerably smaller in the later stages of wetland development most likely due to reduced wetland plant photosynthesis, and (2) CH_4 emissions were large in 2010–2011, and likely similar in 2002–2003. The data illustrate the importance of CH_4 emissions and their potential in the annual GWP balance of a restored peat-building marsh. Despite years of relatively high productivity, variability in the exchange of atmospheric CO_2 and CH_4 observed in this study illustrates that extrapolating long-term carbon dynamics from short-term observations may be inaccurate, thus requiring management techniques that maintain consistent carbon storage rates and/or more comprehensive monitoring and modeling approaches [*Bonn et al*., 2014].

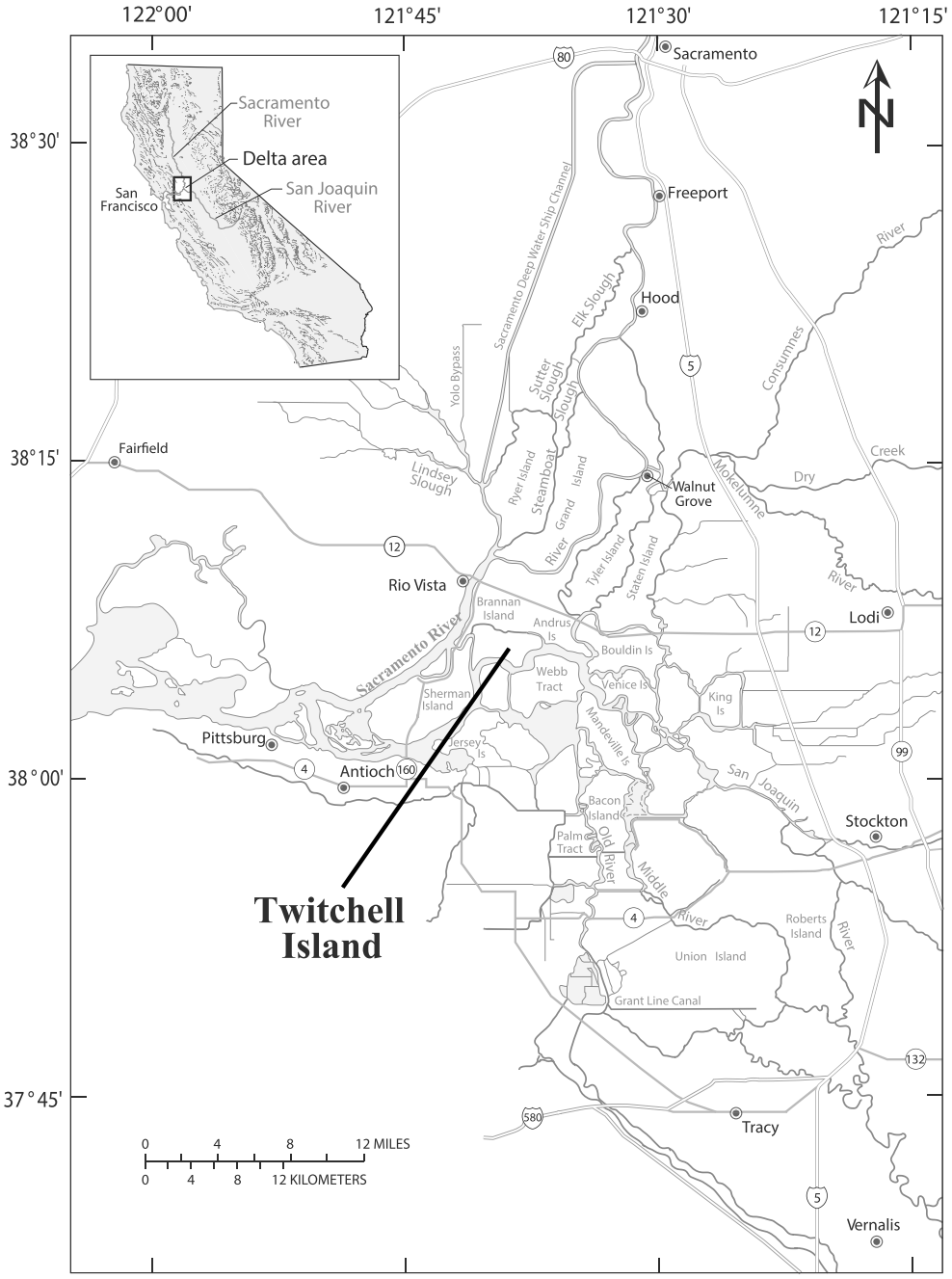

Figure 1. The restored, impounded wetland is located on Twitchell Island in the central Sacramento-San Joaquin Delta.

2. Methods

2.1. Site Description

Our study site was a restored, impounded freshwater wetland on Twitchell Island (38.1068°N, 121.6465°W), located in the central Sacramento-San Joaquin Delta, California, USA (hereafter, the Delta; Figure 1). Historically, the Delta was a tidal fresh water marsh system in dynamic equilibrium with sediment transport, tectonic processes, and sea level rise [*Mount and Twiss*, 2005]. Sediment cores indicate that rich organic (peat) soils up to 15 m thick formed over the past 6700 years, consisting of organic plant fragments of bulrushes (*Schoenoplectus* spp.), cattails (*Typha* spp.), and reeds (*Phragmites* spp.), and inorganic components of alluvial silt and clay [*Drexler et al*., 2009a, 2009b]. Peat soils on Twitchell Island are classified in the Rindge Series as

euic, thermic Typic Medisaprists, formed from tule and reed plant residuals as well as mixed alluvial silts and clays from surrounding Sierra Nevada and Coastal Mountain ranges [*Bossio et al*., 2006; *Drexler et al*., 2009a, 2009b]. *Fleck et al*. [2007] estimates that approximately 4.5 m of peat soil remains under the wetland; however, the upper 1.5 m of soil underwent repeated agricultural draining cycles, altering the structure and reactive properties from preagricultural development. Prior to the construction of the wetland, the site was in agricultural production since the midnineteenth century [*Miller et al*., 2008], typically *Zea mays* L. [*Fleck et al*., 2004].

The wetland was constructed in 1997 to test the extent to which wetlands could slow oxidation of the underlying peat soils and if wetland plants could be used to accrete land surface in subsided Delta islands. Soil was excavated and used to build a perimeter berm around a basin that was leveled to approximately 4.5 m below sea level [*Miller and Fujii*, 2010]. Water levels were maintained at 55 cm using weirs, similar to hydrological management of rice fields, with water fed by siphon from the San Joaquin River (Figure S1 in the supporting information). Other studies at the site report estimated mean flow rates to be 900 ± 200 L/min and approximate residence time of 13 days [*Gamble et al*., 2003; *Miller et al*., 2008]. They also report that specific conductance varied by season and ranged from 0.1 to 1.1 μS/cm, while periodic samples of pH ranged from 7 to 8.5, with a mean value of 7.5, and dissolved oxygen was above 50% measured at the inlet [*Miller et al*., 2008]. In 2010, the flow rate was reduced to 400 L/min—sufficient to minimize outflow and maintain the water level at 55 cm from the peat surface. A more detailed description of the study site and history can be found in *Miller et al*. [2008] and *Miller and Fujii* [2010].

To establish the wetland, initial populations of hardstem bulrush (*Schoenoplectus acutus*) were planted immediately following construction. By the summer of 1999, the plant community composition developed into a mosaic of roughly half open water and submerged aquatic vegetation, and half-emergent vegetation that included the planted bulrush and large areas of volunteer cattails (*Typha* spp.) [*Miller and Fujii*, 2010]. Vegetation mapping and field surveys from 2003 to 2012 indicate no major shifts in vegetation coverage, with the same observed distributions of open water versus emergent species since 2003 [*Byrd et al*., 2014]. Over a 10 year period (1997–2008), the emergent wetlands accreted on average 28–47 cm [*Miller et al*., 2008] of belowground detrital biomass. *Miller and Fujii* [2010] reported that mean annual emergent marsh carbon inputs into the wetland, as net primary productivity (NPP), ranged from 1295 to 2806 g C m^{-2} with the highest and lowest NPP recorded in 1999 and 2000, respectively. They reported NPP values of 2581 g C m^{-2} in 2001 and 1358 g C m^{-2} in 2006. NPP apparently continued to decline slowly to as recently as 2012 [*Byrd et al*., 2014]. Further, the area extent of emergent marsh had only a modest increase since 1999 [*Miller and Fujii*, 2010], and the absorption or reflection of incoming photosynthetic active radiation measured by *Schile et al*. [2013] suggests the presence of dense plant litter layer. The amount of dead material present as thatch in the wetland appears to have increased over this period, supported by changes in satellite data (normal difference vegetation index, NDVI) during August from 2002 to 2012 (see supporting information for further details).

2.2. Regional Climate

Regional climate in the Delta is classified as Mediterranean, characterized by warm, dry summers and cool, wet winters. The mean annual values of air temperature, solar radiation, and precipitation were 15.11 ± 0.58°C, 6566 ± 362 MJ m^{-2}, and 325 ± 120 mm, respectively. These values were from the nearby climate station on Twitchell Island (California Irrigation Management Information System (CIMIS) station #140), approximately 1 km to the NW of the study site and represents the period between 1997 and 2015. Outside the winter months, Twitchell Island experiences a persistent wind from the west, usually peaking in the late afternoon and evening, but elevated wind speeds occur throughout the diurnal period (Figure S2). Most flux studies experience higher percentages of missing data due to stable conditions that form at night and early morning hours [*Aubinet et al*., 2012; *Moffat et al*., 2007]; however, in our case, mean hour frictional velocity (u_*; m s^{-1}), a proxy for turbulence, remained high for both day ($u_* = 0.5$) and night ($u_* = 0.4$) conditions.

2.3. Greenhouse Gas and Micrometeorological Measurements

We used the eddy covariance (EC) technique, a statistical approach to measure the turbulent exchange of energy and mass between the atmosphere and land surfaces [*Baldocchi*, 2003], to assess surface energy and CO_2 fluxes during two separate periods: 2002–2003 and 2010–2011. Additionally, measurements of CH_4 fluxes were collected from April of 2010 to April of 2011. Due to power disruptions, moisture, and

Table 1. Eddy Covariance and Ancillary Equipment[a]

	2002	2003	2010	2011
EC instrumentation height (m)	2.9	2.9	2.9	2.9
Data collection	Campbell Scientific CR23X & CR10X	Campbell Scientific CR23X & CR10X	Campbell Scientific CR1000/CFM & CR23X-AM16/32	Campbell Scientific CR1000/CFM & CR23X-AM16/32
Data measurement frequency and Storage	10 Hz, 30 min	10 Hz, 30 min	10 Hz, 10 Hz	10 Hz, 10 Hz
Sonic anemometer	Campbell Scientific CSAT3	Campbell Scientific CSAT3	Campbell Scientific CSAT3	Gill WindMaster
CO_2/H_2O analyzer	LI-COR LI7500	LI-COR LI7500	LI-COR LI7500	LI-COR LI7500
CH_4 analyzer	-	-	LGR FGGA	-
Air temperature/vapor pressure	Vaisala HMP45C	Vaisala HMP45C	Vaisala HMP45C	Vaisala HMP45C
Radiation	REBS Q7.1 net radiometer	REBS Q7.1 net radiometer	Kipp & Zonen CNR1	Kipp & Zonen CNR1
Water temperature	Campbell Scientific CS107B	Campbell Scientific CS107B	HOBO TMCx-HD Thermistor	HOBO TMCx-HD Thermistor
Ground heat flux	REBS HFP	REBS HFP	-	-

[a]A detailed description of instrumentation, data collection and processing, references and calibration frequency is presented in the supporting material.

changes in weather patterns, the majority of usable CH_4 flux data coincided with the peak growing season, between spring (April/May) and fall (October/November). In April of 2011, the bulk of the EC tower equipment was removed from the site; however, in late July, flux measurements of CO_2 and water vapor resumed. The tower location and instrument heights (2.9 m above the water surface) remained the same throughout the entire study.

Briefly, instrument and tower configuration were designed to capture the largest footprint possible within the wetland, while reducing the amount of data falling outside its rectangular configuration. Output from EC instrumentation, i.e. the sonic anemometer and gas analyzers were, recorded at 10 Hz and fluxes were either calculated and stored in the data logger or the raw data stored for further processing. Postprocessing included data despiking, maximum lag correlation, and wind coordinate rotation that aligns the local mean wind vector and removes the vertical and lateral means, sonic temperature correction, air density fluctuations, stationarity, and u_* and wind direction filtering. A detailed description of instrumentation, data collection and processing, references, and calibration frequency is presented in the supporting information. Auxiliary measurements of radiation, air and water temperature, and atmospheric humidity were collected at a frequency of 1 Hz, and half-hour averages were saved on a separate data logger. Table 1 summarizes instrumentation, measurement duration, and frequency for operational periods.

2.4. Operational Periods

Data were analyzed and presented in the following manner: (1) all the available data for each year of the study, (2) peak growing season (August) when the system ran continuously during all four years, and (3) two gap-filled annual budgets from 15 April 2002 to 14 April 2003 and 15 April 2010 to 14 April 2011. Over the four years, approximately 20,000 half-hour means were calculated for each of the turbulent fluxes of sensible heat, latent heat, and CO_2. For each year, the percentages of rejected or short missing data gaps for the three fluxes ranged from 38 to 60% (Table 2). These percentages are comparable to the typical range reported by *Moffat et al*. [2007] for missing data using the EC technique but differ from other studies, as the rectangular configuration of the wetland was responsible for almost 80% of the flux data removed via the wind direction filter and not low turbulent, nighttime conditions. For CH_4, rejected or missing values were much higher, ranging from 66 to 88% in 2010 and 2011, respectively (Table 2), which were a result of ancillary power outages used to supply AC power to the CH_4 analyzer and in 2011, larger missing gaps of flux data in January and April when the analyzer was not running. Due to the extended breaks in measurements from year to year and higher percentages of rejected values outside the summer growing season, we examined the surface energy and CO_2 fluxes during the month of August as representation of peak growing season conditions [*Byrd et al*., 2014]. Percentages of observed rejected or missing values were much lower, ranging from 17.4 to 27.1% (Table 2). To estimate the annual budgets between study periods, we chose 15 April through 14 April for both 2002–2003 and 2010–2011 as these dates represented the highest overlap of flux measurements between the two periods. For both years, the percentages of rejected or missing surface

Table 2. Periods of Flux Measurements and Quality Control Statistics

	2002	2003	2010	2011
Dates of operation	5/23–11/18	2/12–2/28, 4/11–11/21	4/23–11/23	1/30–3/28, 7/7–11/28
H rejected/missing (%)	38.0[a], 17.4[b], 67.2[c]	43.0[a], 19.2[b]	40.1[a], 23.5[b], 61.9[c]	56.2[a], 20.1[b]
LE rejected/missing (%)	38.0[a], 17.5[b], 67.3[c]	43.3[a], 20.8[b]	41.2[a], 24.1[b], 62.7[c]	57.0[a], 21.4[b]
CO_2 rejected/missing (%)	38.1[a], 17.6[b], 67.3[c]	43.5[a], 21.0[b]	44.6[a], 27.1[b], 65.0[c]	60.0[a], 23.8[b]
CH_4 rejected/missing (%)	-	-	65.7[a], 42.8[b], 77.9[c,d]	87.5[d]
Energy balance closure	0.76[a], 0.70[b], 0.76[c]	0.91[a], 0.97[b]	0.88[a], 0.74[b], 0.89[c]	0.79[a], 0.85[b]

[a]All available data by dates of operation for each year.
[b]Growing season (August).
[c]15 April through the following 14 April.
[d]CH_4 analyzer not in operation from 23 November 2010 to 30 January 2011.

energy and CO_2 fluxes were 67% in 2002–2003 and 62–65% in 2010–2011, and 77.9% for CH_4 flux in 2010–2011 (Table 2). The higher percentages were mostly due to intermittent days or extended periods of missing data, from 19 November 2002 to 12 February 2003 and 23 November 2010 to 30 January 2011.

2.5. Energy Balance Closure

Energy balance closure was analyzed based on the three previously discussed data periods. We calculated the closure using an equal-weight energy balance ratio approach, similar to equation (2) in *Wilson et al*. [2002] or the "bulk method" described in *Franssen et al*. [2010]. We deviate from the methodology by summing the combined terms over the diel average, applying an equal weight to all times of the day. Using this approach, energy balance closure ranged from 0.76 to 0.91 for all available data in each year, from 0.70 to 0.97 for the month of August, and 0.76 and 0.89 for 2002–2003 and 2010–2011, respectively (Table 2). These values are typical and fall within the range of other terrestrial systems [*Stoy et al*., 2013].

2.6. Gap Filling and Uncertainty Estimation

We used an artificial neural network (ANN) technique both to gap-fill missing fluxes and to estimate annual gap filled and random error uncertainty [*Moffat et al*., 2007; *Richardson et al*., 2008]. Rejected or missing half-hourly flux data were gap filled using ANN model inputs based on season, time of day, net radiation, water temperature, air temperature, and vapor pressure deficit [*Moffat et al*., 2007]. There were very few instances where ancillary micrometeorological data were missing, although when this occurred, measurements of air temperature, vapor pressure, and radiation were supplemented from the nearby CIMIS climate station.

We followed the ANN optimization procedure outlined in *Knox et al*. [2015]. Briefly, we minimized bias in ANN training toward periods of greater data coverage (i.e., summer and daytime conditions), by representatively sampling natural clusters of the environmental data using the *k*-means algorithm in Matlab (MathWorks, Natick, MA, USA). The ANN was trained, validated, and tested each using one third of the representatively sampled data [*Papale and Valentini*, 2003]. Starting weights for the ANN nodes were reinitialized 10 times to avoid local minima. To facilitate generalizability, four different ANN architectures of increasing complexity were tested, and the simplest architecture was chosen for which increasing complexity resulted in less than a 5% gain in model accuracy. This entire procedure was repeated with 20 resampled sets of the data used to train, test, and validate the ANN. Missing half-hour fluxes were replaced by the median value from the 20 resultant predictions.

The annual cumulative uncertainty from gap-filled data and random measurement error was estimated as the variance of 20 ANN-modeled time series. For gap-filled values, the variance of the ANN-modeled output includes both gap filled and random uncertainty, as the residuals of the highly tuned ANN model provide a good, if not conservative estimate of the random flux uncertainty [*Moffat et al*., 2007; *Richardson et al*., 2008]. The random errors in EC fluxes follow a double exponential distribution with a standard deviation that varies with flux magnitude [*Richardson and Hollinger*, 2005]. Here we computed the random uncertainty of measured values by using the residuals of the median ANN predictions to parameterize the double exponential distribution according to flux magnitude [*Moffat et al*., 2007], binned every 5 $\mu mol\, m^{-2}\, s^{-1}$ for CO_2 flux and 50 $\eta mol\, m^{-2}\, s^{-1}$ for CH_4. For each measured value in the time series, we drew 100 random errors using the appropriately parameterized binned distribution, estimating the random flux uncertainty from the variance.

The annual cumulative uncertainty was computed following *Richardson and Hollinger* [2007] by adding the cumulative gap filling and random measurement uncertainties in quadrature.

We minimized uncertainty due to systematic measurement bias through a careful study design, instrument siting, frequent calibration, and proper processing procedures [*Aubinet et al*., 2012]. The effects of three potential systematic biases—high-frequency loss (low-pass filtering), low-frequency loss (high-pass filtering), and the differences in heating of the open-path EC equipment compared to ambient air [*Burba et al*., 2008]—were tested on a subset of summer, growing season data. Results from a one-way analysis of variance (ANOVA) in MATLAB (MathWorks, Natick, MA, USA) indicated there were no significant differences ($p = 0.1673$) between data not corrected and data corrected for the three systematic biases or any combination of the three. We acknowledge these potential biases, where correcting for high-frequency loss slightly increased CO_2 uptake during the day and respired CO_2 at night and where correcting for instrument heating slightly decreased CO_2 uptake during the day, we chose not to apply them here. The correction magnitudes were well within the random uncertainty of half-hourly CO_2 and energy flux measurements and an order of magnitude smaller than the Webb-Pearman-Leuning density corrections [*Burba et al.*, 2008; *Webb et al.*, 1980]. However, a known systematic bias in the closed-path measurements used for CH_4 was corrected prior to the ANN uncertainty estimation [*Detto et al*., 2011]. High-frequency losses in closed-path analyzers such as that used in this study for CH_4 can be quite large [*Aubinet et al*., 2012]. To correct this bias, we followed the approach of *Detto et al*. [2011] by developing a wind speed-dependent correction factor for the closed-path measurements using the ratio of CO_2 measured by the closed-path analyzer to that measured using an open-path analyzer (LI7500). A linear fit was calculated between this ratio and the wind speed for conditions that were dominated by photosynthetic activity or when the CO_2 flux was less than $-5\,\mu mol\,CO_2\,m^{-2}\,s^{-1}$ [*Detto et al*., 2011]. As with *Detto et al*. [2011], at low wind speeds the correction factor was negligible, but an increase in wind speed increased the correction. Uncertainty due to this correction was incorporated into the overall ANN-based assessment of measurement uncertainty by adjusting each half-hour average in the time series to include a randomly assigned error such that the assigned errors together represent the probability distribution of error within the 95% confidence interval of the correlation between the measurement ratio and wind speed.

2.7. NEE Partitioning

NEE of CO_2, measured directly by the EC technique, is the net balance between gross ecosystem production (GEP) and ecosystem respiration (ER):

$$\mathrm{NEE} = \mathrm{GEP} + \mathrm{ER} \tag{1}$$

where negative GEP values represent a net flux of carbon input from the atmosphere to the wetland and positive ER values represent the carbon output from the wetland to the atmosphere. We assume that GEP is zero at night, in the absence of photosynthesis. NEE is the integrated value of GEP and ER, where negative NEE is the net uptake of CO_2 into the wetland and a positive NEE indicates a net CO_2 loss from the combined effects of autotrophic and heterotrophic respiration under reduced or the absence of photosynthesis. In order to estimate GEP and ER, we follow the flux-partitioning algorithm described in *Reichstein et al*. [2005] that uses temperature estimated nighttime respiration values and then extrapolate the results to estimate daytime values of ER using light response curves. As others have found [e.g., *Desai et al*., 2008], this modeled approach may be sensitive to which temperature parameter is used, i.e., water temperature (T_w), air temperature (T_a), or vegetation surface temperature (T_{sfc}), when available. Here we present GEP and ER values using T_a modeled ER values as T_a had the longest, continuous data set and testing showed that there was very little difference in the modeled values between all three temperatures.

2.8. Statistics

Statistical analysis was performed using a one-way analysis of variance (ANOVA) in MATLAB (MathWorks, Natick, MA, USA). Tukey's honestly significant difference criterion with a $p < 0.001$ was used to identify significant differences between (1) the annual variation in environmental variables: solar radiation (R_s), net radiation (R_n), air temperature (T_a), water temperature (T_w), vapor pressure deficit (VPD), wind speed (WS), and precipitation (PPT), (2) average daily flux values of NEE, GEP, and ER with environmental variables of R_n, T_a, T_w, VPD, and WS, and (3) to compare the fluxes of NEE, GEP, and ER between study periods.

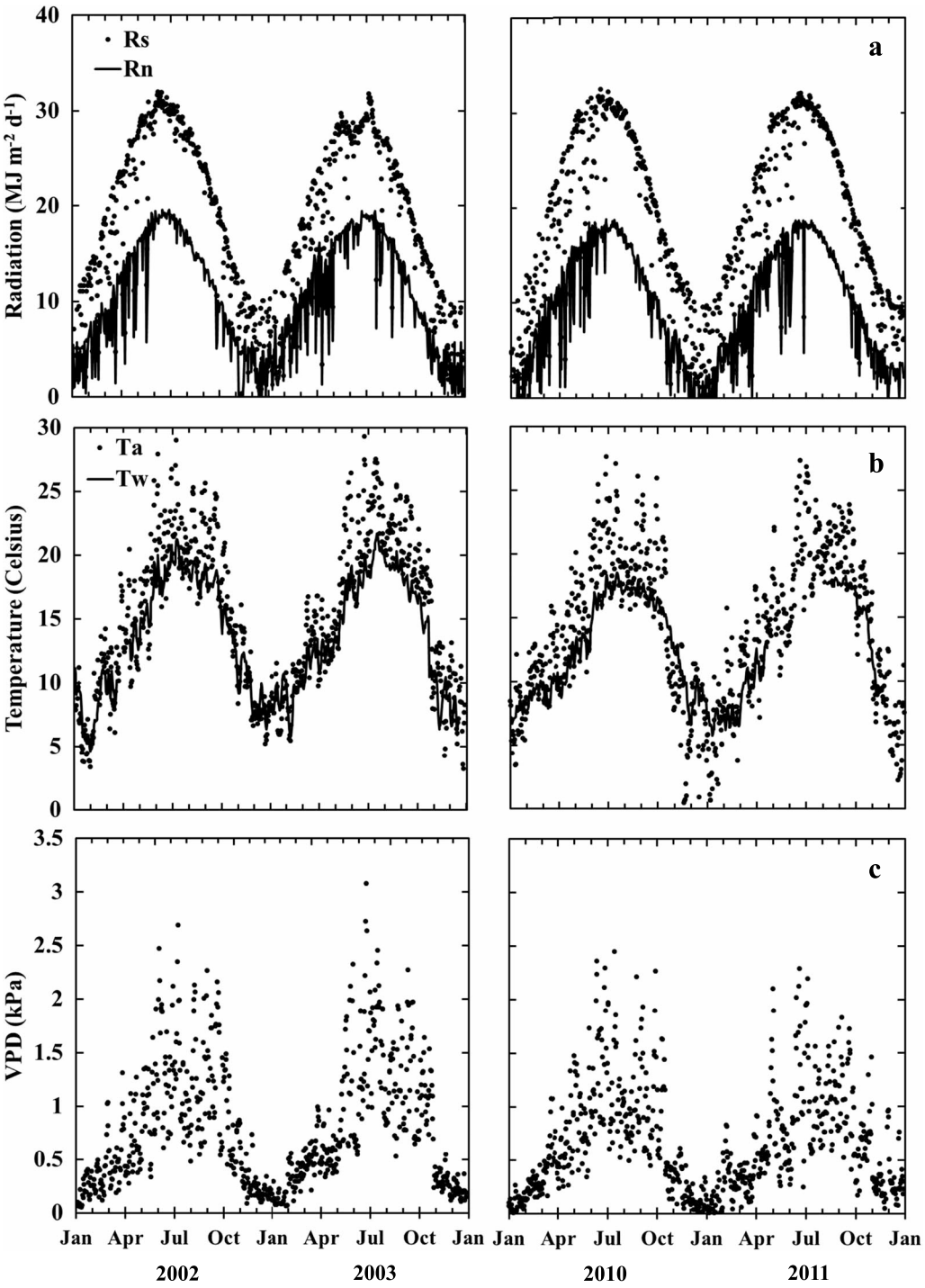

Figure 2. Daily-integrated values of (a) shortwave (R_s) and net radiation (R_n), (b) air (T_a) and water temperatures (T_w), and (c) vapor pressure deficit (VPD). Patterns in the annual plots of environmental conditions show typical Mediterranean climate conditions in the Sacramento-San Joaquin Delta region with maximum values of radiation, temperature, and VPD in the summer and minimum values in the winter. Patterns and magnitude of these conditions were similar for all four years, except for air and water temperatures in 2010 and 2011 when winter air temperatures and summer water temperatures were a few degrees lower.

Table 3. Average Annual and Daily-Integrated Environmental Conditions From 2002–2003 to 2010–2011: Solar Radiation (R_s), Net Radiation (R_n), Air Temperature (T_a), Water Temperature (T_w), Vapor Pressure Deficit (VPD), Wind Speed (WS), and Precipitation (PPT)[a]

Year	R_s (MJ m^{-2} yr^{-1})	R_n (MJ m^{-2} yr^{-1})	Average Daily T_a (Min, Max) (°C)	Average Daily T_w (Min, Max) (°C)	Average Daily VPD (Min, Max) (kPa)	Average Daily WS (Min, Max) (m s^{-1})	PPT January–December (mm)
2002	6839_A	$3809_{A,B,C}$	$14.8_{A,B}$ (3.4, 29.0)	13.5_A (4.6, 21.3)	$0.8_{A,B}$ (0.05, 2.7)	2.1_A (0.6, 6.6)	309_A
2003	6374_A	3839_B	15.5_A (3.2, 29.3)	13.8_A (5.8, 21.7)	0.8_A (0.06, 3.1)	2.1_A (0.5, 6.0)	295_A
2010	6581_A	3326_C	14.0_B (-2.4, 27.7)	12.6_B (6.6, 18.5)	0.7_B (0.0, 2.4)	3.2_B (0.9, 8.4)	468_A
2011	6693_A	$3530_{A,B,C}$	14.1_B (-0.7, 27.4)	12.1_B (5.7, 18.2)	0.7_B (0.0, 2.3)	3.1_B (0.8, 8.4)	344_A

[a]Capital letters indicate significant differences between the mean values (Tukey's honestly significant difference, $p < 0.01$).

3. Results

3.1. Annual Environmental Conditions

Environmental conditions over the two periods were evaluated and compared for significant differences (Figure 2 and Table 3). Daily averages of R_s and R_n were typical for the Mediterranean climate of the Delta region with minimum values less than 10 MJ m^{-2} d^{-1} during the winter for both variables and maximum values greater than 30 MJ m^{-2} d^{-1} for R_s and just under 20 MJ m^{-2} d^{-1} for R_n in the summer. We found that for all four years, mean annual R_s values were similar; however, the pattern of daily values of R_n throughout the year was significantly different and enough to create differences in the annual means. Seasonal patterns of VPD and air temperatures were similar between years. Daily averages of VPD and T_a peaked during the summer at approximately 2.5 kPa and between 25 and 30°C, respectively. Within each study period, 2002–2003 and 2010–2011, we found VPD and T_a to be similar. We also found that these variables were similar between 2002, 2010, and 2011. Average daily wind speeds were found to be similar within, but different between study periods, ranging from 0.5 to 8.4 m s^{-1} and were predominantly from the west in the summer, but were more variable outside the summer growing season (Figure S2).

Annual differences in environmental conditions for all four years, ranged from as low as 7% (R_s) to as high as 49% (PPT) (Table 3). When averaged, total annual R_n values were higher in 2002 and 2003 (3824 MJ m^{-2} yr^{-1}) compared to 2010 and 2011 (3428 MJ m^{-2} yr^{-1}). Average annual T_a across all four years was 14.6°C, 0.5°C lower than both the CIMIS long-term average (1997–2015) and Antioch climate station (1949–1999) [*Hatala et al.*, 2012]. The difference in average T_a was 1.5°C for all four years, which was slightly higher than the annual variability of 1.3°C reported for a temperate wetland in Southern California but lower than the interannual variability for other terrestrial systems compiled in *Rocha and Goulden* [2008]. Average T_w was found to be significantly different and lower in 2010 and 2011; however, minimum temperatures in 2010 were 2°C warmer compared to 2002 (Table 3). Average annual wind speeds were also found to be significantly different between years and greater in 2010 and 2011 (~1 m s^{-1}), while maximum daily average wind speeds were close to 2 m s^{-1} higher compared to 2002 and 2003. It is important to note that the maximum wind speeds occurred during the winter and were caused by the passage of low-pressure systems. Sustained summer, afternoon winds were similar if not slightly higher in the first study period.

We also tested for significant differences in environmental conditions during the spring and summer (1 May to 30 September) for all four years to determine if there were conditions that may have influenced the wetland vegetation during the peak growing season (not shown here). Results from the test showed that the spring and summer means were similar to results of the annual mean testing (Table 3), except for R_n and TW. R_n values were found to be similar in all four years as this period is characterized by clear and warm conditions. T_w was interesting in that 2011 became similar to 2002 and 2003, while 2010 was significantly different from the three other years.

3.2. Variability in Energy Partitioning and Carbon Dioxide Uptake

For both study periods, daily-integrated surface energy and CO_2 fluxes followed the expected seasonal pattern, with maximum energy flux and CO_2 uptake during the peak growing season (Figure 3). Maximum

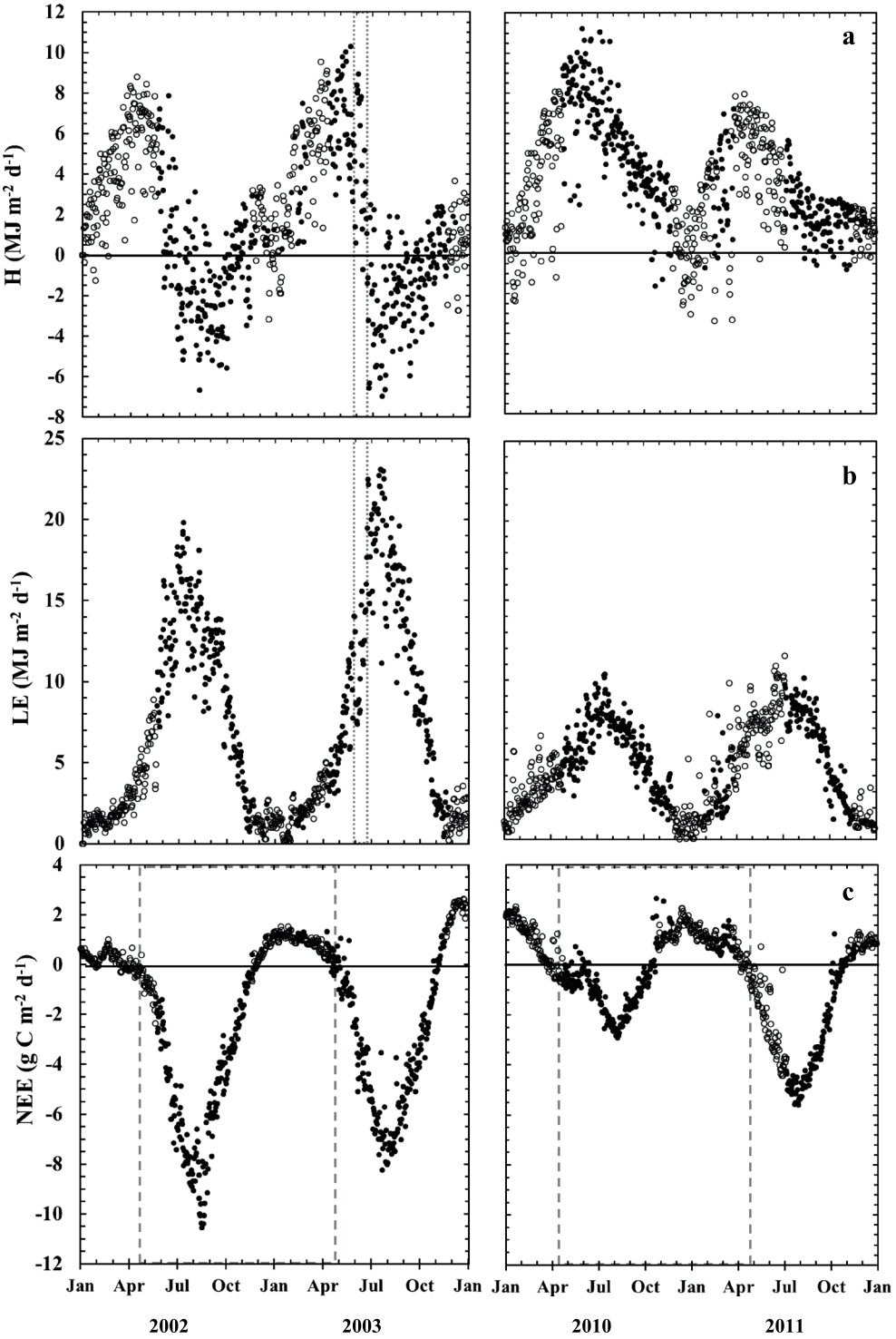

Figure 3. Daily averaged (a) sensible heat flux (H), (b) latent heat flux (LE), and (c) net ecosystem exchange (NEE) for years 2002–2003 and 2010–2011. Closed circles represent days when the eddy covariance system was in operation. Open circles represent days when the system was inoperable or there were no resolvable half-hour fluxes and daily values were estimated from the artificial neural network (ANN) gap-gilling routine. The dotted area in the sensible and latent heat fluxes (2002 and 2003 plots) represents the period when there was a rapid transition of energy partitioning in the canopy (24 May through 27 June 2003). The dashed areas in the NEE plots represent the period used to calculate the annual cumulative sum.

positive values of sensible and latent heat fluxes and maximum negative values of NEE occurred during the spring and summer months, driven by high R_s, warm temperatures, and peak plant photosynthesis. Negative values of sensible heat, minimum values of latent heat, and positive values of NEE occurred during the fall and winter months as solar insolation decreased, temperatures cooled, and wetland plants became dormant.

Despite the similar seasonal timing between study periods, there were significant differences in both the overall magnitude and timing of surface energy and CO_2 fluxes, indicative of changes in wetland carbon sequestration potential. Most notably, sensible heat flux exhibited a large excursion from positive to negative values in 2002 and 2003, as more energy was made available for evaporation. No comparable pattern occurred in 2010 and 2011, and positive values were observed throughout late summer and fall (Figure 3a). The seasonality of the sensible heat flux was also different between study periods, though the seasonality of environmental conditions was consistent. For example, in 2003, the maximum sensible heat flux of $10\,MJ\,m^{-2}\,d^{-1}$ was measured on 24 May and the minimum of $-8\,MJ\,m^{-2}\,d^{-1}$ was measured on 27 June. By comparison, in 2010, an equivalent maximum value of $11\,MJ\,m^{-2}\,d^{-1}$ was reached on 7 July and a greater minimum value of $-2\,MJ\,m^{-2}\,d^{-1}$ occurred much later in the year (8 November) than the minimum observed in the earlier study period. Changes in both the timing and magnitude of the sensible heat flux suggest a change in how energy was partitioned between the two study periods. Negative fluxes of sensible heat in 2002–2003 indicate that sensible heat was extracted from the atmosphere, increasing rates of latent heat. Conversely, in 2010–2011, positive fluxes of sensible heat dissipated energy from the canopy to the atmosphere throughout the growing season.

The difference between study periods was also evident in the maximum values and seasonality of the measured latent heat flux (Figure 3b). In all four years of the study, maximum daily averages peaked during the growing season and were at a minimum during the winter, coinciding with changes in plant phenology and VPD. Peak summer values of latent heat flux in the early study period were above $20\,MJ\,m^{-2}\,d^{-1}$ for both years, while they did not exceed $12\,MJ\,m^{-2}\,d^{-1}$ in the later study period. The observed latent heat flux in 2003 quickly rose from $6.7\,MJ\,m^{-2}\,d^{-1}$ on 24 May to a maximum of $23.6\,MJ\,m^{-2}\,d^{-1}$ on 27 June, coinciding with the sharp decline in sensible heat flux, and then declining to a minimum (~$1\,MJ\,m^{-2}\,d^{-1}$) in late November. By comparison, in 2010, the latent heat flux had a more dampened signal rising from approximately $5\,MJ\,m^{-2}\,d^{-1}$ in May to $11\,MJ\,m^{-2}\,d^{-1}$ on 7 July, and then declining to approximately $1\,MJ\,m^{-2}\,d^{-1}$ in November. The sharp rise in latent heat flux and corresponding decline in sensible heat flux in 2003 indicate the earlier study period exhibited a greater modulation of energy fluxes due to photosynthetic activity, most likely due to higher green leaf area and canopy density. In 2010, both sensible and latent heat fluxes closely tracked the influx of solar radiation and were similar in magnitude throughout the year.

As expected, the seasonal pattern of NEE more closely tracked patterns in latent heat flux and the seasonal growth of the wetland canopy (Figure 3c). Greatest CO_2 uptake was observed in all years from late July through November; however, as with the surface energy fluxes, there were significant differences between study periods. Higher uptake rates were observed in 2002 and 2003 with mean daily NEE values reaching $-10.6\,g\,C\,m^{-2}\,d^{-1}$ and $-8.8\,g\,C\,m^{-2}\,d^{-1}$, respectively. In 2010, CO_2 uptake was greatly reduced, with NEE values less than one third of those measured in the early study period ($-2.9\,g\,C\,m^{-2}\,d^{-1}$). In 2011, maximum CO_2 uptake was $-5.6\,g\,C\,m^{-2}\,d^{-1}$, nearly twice that observed in 2010; however, uptake was approximately one half to one third of the maxima observed in 2002 and 2003.

3.3. Gross Ecosystem Productivity and Respiration During the Peak Growing Season

We compared modeled gross ecosystem productivity (GEP) and respiration (ER) for the month of August to examine differences between the peak growing seasons of the two study periods (Figure 4). NEE values for all years indicated that the wetland was a carbon sink during the month of August. As observed with the longer time series, the average CO_2 uptake was approximately twice as large during August of the earlier study period (2002 and 2003) than in August of the later study period (2010 and 2011). August daily average ER was higher in the early study period than the late study period, but the difference in ER rates between study periods was far lower than for GEP. ER in the early study period averaged $5.0\,g\,C\,m^{-2}\,d^{-1}$, 39% higher than that seen in the later study period, which averaged $3.6\,g\,C\,m^{-2}\,d^{-1}$. The reduced ER rates in 2010 and 2011, may be a function of lower air and water temperatures that were measured in the later study period (~1°C; Table 3) [see *Davidson et al*., 2006]. However, we cannot rule out that the reduction in 2010 and

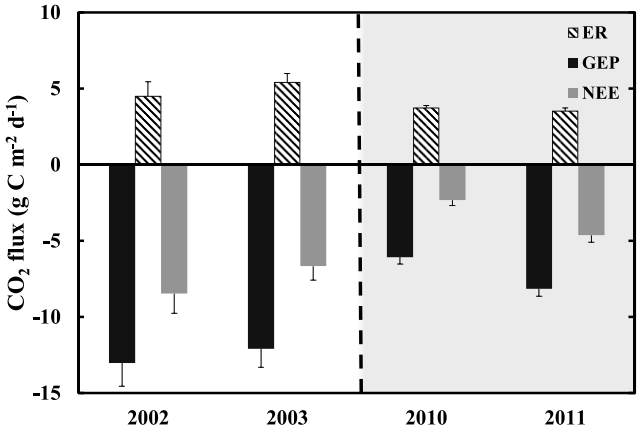

Figure 4. Average daily rates of gross ecosystem production (GEP), ecosystem respiration (ER), and net ecosystem exchange (NEE) for the month of August, a representation of peak growing season conditions. Average values of GEP, ER, and NEE ranged from $-13.0\,g\,C\,m^{-2}\,d^{-1}$ in 2002 to $-6.1\,g\,C\,m^{-2}\,d^{-1}$ in 2010, $5.4\,g\,C\,m^{-2}\,d^{-1}$ in 2003 to $3.5\,g\,C\,m^{-2}\,d^{-1}$ in 2011, and $-8.5\,g\,C\,m^{-2}\,d^{-1}$ in 2002 to $-2.3\,g\,C\,m^{-2}\,d^{-1}$ in 2010, respectively. Error bars represent the standard deviation from all the daily flux values during the month of August.

2011 may also be attributed to lower autotrophic respiration, as GEP and aerenchyma pathway system were reduced [*Aubrey and Teskey*, 2009], and/or lower heterotrophic respiration due to reduced labile carbon availability [*Davidson et al.*, 2006; *Gu et al.*, 2004]. Nevertheless, we surmise that the difference in NEE between study periods was driven by differences in GEP rather than respiration, with GEP in the earlier study period averaging $12.6\,g\,C\,m^{-2}\,d^{-1}$ or 176% compared to the later study period, which averaged $7.1\,g\,C\,m^{-2}\,d^{-1}$.

3.4. Seasonal CH_4 Fluxes

CH_4 emissions were measured in the later study period from 23 April 2010 to 28 March 2011 to quantify the annual ecosystem carbon balance and global warming potential (GWP). The measured annual CH_4 flux was high ($56.3 \pm 6.4\,g\,C\,m^{-2}\,yr^{-1}$) in comparison to values reported in the literature [*Petrescu et al.*, 2015]. As observed in other studies, the seasonal variation of CH_4 emissions was related to that of latent heat flux and NEE, seasonal growth of the wetland plant canopy, peak seasonal temperatures, and bulk or pressurized convective flow through emergent vegetation [*Chanton et al.*, 1993; *Garnet et al.*, 2005; *Kim et al.*, 1999]. Greatest emissions ($188.6–375.9\,mg\,C\,m^{-2}\,d^{-1}$) occurred between July and September, and minimum values ($38.1–102.4\,mg\,C\,m^{-2}\,d^{-1}$) occurred in late January through March (Figure 5). On a molecular carbon basis, the peak in CH_4 flux ($375.9\,mg\,C\,m^{-2}\,d^{-1}$) makes up a small percentage (13.0%) of peak season

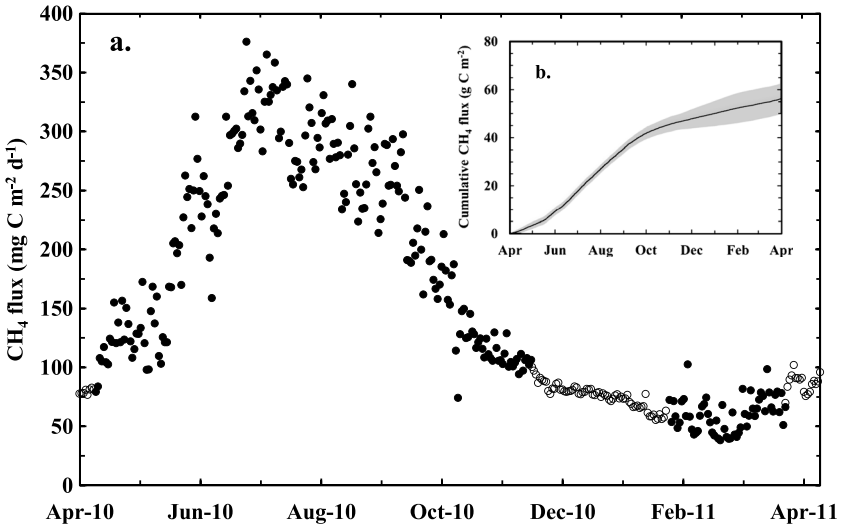

Figure 5. (a) Daily-integrated CH_4 fluxes from 15 April 2010 through 15 April 2011. Closed circles represent days when the eddy covariance system was in operation. Open circles represent days when the system was inoperable or there were no resolvable half-hour fluxes and daily values were estimated from the artificial neural network (ANN) gap-filling routine. CH_4 fluxes increased during the spring, peaked in late July and August, and declined to a minimum in January and February. Seasonal differences correspond to canopy growth, air temperature, and are inversely proportional to daily net ecosystem exchange (NEE) rates. (b) Cumulative daily CH_4 fluxes for the same period resulted in an annual CH_4 flux of $56.3\,g\,C\,m^{-2}$. Gray shaded area represents the uncertainty at the 95% confidence interval from the 20 predictions of the ANN gap-filling routine for both measured and ANN gap-filled values.

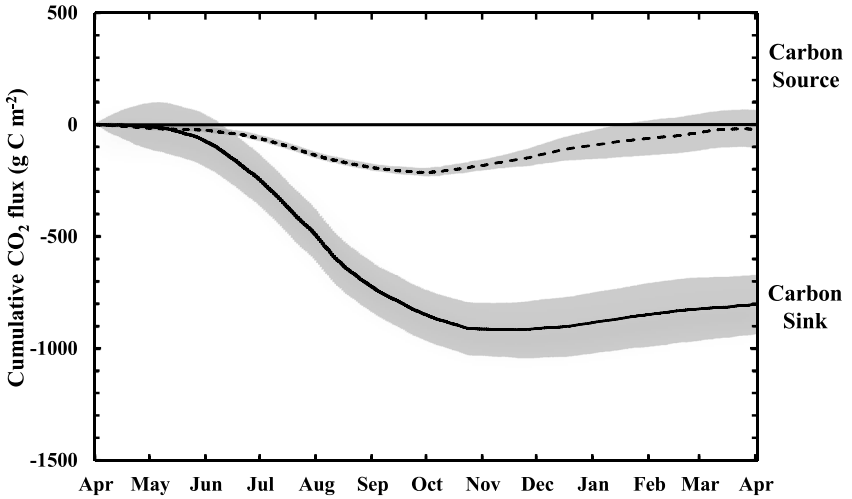

Figure 6. Cumulative net ecosystem exchange (NEE) plotted from 15 April 2002 through 15 April 2003 (solid line) and 15 April 2010 through 15 April 2011 (dashed line). Gray shaded area represents the uncertainty at the 95% confidence interval from the 20 predictions of the artificial neural network (ANN) gap-filling routine for both measured and ANN gap-filled values. In general, uptake of CO_2 was greater in 2002–2003 compared to 2010–2011, a possible indication that conditions were more favorable for potential wetland carbon sequestration and as a result, the wetland was a carbon sink. Conversely, less favorable conditions in 2010–2011 resulted in a near neutral carbon budget.

photosynthetic uptake in 2010 ($-2.9\,g\,C\,m^{-2}\,d^{-1}$). However, using these peak season numbers and the GWP for CH_4 as 34 times the CO_2 equivalent (100 year time horizon; *Myhre et al*. [2013]), the CO_2 equivalent from the CH_4 flux was 158.6% ($4.6\,g\,C\,m^{-2}\,d^{-1}$) of the CO_2 sequestered through photosynthesis (Figure 3; $-2.9\,g\,C\,m^{-2}\,d^{-1}$).

3.5. Cumulative Annual CO_2 and CH_4 Budgets

Comparing the cumulative annual NEE budgets from 15 April 2002 to 14 April 2003 and 15 April 2010 to 14 April 2011 revealed that while the wetland was a net CO_2 sink in 2002–2003 ($-804.2 \pm 131.5\,g\,C\,m^{-2}\,yr^{-1}$), it was near carbon neutral in 2010–2011 ($-21.1 \pm 83.1\,g\,C\,m^{-2}\,yr^{-1}$) (Figure 6). The biggest difference between the two periods was observed during the summer growing season, where in 2002–2003 NEE was almost 4 times greater than the rate measured in 2010–2011. In both periods, winter respiration reduced the cumulative annual NEE magnitude. CH_4 emissions remained positive throughout the year in 2010–2011 with the highest rates observed during the summer growing season (Figure 5a). The annual cumulative CH_4 flux in 2010–2011 (Figure 5b; $56.3\,g\,C\,m^{-2}\,yr^{-1} \pm 6.4\,g\,C\,m^{-2}\,yr^{-1}$) was small on a molar basis in comparison to the cumulative annual CO_2 uptake in 2002–2003 (Figure 6; $-804.2 \pm 131.5\,g\,C\,m^{-2}\,yr^{-1}$) but of similar magnitude to the cumulative annual CO_2 uptake in 2010–2011 (Figure 6; $-21.1 \pm 83.1\,g\,C\,m^{-2}\,yr^{-1}$).

4. Discussion

4.1. Evaluating Differences in Wetland Carbon Uptake

The contrast in the cumulative annual NEE from 2002–2003 ($-804.2 \pm 131.5\,g\,C\,m^{-2}\,yr^{-1}$) to 2010–2011 ($-21.1 \pm 83.1\,g\,C\,m^{-2}\,yr^{-1}$) reveals that the magnitude of carbon uptake may be unpredictable, or that CO_2 uptake may be larger immediately following restoration, as suggested in *Schrier-Uijl et al*. [2014]. Based on the evidence from the seasonal timing and magnitude of sensible and latent heat fluxes, as well as the similarity in ER compared to GEP between study periods (Figure 4), the observed difference appears to reflect a change in wetland photosynthetic production. This is in agreement with biomass sampling, which demonstrated that plant production at the wetland progressively decreased from $2581\,g\,C\,m^{-2}\,yr^{-1}$ in 2001 to $1358\,g\,C\,m^{-2}\,yr^{-1}$ in 2006 [*Miller and Fujii*, 2010]. We note that the differences in magnitude and range in annual NEE measured in this study versus those reported by *Miller and Fujii* [2010] were likely due to the difference between the flux tower footprint and the more widely disbursed biomass sampling plots.

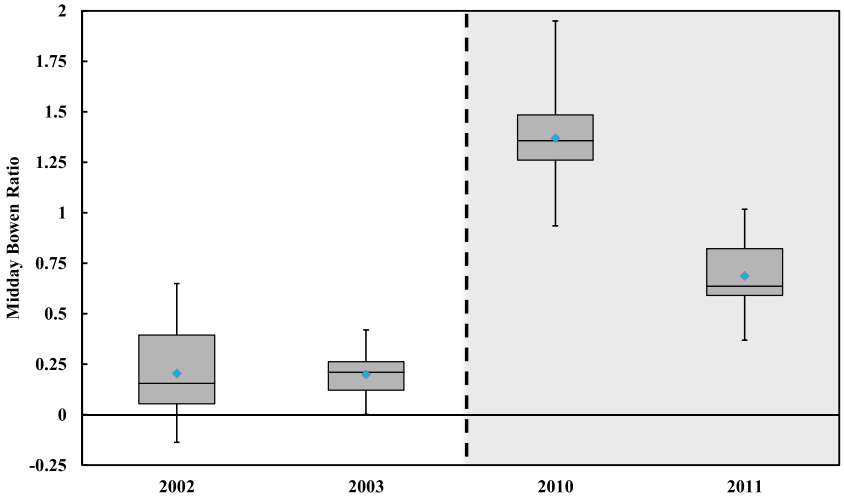

Figure 7. Average midday (1000–1400 local time) Bowen ratios (the ratio of sensible to latent heat fluxes) were compared for the month of August, representing peak growing season conditions. The diamond shape point represents the mean daily value for the month. The median or second quartile is represented by line in the box, while the top and bottom of the box represents the extent of the first and third quartiles, and the whiskers represent the maximum and minimum values. Average daily values were close to 0.2 in 2002 and 2003, 1.4 in 2010, and 0.7 in 2011. The interannual variability in midday Bowen ratios correspond to similar magnitude changes observed for fluxes of LE and CO_2 uptake, where the highest values were observed in 2002 and 2003, and lowest in 2010.

Differences in environmental factors do not explain the observed differences in photosynthetic activity between study periods. We compared mean daily values of GEP, ER, and NEE to environmental conditions (R_n, T_a, VPD, WS and T_w) for the peak growing season (August). Results revealed that GEP was most closely related to T_w (r^2: 0.54; $p < 0.001$). However, we are aware of no direct mechanistic reason that the modest correlation in T_w can account for the observed threefold difference in sink strength between study periods. We surmise that shading can reduce T_w, a secondary effect rather than a driver, and conclude that T_w did not account for the observed differences between study periods. This agrees with the conclusion of *Rocha and Goulden* [2008] from a study of a freshwater marsh in Southern California, where they found no linkage between weather conditions and the interannual variability of NEE, ER, and GEP.

Strong evidence in this study suggests that the lower photosynthetic activity in the later study period is due to the buildup of senescent plant material of mostly dead *S. acutus* stems, as evidenced by the reduction in latent heat flux and CO_2 uptake. Senescent material can either absorb or reflect almost 100% of the incoming photosynthetic active radiation [*Schile et al*., 2013], preventing it from reaching the water surface and subsequent rhizome systems where young shoots growth is stimulated [*Bonneville et al*., 2008]. In the latter period of this study, it is likely that the presence of senescent material shaded water surfaces and impeded plant growth, consistent with our observations of reduced T_w, lower evaporation and transpiration rates, and resultant lower latent heat flux. This result is similar to *Goulden et al*. [2007], who found that the presence of a thick plant litter mat kept water temperatures lower, reduced rates of evaporation, and lowered overall evapotranspiration. That senescent plant material was more abundant in the later study period is indicated by automated photographic monitoring (Figures S3 and S4) and an evaluation of differences in August NDVI values between study periods (Landsat 7 ETM+; Figure S5). NDVI values within the tower footprint were higher during the earlier study period and declined from 2006 to 2010 (Figure S5), suggesting changes to the canopy structure may have altered interception of photosynthetically active radiation over time, consistent with the observed decline in latent heat flux and CO_2 uptake.

To assess the extent to which accumulated senescent plant material affected surface fluxes, we calculated the Bowen ratio, or the ratio of sensible heat versus latent heat, for all four years during the peak growing season. Average Bowen ratios, calculated from midday values (1000–1400 local time) for the month of August, were close to 0.2 in 2002 and 2003, 1.4 in 2010 and 0.7 in 2011 (Figure 7). The higher Bowen ratios observed in 2010 and 2011 indicate that more of the available energy dissipated as sensible heat. These results are similar to results presented by *Rocha and Goulden* [2008] where higher Bowen ratio values were observed in years

when NEE was lower, suggesting senescent material was a driver of the difference. It is instructive that the Bowen ratio values calculated for this study were lower than results presented in *Rocha and Goulden* [2008], where inundation at their site was seasonal rather than continuous, which would lower the effect of the senescent material on the energy balance. Values in this study are similar to values seen in the Florida Everglades [*Malone et al*., 2014; *Schedlbauer et al*., 2012], where inundation was more comparable.

4.2. Implications

Given the interest in utilizing wetlands for carbon offset and trading programs [*Galatowitsch*, 2009], it is useful to examine both the difference in the annual atmospheric carbon mass balance and the concomitant influence of CH_4 on the GWP balance [*Bridgham et al*., 2006]. *Whiting and Chanton* [1993] estimated that on a molar basis, CH_4 emissions make up a small portion (roughly 3%) of the net daily CO_2 uptake from wetlands globally. However, when GWP is considered, the warming effect of CH_4 emitted from freshwater wetlands can strongly reduce or even exceed the cooling effect of photosynthetic CO_2 uptake [*Brix et al*., 2001; *Herbst et al*., 2011; *Olson et al*., 2013]. This study shows that CH_4 emissions can have a significant influence on both the carbon and GWP balances. The mass of carbon from annual CH_4 emissions measured in 2010–2011 was small (8%) on a molar basis ($56.3 \pm 6.4\ g\ C\ m^{-2}\ yr^{-1}$) compared to the CO_2 uptake in 2002–2003 ($-804.2 \pm 131.5\ g\ C\ m^{-2}\ yr^{-1}$). However, because CO_2 uptake in 2010–2011 was lower ($-21.1 \pm 83.1\ g\ C\ m^{-2}\ yr^{-1}$) than in 2002–2003, inclusion of CH_4 emissions in the overall carbon budget resulted in the wetland becoming a small net carbon source by mass ($35.2 \pm 83.3\ g\ C\ m^{-2}\ yr^{-1}$). Using a GWP for CH_4 that is 34 times that of CO_2 [*Myhre et al*., 2013], the GWP influence of CH_4 emissions in 2010–2011 was $696.4 \pm 79.0\ g\ C\ m^{-2}\ yr^{-1}$, which far exceeded photosynthetic cooling in 2010–2011 and represents a value similar (opposite in sign) to the photosynthetic cooling of 2002–2003. We note that CH_4 emission at the Twitchell Island wetland represents one of the highest rates reported in the literature [*Petrescu et al*., 2015]. This finding adds further support to the growing evidence [e.g., *Chu et al*., 2014] that CH_4 plays a significant role in the carbon balance in freshwater wetlands, especially during periods of lower plant productivity.

To assess the extent to which the GWP balance of the wetland may have differed between study periods, we calculated the prospective annual carbon and GWP balances for 2002–2003 using the CH_4 emission values from 2010 to 2011, i.e., we assumed the CH_4 emission rate was similar for both periods. That this assumption is reasonable is supported by direct measurements previously published from static chamber flux measurements made at this wetland [*Miller*, 2011] showing CH_4 emissions to be relatively invariant, with median midday CH_4 emissions of $13.8\ mg\ C\ m^{-2}\ h^{-1}$ and an interquartile range from 8.2 to $21.1\ mg\ C\ m^{-2}\ h^{-1}$ between March and December 2000 through 2003. These chamber values are similar to midday fluxes measured in this study from April to December 2010, where the mean value was $9.8\ mg\ C\ m^{-2}\ h^{-1}$ and the range was from 2.4 to $29.5\ mg\ C\ m^{-2}\ h^{-1}$. This supports that overall CH_4 emissions were presumably similar between the 2002–2003 and 2010–2011 study periods, and differences in emission ranges are most likely due to methodology and an increase in resolvable fluxes from the EC system.

Further supporting the notion that CH_4 emissions were similar between the two study periods are other studies showing that CH_4 fluxes can vary little over time, even in wetlands undergoing changes in succession or hydrological management. For example, a 6 year study at an unaltered postglacial northern latitude ombrotrophic bog by *Roulet et al*. [2007] showed annual CH_4 to be relatively small and consistent, with a mean of $3.7 \pm 0.5\ g\ C\ m^{-2}\ yr^{-1}$ (±1 standard deviation). During a four year study from 2005 to 2008, *Schrier-Uijl et al*. [2014] observed annual CH_4 fluxes to vary between 13.2 and $15.4\ g\ C\ m^{-2}\ yr^{-1}$, 7 years after the conversion from an intensely agriculturally managed peatland to a grass/wetland polder. Even a newly restored wetland on an adjacent island in the Delta reported mean annual CH_4 fluxes of $50 \pm 2\ g\ C\ m^{-2}\ y^{-1}$ from 2012 to 2014 despite large changes in CO_2 uptake [*Sturtevant et al*., 2016]. Moreover, *Chu et al*. [2014] only observed moderate variation in annual CH_4 fluxes from 42.3 to $57.0\ g\ C\ m^{-2}\ yr^{-1}$ in the case where hydrological management changed to a year-round inundation after decades of periodic lowering of summer water levels at a freshwater marsh on the shores of Lake Erie.

However, some studies report lower or variable CH_4 emission rates during earlier periods of wetland establishment. For instance, low CH_4 emission rates were observed the first year following conversion from corn and alfalfa production to rice cultivation at an adjacent site on Twitchell Island [*Hatala et al*., 2012]. Despite the high carbon peat soils found on Twitchell Island, they attributed the lower initial fluxes to the lack of labile

fresh plant material, but that in subsequent years the incorporation of rice straw was suggested to increase labile carbon and resultant CH_4 emissions [*Knox et al*., 2015; *Hatala et al*., 2012]. Lower CH_4 production was also observed after initial inundation of a degraded fen grassland in NE Germany [*Hahn-Schofl et al*., 2011]. Here lower values were attributed to the competition between methanogens and iron- or sulfate-reducing bacteria and/or continued aerobic conditions in the upper peat soil layers. In this study, we do not believe the previous processes played a significant role as the wetland had been inundated for close to 5 years before EC flux measurements began in 2002 and the emergent macrophytic canopy was over 50% in coverage, providing sufficient labile carbon from fresh and dead plant material as well as root exudates.

Our study shows that in 2010–2011 the radiative forcing associated with the measured CH_4 emissions in combination with the reduced photosynthetic uptake of CO_2 was a large net GWP source to the atmosphere (675.3 ± 114.7 g C m^{-2} yr^{-1}). When we included 2010–2011 CH_4 emissions in the GWP balance for 2002–2003, GWP went from -804.2 ± 131.5 g C m^{-2} yr^{-1} to near equilibrium at -107.8 ± 153.4 g C m^{-2} yr^{-1}. Recomputing these results using a ±20% uncertainty bound on the CH_4 emissions yields a GWP range from −247 to 34 g C m^{-2} yr^{-1}. This exercise shows that even over this broad range of potential variability, CH_4 can dramatically lower or even reverse the GWP balance.

Although simple GWP calculations are appropriate for evaluation of benefits in the context of carbon markets, *Neubauer and Megonigal* [2015] stress that continuous emission sources such as wetlands should be evaluated differently to account for the eventual new steady state in atmospheric concentrations. They introduce a new metric, the sustained flux GWP (SGWP), which varies over time to account for the changing radiative forcing balance in the atmosphere. Evaluating the wetland performance using this new metric reveals that wetland productivity and emissions in the more productive 2002–2003 period result in a SGWP of 1161.2 ± 259.3 g C m^{-2} yr^{-1} at 20 years, 117.1 ± 168.2 g C m^{-2} yr^{-1} at 100 years—the time scale relevant to carbon markets—and becomes a sink at 500 years (-517.6 ± 135.6 g C m^{-2} yr^{-1}). In contrast, using the wetland performance characteristics for the less productive study period, 2010–2011, yields a SGWP of 1944.3 ± 238.4 g C m^{-2} yr^{-1} at 20 years and 900.2 ± 133.7 g C m^{-2} yr^{-1} at 100 years, and still shows the wetland to be a significant SGWP source at 500 years (265.5 ± 89.3 g C m^{-2} yr^{-1}). Presuming that the 2010–2011 period represents future ongoing wetland performance, it appears that this wetland will be a SGWP source when integrated over time scales relevant to carbon offset and trading programs.

It is also important to realize that the EC study presented here did not account for carbon loss due to lateral and seepage flux of dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC), which can be significant [*Bergamaschi et al*., 2012]. We estimated the loss using data from *Fleck et al*. [2007], which examined DOC export from our study site, and based on these calculations, DOC export could account for approximately 12% of the measured cumulative annual NEE in 2002–2003 but a much greater fraction, approximately 36%, in 2010–2011. We could not find data that permitted us to estimate losses due to DIC seepage and export, so that term remains unconstrained but would further reduce the NEE. These results indicate that lateral and seepage fluxes of DOC can be important contributors to the net ecosystem carbon balance and need to be included in wetland monitoring where carbon accounting is a goal.

The trajectory of carbon sequestration potential for this wetland is not clear. It is possible that it will follow a trajectory, where the first few years are highly productive but at some point, the system reaches a steady state condition of reduced rates of CO_2 uptake. This idea is similar to what is described by *Craft et al*. [2003] in their conceptual model of salt marsh ecosystem development or suggested by *Schrier-Uijl et al*. [2014] that after restoration the ecosystem sink potential is higher before stabilizing at a lower level years to decades later; however, interannual variability can continue. After 50 years of restoration of a temperate freshwater wetland and under comparable climatic conditions, *Rocha and Goulden* [2008] showed that vegetation dynamics strongly influenced the variability in annual cumulative NEE from a modest sink to a source.

The initial declines and continued variability observed in the annual wetland CO_2 uptake highlight the need for long-term continuous monitoring of restoration projects. The differences between years and study periods clearly show that sampling restored wetlands such as the one studied here for only 1 or 2 years will likely not provide an accurate assessment of long-term surface energy flux dynamics or GWP balance. This assessment is similar to that of *Roulet et al*. [2007], who noted that using any single year of a 6 year study to assess longer-term wetland carbon sequestration potential would result in errors as high as 4 times the annual flux. Whether a restored wetland becomes a net source or sink over longer periods of time depends on the

average of this considerable interannual variation, which is difficult to control or predict. Based on our observations in 2010–2011, even careful vegetation selection and water management did not lead to consistent annual CO_2 uptake.

5. Conclusion

Restoration of freshwater wetlands can enhance a multitude of ecosystem services, for example, expanding wildlife habitat, improving coastal protection in the face of sea level rise, and reducing atmospheric CO_2 through carbon sequestration. High rates of productivity and lower rates of remineralization in wetlands, such as those shown here, have stimulated interest in potentially using wetlands in biological carbon sequestration projects, perhaps leading to their use in emission reduction and trading programs. The concern this study highlights is that, while wetlands can sequester carbon in their soils, the interannual variability in CO_2 uptake, medium-term changes in wetland production, CH_4 emissions, and the effects of DOC export may make confident prediction or simple evaluation of the annual net GWP balance difficult. Indeed, our results show that intensive continuous monitoring is necessary for quantification of wetland carbon sequestration potential.

This study adds to the considerable evidence showing that reestablishment of wetlands initially results in high plant productivity and a net negative GWP balance, assuming the GWP of CH_4 is 34 times that of CO_2 for a 100 year time horizon. However, reduction in CO_2 uptake, observed here in 2010–2011, alters the overall carbon budget and GWP balance over time. Further, results of our study show that productivity can vary considerably between and among years, making evaluations of wetland performance difficult. Given the observed short- to medium-term variability for this wetland restoration project, it may be challenging to quantify net GWP reductions within the constraints of carbon offset and trading programs.

Acknowledgments

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