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Agricultural reclamation effects on ecosystem CO₂ exchange of a coastal wetland in the Yellow River Delta

Guangxuan Han^a, Qinghui Xing^a, Junbao Yu^{a,*}, Yiqi Luo^b, Dejun Li^b, Liqiong Yang^a,
Guangmei Wang^a, Peili Mao^a, Baohua Xie^a, Nate Mickle^b

^a Key Laboratory of Coastal Zone Environmental Processes and Ecological Remediation, Yantai Institute of Coastal Zone Research, Chinese Academy of Sciences, Yantai, Shandong 264003, China

^b Department of Microbiology and Plant Biology, University of Oklahoma, Norman, OK 73019, USA

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ABSTRACT

Little is known about the impacts of agricultural exploitation of coastal wetlands on ecosystem CO₂ exchange, although coastal wetlands have been widely reclaimed for agricultural use across the world. We measured net ecosystem CO₂ exchange (NEE) and its major components, gross primary production (GPP) and ecosystem respiration (R_{eco}) using an eddy covariance flux technique in a natural coastal wetland (reed) and an adjacent, newly reclaimed farmland (cotton) in the Yellow River Delta, China. The results showed that agricultural reclamation changed the ecosystem CO₂ exchange of the coastal wetland at three distinct levels. Initially, the conversion from the wetland to farmland changed the light response parameters (α , A_{max} , and $R_{\text{eco, day}}$) of NEE and temperature sensitivity (Q_{10}) of R_{eco} mainly by changing the dominant vegetation type. Over the growing season, NEE, R_{eco} and GPP were significantly correlated with LAI at both sites and aboveground biomass at the farmland site. Next, the reclamation of wetland modified the diurnal and seasonal dynamics of ecosystem CO₂ exchange. Significant differences in diurnal variations of NEE between the wetland and farmland sites were found during the growing season (with the exception of June and July). Seasonal means of daily GPP and R_{eco} values at the wetland site were higher than those at the farmland. Ultimately, the agricultural reclamation altered the CO₂ sequestration capacity of the coastal wetland. The cumulative NEE in the wetland ($-237.4 \text{ g C m}^{-2}$) was higher than that in the farmland ($-202.0 \text{ g C m}^{-2}$). When biomass removal was taken into account, the farmland was a strong source for CO₂ of around 131.9 g C m^{-2} during the growing season. Overall, land use changes by reclamation altered ecosystem CO₂ exchange at several ecological scales by changing the dominant vegetation type and altering the ecosystem's natural development.

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1. Introduction

Coastal wetlands, the interfaces between terrestrial and ocean ecosystems, play an important role in the global C cycle by acting as natural carbon (C) sinks (Choi and Wang, 2004; Laffoley and Grimsditch, 2009; Crooks et al., 2011). Globally, at least 430 Tg of C is stored in the upper 50 cm of tidal salt marsh soils (Chmura et al., 2003). Coastal wetlands accumulate organic matter because of their relatively high net primary productivity (NPP) coupled with a relatively low rate of decomposition of accumulated organic matter (Vicari et al., 2011; Crooks et al., 2011; Hu et al., 2012). Furthermore, methane (CH₄) emissions are much lower from coastal wetlands compared to freshwater wetlands, mainly because the relatively high inputs of sulfate from marine waters promote sulfate

reduction and hinders CH₄ production (Poffenbarger et al., 2011; Callaway et al., 2012). Therefore, coastal wetlands could be more valuable C sinks per unit area than other ecosystems because of their higher rates of C sequestration and lower CH₄ emissions (Choi and Wang, 2004; Crooks et al., 2011).

Unfortunately, coastal wetlands are reclaimed for agriculture in many parts of the world (Hassan et al., 2005; Laffoley and Grimsditch, 2009; Crooks et al., 2011). An estimate shows that about 50% of the global wetlands have been exploited for agricultural and other land uses (Verhoeven and Setter, 2010). In China, more than half (approximately 7082.0 km²) of salt marshes have been reclaimed for other land uses, which exceeds the area of China's marshes today (Yang and Chen, 1995). Various studies have shown that reclamation of wetlands for agricultural or other land uses not only halts ongoing C sequestration but releases soil C stores (e.g. Santín et al., 2009; Wang et al., 2010; Crooks et al., 2011). On one hand, reclamation can have a large impact on wetland hydrology, which consequently changes the temperature and anaerobic

* Corresponding author. Tel.: +86 535 2109113; fax: +86 535 2109000.

E-mail addresses: gxhan@yic.ac.cn (G. Han), jbyu@yic.ac.cn (J. Yu).

conditions in wetlands, leading to emissions of greenhouse gases. On the other hand, when the soil layer of wetlands is reclaimed and oxygen enters soils, large amounts of C are released into the atmosphere in the form of CO₂ (Crooks et al., 2011). Moreover, wetland reclamation can influence the quality of soil organic matter through changes in the plant community composition, productivity, or belowground allocation in wetlands (Keller et al., 2004). Thus, it is important to quantify the changes in C cycles caused by the conversion from wetland to farmland. However, little information is available on the effects of reclamation on C budgets in coastal wetlands at ecosystem or regional scales.

As a typical coastal wetland, the Yellow River Delta is one of the most active regions of land–ocean interaction in the world. In most of the area, groundwater levels range from 1 to 3 m, with high water salinity (5–30 g L⁻¹) (Yang et al., 2009; Fan et al., 2011). Driven by strong evaporation, soluble salt in the shallow water table is transported upward to the root zone and soil surface through capillary rise (Yao and Yang, 2010; Zhang et al., 2011). Consequently, the Yellow River Delta is naturally characterized by extensive coverage of saline soils, which accounts for 47–70% of the land area of the Yellow River Delta (Xie et al., 2011). Opposing these effects, the freshwater inflow from river runoff and net precipitation can leach salts from the plant root zone in coastal wetlands. Therefore, under the combined action between freshwater and seawater, as well as between groundwater and surface water, the vegetation in the coastal wetland exhibits a patchy distribution, and the natural vegetation is composed of aquatic and halophytic plant communities, dominated by herb and shrub species (Fan et al., 2011; Xie et al., 2011; Zhang et al., 2011).

Meanwhile, as a significant agricultural production base, the Yellow River Delta has been undergoing extensive and rapid development of agriculture over recent decades (Zhang et al., 2011; Huang et al., 2012). Since 1981, farmers have been cultivating cotton and maize in fields with low salinity (Yao and Yang, 2010), and a large portion of coastal wetlands have been converted to dry lands for salt-tolerant crops in recent years (Huang et al., 2012). For example, over the past 20 years natural wetlands land cover has decreased by 38.6% from 2566 km² in 1986 to 1575 km² in 2008 (Wang et al., 2012). Among these natural wetlands, the marsh wetland was reduced by 65.09 km² during the period from 1986 to 2005 (Huang et al., 2012). As previously described, at both regional and global scales, wetland conversion to agricultural lands has a great effect on C cycles, especially net ecosystem CO₂ exchange (NEE). Studies have also been undertaken concerning changes of soil C content in reclaimed wetlands in the Yellow River Delta (e.g. Yang et al., 2009; Huang et al., 2012; Yu et al., 2012). However, to date few studies have focused on the effects of reclamation effect on NEE and its controlling factors in coastal wetlands.

The eddy covariance (EC) technique is the most widespread and commonly used method for the spatial integration of CO₂ fluxes at landscape scales and is also applied to measure the CO₂ exchange between terrestrial ecosystems and the atmosphere. Therefore, the EC methods can improve our understanding of the impact of reclamation on ecosystem scale C budgets, since it can provide continuous, long-term flux information integrated at the ecosystem scale (Baldocchi, 2003). In this study, we set up paired EC flux towers to measure NEE in two adjacent ecosystems, a natural coastal wetland (reed) and a newly reclaimed farmland (cotton) in the Yellow River Delta, China. The objectives of this study were (1) to determine the diurnal and seasonal variation of ecosystem CO₂ exchange and calculate the C balance during the growing season for a wetland and a neighboring farmland; (2) to quantify and compare the ecosystem CO₂ exchange of the two ecosystems response to meteorological conditions and phenological development; (3)

to investigate the impact of reclamation of a natural wetland on ecosystem CO₂ sequestration.

2. Materials and methods

2.1. Site description

The study was conducted in Yellow River Delta Ecological Research Station of Coastal Wetland (37°45'50"N, 118°59'24"E), Chinese Academy of Sciences. The experimental site has a warm-temperate and continental monsoon climate with distinctive seasons. The annual mean temperature is 12.9°C, with minimum and maximum mean daily temperatures of -2.8°C in January and 26.7°C in July, respectively. The average annual precipitation is 550–640 mm, with nearly 70% of the precipitation falling between May and September. The prevailing wind direction in the growing season is from the northeast to the southeast (Han et al., 2013). Soil type in the Yellow River Delta gradually varies from fluvo-aquic to saline soil, and the soil texture is mainly sandy clay loam. Due to the flat terrain and high groundwater table, the entire area is covered mainly by wet and saline soil. The natural vegetation in this area consists of salt tolerant herbs, grasses, and shrubs. The gradually increasing groundwater table and salinity from inland to the coast result in a vegetation gradient from aquatic to halophytic communities (Fan et al., 2011).

Based on the current and historic land use, two study sites, a natural coastal wetland and a recently reclaimed farmland were chosen within the study area. The distance between the two towers is 800 m. The terrain at the two sites is flat with a sufficient fetch to meet the basic assumption for proper application of the EC technique. The coastal wetland ecosystem was dominated by *Phragmites communis*, *Suaeda heteroptera*, *Apocynum venetum*, *Sonchus brachyotus* and *Tamarix chinensis* community. During our study, its canopy was 0.3–1.7 m high, with the canopy closure index ranging from 0.3 to 0.8. The growing season of the coastal wetland ecosystem spans from May to October. During the rainy season (mid-July to mid-August), surface ponding (often less than 5 cm) was often observed in the coastal wetlands for time periods of less than 10 days (Han et al., 2013). The farmland, which was reclaimed in April 2008, was used for planting of cotton (*Gossypium hirsutum* L.). Cotton was planted in early May at a moderate density (5.3 plants m⁻²) and harvested at the end of October in the Yellow River Delta. After sowing, soils were mulched with transparent plastic film (0.008 mm) along the rows, which can reduce salt accumulation and moisture loss in the soils. The maximum canopy height at the peak of the growing season (early July to mid-August) reached up to 1.50 m. After cotton was picked, the cotton plants including roots were harvested and taken away from the fields in order to control pests and diseases or to be burned as domestic fuel. The soil physical and chemical properties in the natural wetland and the reclaimed farmland are presented in Table 1.

2.2. Eddy covariance and meteorological measurements

Eddy covariance and microclimate measurements were conducted at both research sites during the 2011 growing season (May–October). More details about the monitoring system are presented elsewhere (Han et al., 2013). Ecosystem CO₂ fluxes were measured using a paired EC system mounted 3.0 and 2.8 m above the soil surface for the coastal wetland and farmland ecosystems, respectively. The EC system included a three-axis sonic anemometer (CSAT-3, Campbell Scientific Inc., USA) and open path infrared gas analyzer (IRGA, LI-7500, Li-COR Inc., USA). Calculations with a footprint model (Hsieh et al., 2000) indicated that approximately 86% and 70% of the cumulative flux footprint originated within

Table 1

Soil physical and chemical properties in the natural wetland and the reclaimed farmland in the Yellow River Delta.

Site	Depth (cm)	pH	Salinity (‰)	TOC (%)	Total C (%)	Total N (%)	NH ₄ ⁺ -N (mg kg ⁻¹)	NO ₃ ⁻ -N (mg kg ⁻¹)
Wetland	10	7.77 ± 0.07	0.95 ± 0.09	0.65 ± 0.04	1.47 ± 0.11	0.07 ± 0.00	0.55 ± 0.04	1.40 ± 0.07
	20	8.00 ± 0.03	0.54 ± 0.02	0.30 ± 0.01	1.49 ± 0.08	0.05 ± 0.00	0.47 ± 0.01	0.53 ± 0.03
Farmland	10	8.04 ± 0.05	0.45 ± 0.00	0.77 ± 0.05	1.75 ± 0.15	0.08 ± 0.01	0.64 ± 0.02	1.37 ± 0.02
	20	8.19 ± 0.02	0.22 ± 0.01	0.52 ± 0.05	1.48 ± 0.03	0.07 ± 0.00	0.55 ± 0.02	0.83 ± 0.02

TOC = total organic carbon, Total C = total carbon content, Total N = total nitrogen content, NH₄⁺-N = extractable ammonium nitrogen, NO₃⁻-N = extractable nitrate nitrogen. The average values and standard error are presented ($n = 3$).

200 m upwind of the towers in coastal wetland and farmland ecosystems, respectively. For each of the two flux towers, the fetch from all directions was more than 300 m, so the majority of the measured fluxes came from each of the two ecosystems. The flux data were recorded at 10 Hz by a datalogger (CR1000, Campbell Scientific Inc., USA) at 30 min intervals. In both the coastal wetland and farmland ecosystems, meteorological parameters were measured simultaneously with the same array of sensors, including net radiation (CNR4, Kippand Zonen USA Inc., Bohemia, NY, USA), photosynthetically active radiation (PAR) (LI-190SB, Li-Cor Inc., USA), air temperature (HMP45C, Vaisala, Helsinki, Finland), wind speed and direction (034B, Campbell Scientific Inc., USA), and precipitation (TE525 tipping bucket gauge, Texas Electronics, Texas, USA). Soil temperature was measured at five depths (5, 10, 20, 30 and 50 cm) with thermistors (109SS, Campbell Scientific Inc., USA). Soil volumetric water content was measured by time domain reflectometry probes (EnviroSMART SDI-12, Sentek Pty Ltd., USA) at seven depths (5, 10, 20, 40, 60, 80 and 100 cm). All meteorological data were measured every 15 s and then averaged half-hourly.

2.3. Aboveground biomass and leaf area index measurements

During the vegetative growth period (May–October), the aboveground biomass (AGB) for the coastal wetland and farmland were measured by harvesting the vegetation approximately every 2 weeks. At each site, five 0.5 m × 0.5 m squares were randomly chosen within a radius of 200 m around the observation tower. Live plants were clipped at 1 cm above the ground level. Plant AGB was oven dried at 80 °C for 48 h and weighed. The leaf area index (LAI) of the dominant species at the two sites was also measured using a portable meter (LI-2000; Li-Cor, Inc.). LAI was determined at 2-week intervals at the two sites. The sampling plot was 0.5 m × 0.5 m, and five replicates were taken on each measurement day.

2.4. Flux data processing and quality control

Raw data were processed using the post-processing software EdiRe (University of Edinburg, Scotland) to determine NEE with an averaged half-hourly period. Data processing followed standard methods and included despiking, coordinate rotation, time lag corrections, and Webb–Pearman–Leuning (WPL) correction (Webb et al., 1980). Subsequently, quality filtering was applied to the half-hour flux data according to the following rejection criteria: (1) incomplete half-hour measurements; (2) excessive spikes in the sonic and IRGA data; (3) precipitation or condensation on the IRGA or sonic anemometer; (4) biologically impossible values of NEE for the reed wetlands ($|NEE| > 60 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) (Zhou et al., 2009; Han et al., 2013); (5) the flux data under nocturnal low atmospheric turbulence conditions were excluded based on friction velocity (u^*). Nighttime CO₂ fluxes were filtered when $u^* < 0.15 \text{ m s}^{-1}$ (Han et al., 2013). Negative nighttime CO₂ fluxes were also removed from the datasets. After post-processing and quality filtering, 63.0% of the CO₂ flux data for the coastal wetland (81.8% from daytime, 44.2% from nighttime) and 64.5% (74.0% from daytime, 55.0% from nighttime) for the farmland were suitable for analysis.

2.5. Flux gap filling

In order to provide estimates for the balance of NEE, the data gaps were filled with the following procedure. Small gaps (<2 h) were filled by linear interpolation using the neighboring measurements. For large gaps (≥ 2 h), the missing NEE data were filled based on empirical models separately for daytime and nighttime data (Gao et al., 2012; Han et al., 2013).

The missing daytime NEE data during the growing season were gap-filled using the Michaelis–Menten model (Falge et al., 2001):

$$NEE = -\frac{A_{\max}\alpha PAR}{A_{\max} + \alpha PAR} + R_{\text{eco,day}} \quad (1)$$

where the coefficient α is apparent quantum yield ($\mu\text{mol CO}_2 \mu\text{mol}^{-1} \text{ photon}$), A_{\max} is light-saturated net CO₂ exchange ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), and $R_{\text{eco,day}}$ is the ecosystem respiration in the daytime ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$). A_{\max} and $R_{\text{eco,day}}$ are fitted parameters of the observed valid data using the nonlinear regression from SPSS 13.0.

The missing nighttime NEE, which is equivalent to nighttime ecosystem respiration ($R_{\text{eco,night}}$), were filled using the van't Hoff (Lloyd and Taylor, 1994) empirical exponential function:

$$R_{\text{eco,night}} = a \exp(bT_{\text{soil}}) \quad (2)$$

where $R_{\text{eco,night}}$ is nighttime NEE (i.e., ecosystem respiration), T_{soil} is soil temperature at depth of 5 cm, a and b are two empirical coefficients.

Q_{10} can be estimated as

$$Q_{10} = \exp(10b) \quad (3)$$

Based on the assumption that daytime respiration ($R_{\text{eco,day}}$) was of similar magnitude and responsiveness as $R_{\text{eco,night}}$, $R_{\text{eco,day}}$ was determined using the predictive relationships developed for nighttime periods (Falge et al., 2001; Alberto et al., 2009; Zhou et al., 2009; Schedlbauer et al., 2010). Using the $R_{\text{eco,day}}$ values, GPP was calculated as the balance between daytime NEE and $R_{\text{eco,day}}$:

$$GPP = R_{\text{eco}} - NEE \quad (4)$$

Due to seasonal changes in plant biomass and meteorological conditions, the NEE–PAR and $R_{\text{eco}}-T_{\text{soil}}$ relationships vary in time. Therefore, the relationships were separately derived monthly during the growing seasons. In cases where empirical relationships could not be developed as a result of missing meteorological data, mean diurnal variation was used to fill the gap (Falge et al., 2001).

2.6. Statistical analysis

A paired samples t -test was used to test the significant differences in environmental and biotic factors (SWC, AGB, and LAI), diurnal variations of NEE, and the mean daily values of NEE, GPP, and R_{eco} between the wetland and the farmland. In all tests, a significance level of 0.05 was used. Exponential regression analysis was used to examine the relationships between nighttime NEE (R_{eco}) and soil temperature under different SWC levels at both

sites. Nonlinear regression analyses were used to describe the relationships between ecosystem CO₂ flux (NEE, R_{eco} and GPP) and vegetation development indices (AGB and LAI). All statistical analyses were performed using SPSS 11.5 (SPSS for Windows, Version 11.5, Chicago, IL, USA).

3. Results

3.1. Meteorological and environmental conditions

The temporal pattern of PAR was similar for the wetland and farmland. Daily PAR reached its maximum in late June and then decreased gradually (Fig. 1a). Average monthly PAR reached its maximum in June (689.0 μmol m⁻² s⁻¹), and then decreased gradually (down to 340.3 μmol m⁻² s⁻¹ in October). Mean daily soil temperature at the 5 cm depth in the wetland was 22.2 °C and was 2.2 °C higher than mean daily soil temperature in the farmland (Fig. 1b). Soil temperature was significantly greater in the wetland than in the farmland from May to August, but there was no significant difference between the two sites from September to October. Monthly precipitation (mm) showed a large seasonal variation ranging from 10.4 to 215.3 mm (Fig. 1c). The total precipitation during the growing season was 496.2 mm and the greatest daily rainfall of 71.8 mm occurred on 6 August because of Typhoon Damrey (Fig. 1c). The seasonal patterns of SWC were similar for the two sites, but mean SWC at the wetland (47.9%) was significantly greater (P < 0.001) than in the farmland (39.9%) (Fig. 1c).

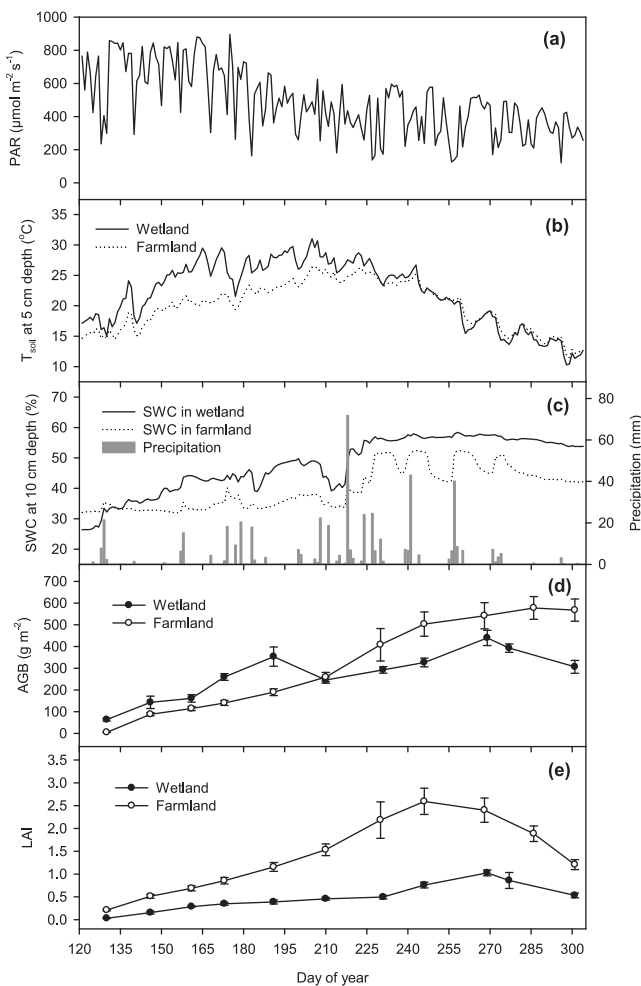


Fig. 1. Seasonal variations of (a) daily photosynthetic available radiation (PAR), (b) daily soil temperature (T_{soil}) at 10 cm depth, (c) daily precipitation and daily mean soil water content (SWC) at 5 cm depth, (d) aboveground biomass (AGB) and (e) leaf area index (LAI) during the growing season (from May to October) at the wetland and farmland sites in the Yellow River Delta. Values of AGB and LAI represent the mean \pm S.E., $n = 5$.

3.2. AGB and LAI

The AGB at the two sites increased through the growing season until a maximum value was reached on October 4 at the wetland site or October 13 at the farmland site, followed by a decline as the community senesced (Fig. 1d). The measured maximum AGB was 440.0 and 577.8 g m⁻² for the wetland and farmland, respectively. From May to July, the AGB of the wetland (204.6 g m⁻²) was significantly higher than that of the farmland (133.0 g m⁻²) (P < 0.001), however, from August to October, the farmland (519.9 g m⁻²) had a higher AGB than the wetland (351.7 g m⁻²) (P < 0.001) (Fig. 1d). The seasonal dynamics of LAI showed one-peak patterns at both sites (Fig. 1e). LAI began to increase in early May and reached the maximum of 2.59 at the farmland site on September 3 and 1.03 at the wetland site on September 26, respectively. Then, LAI decreased sharply in September for the wetland site and October for the farmland site as a consequence of senescence (Fig. 1e). The farmland had a significantly higher LAI than the wetland throughout the growing season (P < 0.001), and the averaged LAI at the farmland site (1.38) was 2.5 times that at the wetland site (0.49).

3.3. Diurnal patterns of NEE

Diurnal patterns of NEE clearly showed significant inter-seasonal and inter-site differences (Fig. 2). At both sites, diurnal patterns of NEE among different month were very similar in shape but varied substantially in amplitude. From May to October, average NEE for each month was negative (a CO₂ sink) during the daytime and positive (a CO₂ source) during the nighttime for both wetland and farmland sites. After sunrise, the NEE moved from a positive value (release) to a negative value (uptake), and CO₂ uptake increased gradually till peak values occurred between 11:00 and 14:00 h. Then, CO₂ uptake declined rapidly through the afternoon, and turned to a release of CO₂ after sunset. At the wetland site, the peak CO₂ release and uptake rates occurred in July and were 3.48 and -9.94 μmol CO₂ m⁻² s⁻¹, respectively (Fig. 2c). At the farmland site, however, the peak CO₂ release and uptake were 2.85 and -10.20 μmol CO₂ m⁻² s⁻¹ occurred in August, respectively (Fig. 2d). The length of duration of positive and negative values changed clearly among different months because of their differences in photoperiod (the time between sunrise and sunset). For example, the duration of negative values during the main growth season (July and August) were longer than those during the early and late growing season (May or October).

Significant differences in diurnal variations of NEE between the wetland and farmland sites were found in May, August, September and October (P < 0.001). During the early vegetative stage, the wetland had more negative daytime NEE than the farmland because cotton was seeded in early May and the plants were relatively small (Height < 0.25 m in May). Due to the rapid growth of the cotton from June to July (the average plant height was > 0.9 m at the end of July), there was no significant difference between the mean diurnal patterns for NEE observed at the two sites (P = 0.10 in June; P = 0.16 in July). However, from August to October, the NEE in the wetland became less negative than that in the farmland (both P < 0.001), because the AGB and LAI at the farmland sites were higher than those at the wetland site (Fig. 1d, e).

3.4. Seasonal variation of ecosystem CO₂ exchange

During the growing season, the daily NEE, R_{eco} and GPP showed distinct seasonal patterns at both sites (Fig. 3), which indicates their responses to the combined effects of weather and vegetation growth. At the beginning of the growing season (early May), because the biomass and LAI in the wetland and farmland were still small (Fig. 1d, e), the daily GPP and R_{eco} of the two ecosystems were much lower, and the daily NEE values at both sites were near zero or slightly positive (release CO₂ to the atmosphere) (Fig. 3). With biomass development and temperature increase, GPP and R_{eco} at these two sites began to increase gradually and reached their peak values during July and August. The maximum GPP (7.41 and 7.92 g C m⁻² d⁻¹ at the wetland and farmland sites, respectively) both occurred on August 22 when the LAIs at both sites were relatively high (Fig. 1e) and soil temperatures rose to 24 °C (Fig. 1b). In addition, daily R_{eco} at these two sites also peaked in July and August with biomass development and high temperature. Maximum daily R_{eco} was 4.02 g C m⁻² d⁻¹ at the wetland site on July 24, and 3.14 g C m⁻² d⁻¹ on August 11 were obtained at the farmland site. Along with the rise of LAI (Fig. 1e), the net CO₂ absorption of the two ecosystems increased and reached the peak period of CO₂ uptake during the peak growing season. In the wetland, the peak period of CO₂ uptake was from June to August with a maximum value of NEE approaching -5.45 g C m⁻² d⁻¹ on August 22, while that of the farmland was from July to August with the maximum net uptake of C of -5.34 g C m⁻² d⁻¹ occurring on August 17 (Fig. 3, Table 2). The peak growing season (from June to August) coincided with higher precipitation (18.2, 17.9 and 24.5 mm on 23 June, 2 July, and 15 August, respectively), leading to a decrease in GPP of the two ecosystems and several shifts from being carbon sinks to short-term CO₂ emissions (Fig. 3). During September and October, GPP, R_{eco} and net CO₂ uptake of these two ecosystems was reduced, apparently with the decrease of solar radiation and temperature. During the late senescence stages (from October 8 to October 31) when soil temperatures decreased to 15 °C and all the leaves were yellow or had fallen, NEE started to become less negative as GPP decreased, so that each of these two ecosystems shifted into net CO₂ release (Fig. 3).

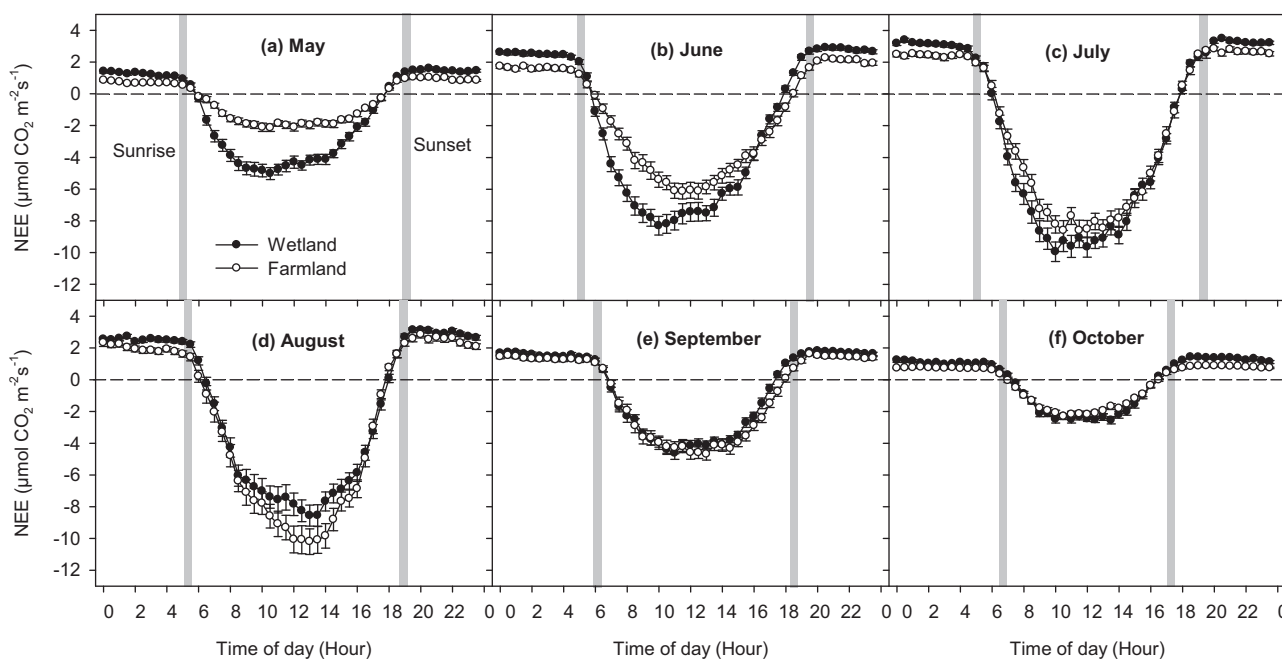


Fig. 2. Diurnal patterns of monthly averaged net ecosystem CO₂ exchange (NEE) at the wetland and farmland sites in the Yellow River Delta during the growing season of 2011. Grey bars denote the monthly varying times of sunrise and sunset.

In 2011, the mean daily NEE values at the wetland site were higher in the early growing periods (May and June) and lower in the mid-growing period (August) than those at the farmland site ($P < 0.05$) (Table 2); whereas there were no differences in NEE between the two ecosystems during July, September and October ($P > 0.05$) (Table 2). On a daily basis, distinctly different seasonal variations in R_{eco} can be seen between the two sites during the entire growing season (Fig. 3). In each month, R_{eco} in the wetland was higher than that in the farmland ($P < 0.05$) (Table 2). From May to July, GPP in the wetland was higher than that in the farmland, but slightly

lower during the late senescence stage (October). However, there were no significant differences in GPP between the two ecosystems in other months (August and September).

3.5. Cumulative carbon balance during the growing season

During the growing season from May to October, the number of net sink days in the wetland and the farmland were 157 and 164 days, resulting in respective cumulative NEE values of -237.4 and $-202.0 \text{ g C m}^{-2}$ (Fig. 4). Although both sites functioned as net CO₂ sink and the magnitudes of net CO₂ uptake were close, the cumulative NEE in the wetland was consistently higher than that in the farmland throughout the entire growing season. Thus, the conversion from wetland to farmland caused a decrease in C sink of approximately 35 g C m^{-2} during the growing season. Furthermore, the cumulative assimilation and respiration values in these two ecosystems showed distinctly different patterns (Fig. 4). Approximately 585.7 and 494.8 g C m^{-2} were assimilated by GPP, and 348.3 and 292.7 g C m^{-2} were released by R_{eco} at the wetland and farmland sites, respectively. The cumulative GPP and R_{eco} at the wetland site were both 1.2 times higher than those at the farmland site. Monthly R_{eco} /GPP ratio ranged from 0.54 to 0.80 in the wetland and from 0.52 to 0.78 in the farmland. For the entire growing season, the ratio of R_{eco} /GPP in wetland was 0.62 while it was 0.63 for the farmland, supporting other results that these two ecosystems were acting as net C sinks.

4. Discussions

4.1. Daytime NEE in response to PAR

During each month of the growing season, the response of daytime NEE to PAR can be expressed by a rectangular hyperbolic function (Eq. (1)) at both sites. During the main growing season (from June to September), PAR explained more than 45% of the variability in half-hourly NEE at the both sites. Conversely, during the early and late growing season (May or October), PAR accounted for less of the variation in NEE and was generally lower with low temperature and a low LAI (Table 3). Altogether, the changes in PAR explained 38% and 23% of the variations in NEE for the wetland and farmland ecosystems, respectively (both $P < 0.001$) (Table 3).

The light response parameters in each month at both sites, including apparent quantum yield (α), light-saturated net CO₂ exchange (A_{max}) and ecosystem respiration in the daytime ($R_{eco,day}$), are shown in Table 3. The seasonal variations of parameters (α , A_{max} and $R_{eco,day}$) could be represented

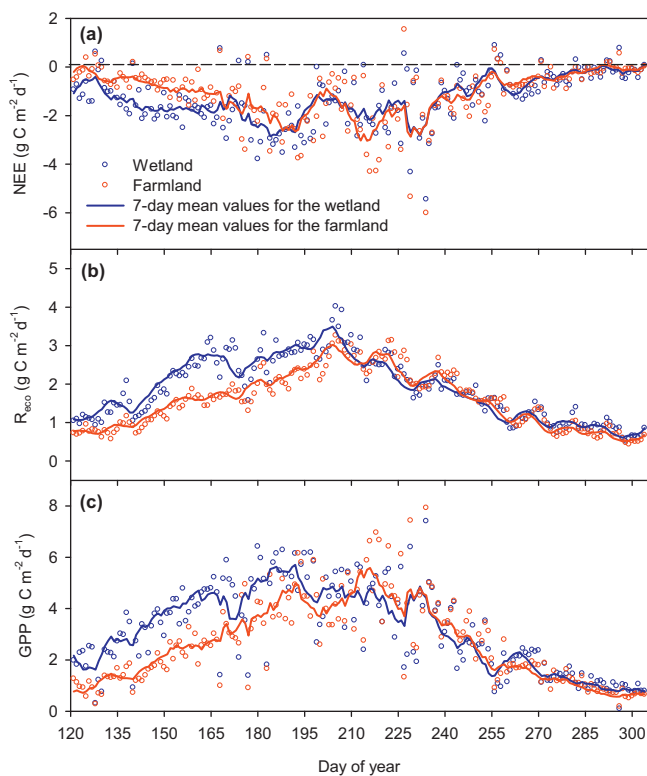


Fig. 3. Seasonal variation in daily (a) NEE, (b) R_{eco} and (c) GPP in growing season of 2011 at the wetland and farmland sites in the Yellow River Delta. Solid lines represent 7-day running mean values.

Table 2
 Monthly mean values of NEE, R_{eco} and GPP in growing season of 2011 at the wetland and farmland sites in the Yellow River Delta.

Month	NEE ($g C m^{-2} d^{-1}$)		R_{eco} ($g C m^{-2} d^{-1}$)		GPP ($g C m^{-2} d^{-1}$)		F-value	R_{eco}/GPP		F-value
	Wetland	Farmland	Wetland	Farmland	Wetland	Farmland		Wetland	Farmland	
May	-1.17 ± 0.13 a	-0.40 ± 0.06ae	1.36 ± 0.06 a	0.89 ± 0.03 a	2.53 ± 0.17 a	1.29 ± 0.08 a	71.1***	0.54	0.69	51.3***
June	-1.81 ± 0.17 b	-1.22 ± 0.15 b	2.51 ± 0.07 b	1.66 ± 0.04 b	4.31 ± 0.21 b	2.88 ± 0.16 b	128.0***	0.58	0.58	34.4***
July	-1.97 ± 0.19 b	-1.78 ± 0.19 c	2.99 ± 0.07 c	2.44 ± 0.07 c	4.95 ± 0.20 c	4.22 ± 0.18 c	55.6***	0.60	0.58	13.2**
August	-1.89 ± 0.23 b	-2.23 ± 0.28 d	2.26 ± 0.07 b	2.41 ± 0.06 c	4.15 ± 0.24 b	4.64 ± 0.28 c	19.2***	0.54	0.52	0.4
September	-0.69 ± 0.12 c	-0.75 ± 0.12ab	1.39 ± 0.06 d	1.43 ± 0.06 d	2.08 ± 0.12 a	2.17 ± 0.14 d	6.7	0.67	0.66	0.2
October	-0.21 ± 0.06 d	-0.20 ± 0.05 e	0.86 ± 0.05 e	0.71 ± 0.03 a	1.07 ± 0.06 d	0.91 ± 0.06 a	44.3***	0.80	0.78	8.1**
Growing season	-1.29 ± 0.08	-1.10 ± 0.09	1.89 ± 0.06	1.59 ± 0.05	3.18 ± 0.12	2.69 ± 0.12	13.8***	0.62	0.63	7.9*

Different letters indicate significant difference ($P < 0.05$) among months at each site. Asterisks indicate differences in the monthly mean values between the wetland and farmland.

* $P < 0.05$, *t*-test.
 ** $P < 0.01$, *t*-test.
 *** $P < 0.001$, *t*-test.

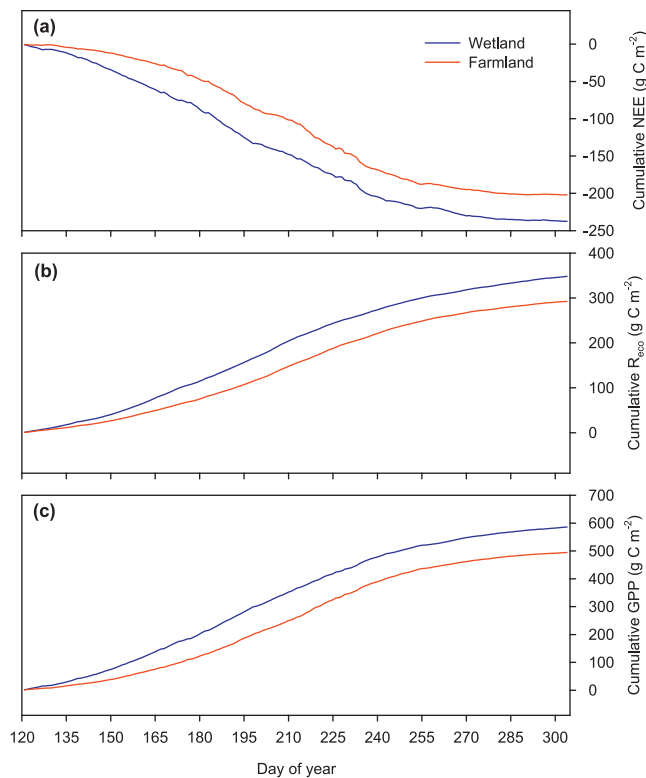


Fig. 4. Cumulative flux for (a) NEE, (b) R_{eco} and (c) GPP at the wetland and farmland sites in the Yellow River Delta during the 2011 growing season.

as single peak curves and be described by quadratic models, which were similar to those observed at other wetland ecosystems of Glenn et al. (2006), Syed et al. (2006), Zhou et al. (2009), and Hao et al. (2011). During the growing season, the quantum yield α was $0.033 \mu mol CO_2 \mu mol^{-1}$ photons ranging from 0.010 to $0.058 \mu mol CO_2 \mu mol^{-1}$ photons at the wetland site, which was close to an estuarine wetland (from 0.009 to $0.086 \mu mol CO_2 \mu mol^{-1}$ photons) in coastal Shanghai (Guo et al., 2009) and a freshwater tidal wetland (from 0.0098 to $0.036 \mu mol CO_2 \mu mol^{-1}$ photons) in Northeast China, and higher than other EC studies in Alpine wetland ecosystems in China (e.g. Hirota et al., 2006; Hao et al., 2011). The monthly mean A_{max} was from 7.77 to $20.15 \mu mol CO_2 m^{-2} s^{-1}$ at the wetland site, within the range of A_{max} (10 – $44 \mu mol CO_2 m^{-2} s^{-1}$) in coastal/estuarine wetlands given by other studies (e.g. Kathilankal et al., 2008; Yan et al., 2008; Guo et al., 2009; Zhou et al., 2009). For example, the A_{max} values of 15 – $29 \mu mol CO_2 m^{-2} s^{-1}$ were observed in a tidal wetland of Northeast China (Zhou et al., 2009). The $R_{eco, day}$ values for the wetland were in the range 1.39 – $6.36 \mu mol CO_2 m^{-2} s^{-1}$, which changed with plant growth and temperature because of their link to photosynthesis (Han et al., 2013). Compared to the wetland ecosystem, the farmland had lower A_{max} (5.13 – $20.30 \mu mol CO_2 m^{-2} s^{-1}$), indicating larger capacity for C uptake in the wetland than in the farmland. From May to July, the magnitudes of α and $R_{eco, day}$ for the farmland were much lower than those for the wetland, however, from August to October they were higher in the farmland than those in the wetland, which might result from the differences in vegetative growth in the two ecosystems.

4.2. Nighttime NEE response to soil temperature under different SWC levels

During the entire growing season, an exponential relationship existed between nighttime NEE (R_{eco}) and soil temperature at 5 cm

Table 3

Parameters of the rectangular hyperbolic response of daytime net ecosystem CO₂ exchange (NEE) to photosynthetically active radiation (PAR) as described in Eq. (1) during the growing season (from May to October) at the wetland and the farmland sites.

Vegetation	Month	α ($\mu\text{mol CO}_2 \mu\text{mol}^{-1} \text{ photons}$)	A_{max} ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$)	$R_{\text{eco, day}}$ ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$)	n	r^2	P -value
Wetland	May	0.012 ± 0.005	9.53 ± 0.57	1.52 ± 1.02	493	0.39	<0.0001
	June	0.038 ± 0.015	16.81 ± 1.45	5.64 ± 1.97	660	0.45	<0.0001
	July	0.058 ± 0.018	20.15 ± 1.31	6.36 ± 1.80	665	0.49	<0.0001
	August	0.041 ± 0.010	18.51 ± 0.72	4.36 ± 1.14	588	0.58	<0.0001
	September	0.015 ± 0.003	9.93 ± 0.72	1.39 ± 0.48	445	0.59	<0.0001
	October	0.010 ± 0.003	7.77 ± 3.20	1.48 ± 0.39	373	0.55	<0.0001
	Growing season	0.033 ± 0.005	14.24 ± 0.41	3.78 ± 0.60	3225	0.38	<0.0001
Farmland	May	0.009 ± 0.004	5.64 ± 0.33	1.82 ± 0.57	476	0.44	<0.0001
	June	0.025 ± 0.010	13.21 ± 0.98	4.48 ± 1.47	656	0.45	<0.0001
	July	0.039 ± 0.012	17.00 ± 0.92	4.49 ± 1.37	686	0.46	<0.0001
	August	0.044 ± 0.011	20.30 ± 0.83	4.45 ± 1.31	636	0.52	<0.0001
	September	0.063 ± 0.012	10.88 ± 1.61	4.76 ± 1.84	511	0.49	<0.0001
	October	0.011 ± 0.003	5.13 ± 0.22	1.29 ± 0.37	551	0.42	<0.0001
	Growing season	0.037 ± 0.009	11.82 ± 0.65	3.87 ± 0.85	3516	0.23	<0.0001

Parameters, α is the apparent quantum yield; A_{max} is the light-saturated net CO₂ exchange; $R_{\text{eco, day}}$ is the ecosystem respiration in the daytime; n is the number of half hourly measurement; r^2 is the coefficient of determination; P -value indicates the level of significance.

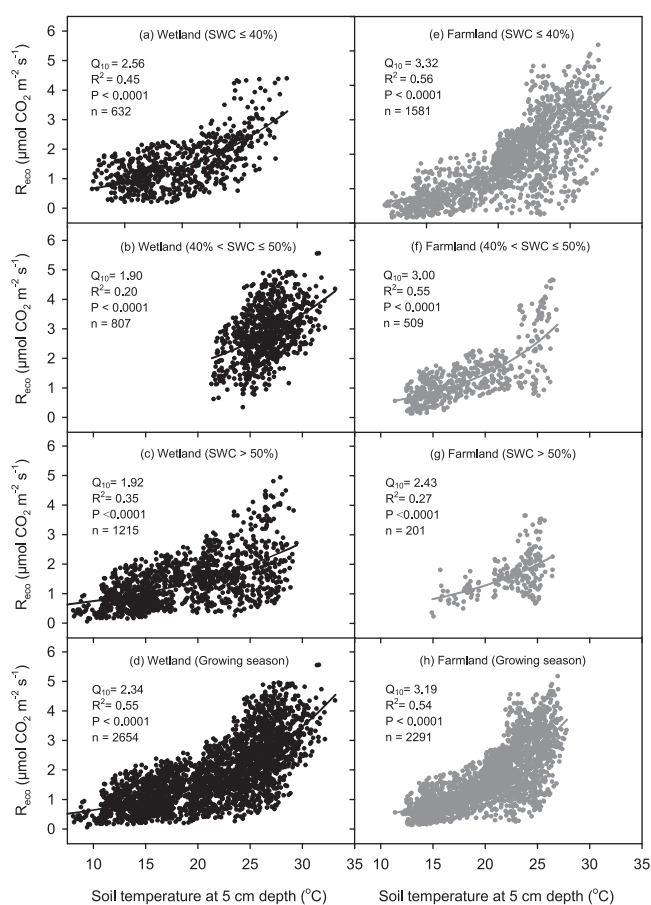


Fig. 5. Relationship between nighttime NEE (R_{eco}) and soil temperature at 5 cm depth under different SWC during the 2011 growing season at the wetland site (a, b, c, d) and farmland site (e, f, g, h). The regression curves were fitted with Eq. (2) based on the observed data.

depth at both sites (Fig. 5d, h). Soil temperature accounted for variation in R_{eco} of approximately 55% at the wetland site and 54% at the farmland site, respectively. Many studies have shown that ecosystem respiration in wetlands was positively related to temperature and could be expressed by the exponential function (Bonneville et al., 2008; Zhou et al., 2009; Schedlbauer et al., 2010). For the growing season, the Q_{10} of R_{eco} was estimated to be 2.3 for the wetland and 3.2 for the farmland. Comparison of Q_{10} in the two

ecosystems suggests that the farmland was more sensitive to the change of temperature than the wetland probably because of the higher biomass in the farmland. Previous studies in other ecosystems have shown that R_{eco} increased with increasing biomass (Xu and Baldocchi, 2004; Shimoda et al., 2009). The AGB of the farmland was much higher than that of the wetland ($P < 0.01$) (Fig. 1d, e), which could explain its higher Q_{10} value. The Q_{10} in the wetland during the growing season was higher than the value obtained in a short-hydroperiod Florida Everglades marsh (Schedlbauer et al., 2010), a large peatland (Bubier et al., 2003), and aerobic rice fields (Alberto et al., 2009), and lower than the values obtained by the studies in a deep-water wetland (Hirota et al., 2006) and a temperate cattail marsh (Bonneville et al., 2008), but was within the range (1.8–8.9) given in other observations in wetlands (Zhou et al., 2009; Zhao et al., 2010).

Due to frequent rain events, SWC was always higher than 26% at the wetland site and 31% at the farmland site during the growing season (Fig. 1c). Moreover, during the rainy season (mid-July–mid-August), surface ponding (often less than 5 cm) was often observed at both sites for time periods of less than 10 days (Han et al., 2013). Thus, during the entire growing season, the available soil water was sufficient or high at both sites. In order to further examine the effect of SWC on the response of R_{eco} to soil temperature, the nighttime NEE data were separated into three SWC classes (medium SWC $\leq 40\%$, high 40% < SWC $\leq 50\%$, and surface ponding SWC > 50%). The results showed that Q_{10} evidently declined with increases in soil moisture at both sites, probably because the available soil water was sufficient or causing soil surface ponding. The highest Q_{10} appeared in relatively moderate soil moisture condition (SWC $\leq 40\%$), and Q_{10} declined in response to an increase in SWC (40% < SWC $\leq 50\%$). When soil surface ponding occurred (SWC > 50%), Q_{10} was the minimum at both sites (Fig. 5c, g).

Our results clearly illustrate that soil moisture can also have an important influence on the Q_{10} value of ecosystem respiration, and the same results have been reported in other previous studies. The Q_{10} values decreased with increasing moisture content when soil retained water at contents higher than the optimum water content (Wen et al., 2006). Q_{10} was the highest at the medium SWC, while Q_{10} at low SWC was similar to that at high SWC (Zhang et al., 2007; Yang et al., 2011). Alberto et al. (2009) also observed that soil moisture had an important influence on R_{eco} in an aerobic rice field, and that the relationship could be described by a quadratic equation. In very wet soil, limitation of soil–atmosphere gas exchange can lead to low concentrations of oxygen in the soil, which results in constraints on aerobic respiration of the soil bio-community (Wen

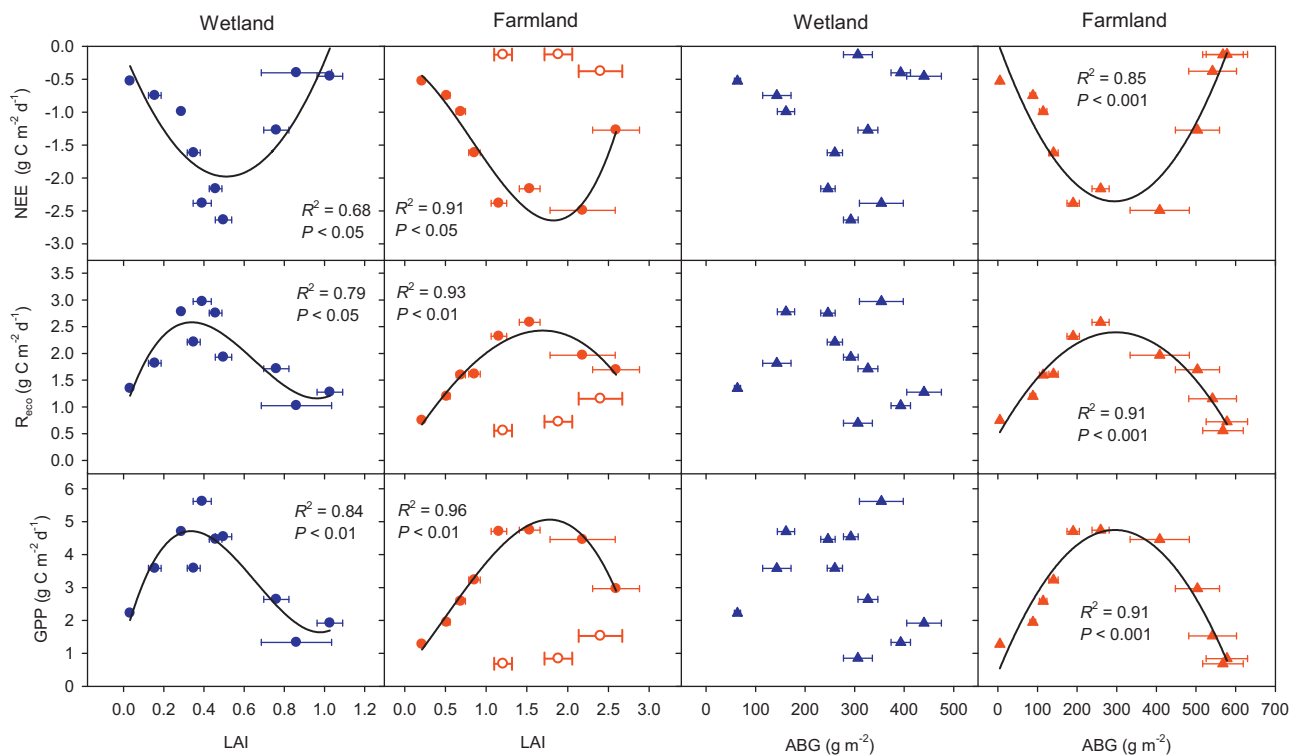


Fig. 6. Relationships between the 7-day average NEE, R_{eco} and GPP in response to LAI and ABG in the wetland and farmland. Data were obtained from the growing season in 2011. ● during the early and peak growing season in the farmland; ○ at the end of growing season in the farmland.

et al., 2006), thereby affecting the Q_{10} value of ecosystem respiration.

4.3. Ecosystem CO_2 exchange in response to LAI and AGB

In order to investigate the relationship between daily CO_2 flux and LAI, 7 days of CO_2 flux around each LAI measurement were averaged. The results show that NEE, R_{eco} and GPP were significantly related to LAI, and their peak values corresponded approximately to a relatively higher LAI at both sites (Fig. 6). At the wetland site, it showed a quadratic trend ($R^2=0.68$) between NEE and LAI, and there were strong polynomial relationships between R_{eco} ($R^2=0.79$), GPP ($R^2=0.84$) and LAI, respectively. In the farmland ecosystem, there were no significant relationships between NEE, R_{eco} , GPP and LAI for the complete length of the growing season. However, during the early and peak growing season, the temporal variations of GPP, R_{eco} and NEE in the farmland strongly depended on LAI and AGB (Fig. 6). At the wetland site, there were no significant correlations between ecosystem CO_2 fluxes and AGB.

The strong influence of vegetation properties (e.g. LAI, AGB) on CO_2 fluxes (NEE, R_{eco} , GPP) were also documented for other wetlands (Bonneville et al., 2008; Lund et al., 2010) and farmlands (Suyker et al., 2004). Unlike a forest ecosystem, the plant biomass and LAI in a wetland or farmland ecosystem exhibit large seasonal changes, and this might affect ecosystem CO_2 exchange. For example, based on synthesizing data on CO_2 exchange obtained from 12 wetland sites, Lund et al. (2010) found that annual GPP and NEE correlated significantly with LAI in those wetland sites. In a temperate cattail marsh, aboveground live biomass showed a strong correlation with 24-h average NEE and with daytime GPP (Bonneville et al., 2008). In an irrigated and rainfed maize field, daytime NEE was closely linked to LAI regardless of the growth stage (Suyker et al., 2004). In mountain grasslands, the seasonal and interannual variation of NEE, GPP and R_{eco} were also affected by AGB and LAI (Schmitt et al., 2010).

The significant correlations between LAI, AGB and ecosystem CO_2 flux illustrate that vegetation structural characteristics are important for the seasonal ecosystem CO_2 exchange, and the canopy development is an important biological process regulating CO_2 flux (Lund et al., 2010; Polley et al., 2010). To begin with, leaf area determines the amount of available photosynthetic material and the amount of light intercepted by the vegetation (Goldstein et al., 2000), therefore a high leaf area allows for high light absorption capacity on an ecosystem scale and strong photosynthetic CO_2 uptake (Lund et al., 2010). Secondly, through regulating substrate availability and fresh litter input, plant AGB is good proxy for accounting for variations in both autotrophic and heterotrophic capacity for respiration (Flanagan and Johnson, 2005), and therefore the variation in the amount of AGB modulates the variability in R_{eco} (Wohlfahrt et al., 2008). Furthermore, canopy photosynthesis is the best indicator of R_{eco} by controlling the assimilate supply for the autotrophic respiration and heterotrophic respiration through roots exudates (Davidson et al., 2006). Thus, LAI may strongly regulate the variations in ecosystem respiration (Aires et al., 2008). Finally, canopy biomass and structure allocation may affect NEE by changing LAI (Cheng et al., 2009). Overall, the seasonal variations of vegetation characteristics would result in significant changes in NEE by altering both GPP and R_{eco} . Thus, when predicting wetland CO_2 exchange, vegetation development such as leaf area and AGB should be taken into account.

4.4. Effect of wetland reclamation on carbon sequestration

The land use conversion of these wetlands for agricultural production will affect wetland vegetation, soil properties, and the process of biometabolism and material exchange (Huang et al., 2012), and subsequently lead to the change in net C stores (Jauhainen et al., 2008). In the Yellow River Delta, plowing is a traditional practice before planting cotton during spring, which can clear wetland vegetation and disturb initial soil properties,

Table 4
Effects of land conversion on net ecosystem CO₂ exchange (NEE) in different wetland and grassland ecosystems.

Site	Location	Dominant vegetation		Years of conversion	Period of observation	NEE (g CO ₂ m ⁻² d ⁻¹) ^a		Measurement methods	Citation
		Before conversion	After conversion			Before conversion	After conversion		
Tropical wetlands	Shore of Lake Victoria, Uganda	<i>Cyperus papyrus</i>	<i>Colocasia esculenta</i>	Since 1950's	2003–2004	–106.4 to –133.1 ^b	–57.0 to –76.0 ^b	Eddy covariance techniques	Saunders et al. (2012)
Boreal peatland	Northwestern Ontario, Canada	Mosses, shrubs and sedges	Standing water, bare peat surface, and dead vegetation	April 2008	May–August in 2008 and 2009	2008: –2.4 2009: –1.6	2008: 7.5	Chamber method	Wilhelm (2010)
Semiarid grassland	Inner Mongolia of Northern China	<i>Stipa kryroii</i> , <i>Agropyron cristatum</i> , <i>Artemisia frigida</i>	<i>Triticum aestivum</i> , <i>Avena nuda</i> , <i>Fagopyrum esculentum</i>	For decades	May–September 2005	–0.84	–0.63	Eddy covariance techniques	Zhang et al. (2007)
Grassland steppe	Southeastern Inner Mongolia, China	<i>S. kryroii</i> , <i>A. frigid</i> , <i>Leymus Chinensis</i> , <i>A. cristatum</i>	<i>T. aestivum</i>	1960's or 1970's	May–September 2006	–20.3	–29.1	Chamber method	Zhang et al. (2008)
Temperate grasslands	Co. Cork southern Ireland	<i>Lolium perenne</i> , <i>Alopecurus pratensis</i> , <i>Holcus lanatus</i>	<i>L. perenne</i> , <i>A. pratensis</i> , <i>H. lanatus</i>	Autumn 2003	March 2004–March 2005	–0.42	–1.04	Chamber method	Byrne et al. (2005)
Asmesophilic grasslands	Near the village of Mehrstedt in Thuringia, Germany	<i>Trisetum flavescens</i>	Broadleaf and coniferous tree species	2003	2004–2006	2004: –1.51 2005: 0.34 2006: –0.82	2004: –0.34 2005: 0.19 2006: –0.62	Eddy covariance techniques	Don et al. (2009)
Amazonian pasture	Santarém region, Brazil	<i>Brachiara brizantha</i>	<i>Oryza sativa</i>	November 2001	2001–2002	–5.18	–14.69	Eddy covariance techniques	Sakai et al. (2004)
Coastal wetlands	The Yellow River Delta, China	<i>P. communis</i> , <i>S. heteroptera</i> , <i>A. venetum</i>	<i>Gossypium hirsutum</i>	April 2008	May–October 2011	–4.73	–4.03	Eddy covariance techniques	This study

^a Positive values indicate CO₂ uptake and negative values indicate CO₂ release.

^b Peak rates of NEE during the growing season.

therefore changing the C cycles in the wetland ecosystems. Our results showed that agricultural reclamation changed the ecosystem CO₂ exchange of the coastal wetland at three distinct levels. Initially, the conversion from the wetland to farmland changed the NEE light response parameters (α , A_{\max} , and $R_{\text{eco, day}}$) and the Q_{10} of R_{eco} mainly by changing the vegetation types. Contributing to the temporal variations of NEE, R_{eco} and GPP were significantly correlated with LAI at both sites and aboveground biomass at the farmland site (Fig. 6). The NEE light response parameters showed marked differences between the two vegetation types, as well as among each month (Table 3). For the entire growing season, the Q_{10} of R_{eco} of the farmland (3.2) was higher than that of the wetland (2.3), suggesting the farmland was more sensitive to the change of temperature than the wetland. In addition, the reclamation of wetlands modified the diurnal and seasonal dynamics of ecosystem CO₂ exchange. At the diurnal scale, significant differences in variations of NEE between the wetland and farmland sites were found during the growing season (except during June and July) (Fig. 2); at seasonal scales, the mean daily GPP and R_{eco} values at the wetland site were higher than those at the farmland site (Table 2). Lastly, the agricultural reclamation changed the CO₂ sequestration capacity of the coastal wetland. The cumulative NEE in the wetland ($-237.4 \text{ g C m}^{-2}$) was higher than that in the farmland ($-202.0 \text{ g C m}^{-2}$) through the entire growing season, suggesting the conversion from wetland to farmland caused a decrease in CO₂ sequestration.

However, in this region, it is common practice to harvest cotton plants after cotton is picked. In order to assess the C removal through cotton biomass harvesting, a conversion factor of 0.45 was used to convert biomass to C content for cotton (Fang et al., 2007). Assuming that the ratio of root to canopy of cotton is 0.2, about 681.5 g m^{-2} cotton biomass was removed from the farmland. Thus, C export through biomass removal was approximately 333.9 g C m^{-2} , leading to a net loss of C from the farmland of 131.9 g C m^{-2} during the growing season.

Previous studies have also shown that reclamation of wetlands for agricultural uses can change the C assimilation and sequestration potential of wetlands through changing the plant community composition and productivity, or through soil compaction and mineralization in wetlands (Nieveen et al., 2005; Jauhainen et al., 2008; Saunders et al., 2012). For example, the tropical *papyrus* wetlands have the potential to act as a sink for significant amounts of carbon of $10 \text{ t C ha}^{-1} \text{ yr}^{-1}$, but the land use conversion of these wetlands for agricultural production led to a reduction in the carbon assimilation (Saunders et al., 2012). Furthermore, when carbon exports from crop biomass removal were accounted for these wetlands represent a significant net loss of carbon ($10.2 \text{ t C ha}^{-1} \text{ yr}^{-1}$). Similarly, many studies have found that grassland conversion also changed ecosystem C fluxes (Table 4), reducing (e.g. Zhang et al., 2007; Schmitt et al., 2010) or improving (e.g. Zhang et al., 2008; Sakai et al., 2004; Byrne et al., 2005) the carbon sequestration of grasslands. Additionally, the years since the conversion might have significant effects on ecosystem C cycles (Don et al., 2009; Wilhelm, 2010), therefore, it is essential to have a more long-term, continuous measurement of ecosystem CO₂ exchange and other environmental factors in this region.

In addition, the wetlands are also natural sources of greenhouse gases such as CH₄ and N₂O, and the wetland reclamation can dramatically alter the exchange of greenhouse gases between wetlands and the atmosphere. On one hand, conversion of wetlands to drylands can result in a shift from a strong CH₄ source to a weak sink (Jiang et al., 2009); on the other hand, conversion of wetlands to agricultural lands with cropping activities can potentially lead to increases in N₂O emission (Roulet, 2000). Therefore, future research requires the establishment of a full wetland ecosystem or land cover greenhouse gas measurement program (Chasmer et al.,

2012), including CO₂ fluxes as well as dissolved organic C, and soil CH₄ and N₂O emission, in order to clarify the responses of wetland C exchange caused by land conversion in the long-term.

5. Conclusions

Our study demonstrates that agricultural reclamation changed the light response of NEE and the temperature response of R_{eco} of the coastal wetland mainly by changing the LAI and AGB of vegetations. As a consequence, the reclamation modified the diurnal and seasonal dynamics of ecosystem CO₂ exchange. Ultimately, the agricultural reclamation reduced the CO₂ sequestration capacity of the coastal wetland. Moreover, when biomass removal was taken into account, the farmland was a strong source for CO₂ during the growing season. Although the present study has made clear how agricultural reclamation affect the ecosystem CO₂ exchanges at a coastal wetland, it is not enough for us to evaluate the actual effects of wetland conversion on C sinks and sources based on the data only from one growing season. Therefore, it is necessary to carry out further measurements to assess the long-term effect of agricultural reclamation on the ecosystem CO₂ exchanges in this region. In addition, agricultural reclamation may play a key role in changing the other processes involving CH₄, N₂O and dissolved organic C in wetlands. Hence, a comprehensive study is required to improve our knowledge on the potential impacts of land conversion on C sequestration in coastal wetlands.

Acknowledgments

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