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Carbon budgets of wetland ecosystems in China

Derong Xiao[1] | Lei Deng[2,3] | Dong-Gill Kim[4] | Chunbo Huang[5] | Kun Tian[1]

[1]National Plateau Wetlands Research Center, Southwest Forestry University, Kunming, Yunnan, China
[2]State Key Laboratory of Soil Erosion and Dryland Farming on the Loess Plateau, Northwest A&F University, Yangling, Shaanxi, China
[3]Institute of Soil and Water Conservation, Chinese Academy of Sciences and Ministry of Water Resources, Yangling, Shaanxi, China
[4]Wondo Genet College of Forestry and Natural Resources, Hawassa University, Shashemene, Ethiopia
[5]College of Horticulture and Forestry Sciences, Huazhong Agricultural University, Wuhan, Hubei, China

Correspondence
Lei Deng, State Key Laboratory of Soil Erosion and Dryland Farming on the Loess Plateau, Northwest A&F University, Yangling, Shaanxi, China.
Email: leideng@ms.iswc.ac.cn

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Abstract

Wetlands contain a large proportion of carbon (C) in the biosphere and partly affect climate by regulating C cycles of terrestrial ecosystems. China contains Asia's largest wetlands, accounting for about 10% of the global wetland area. Although previous studies attempted to estimate C budget in China's wetlands, uncertainties remain. We conducted a synthesis to estimate C uptake and emission of wetland ecosystems in China using a dataset compiled from published literature. The dataset comprised 193 studies, including 370 sites representing coastal, river, lake and marsh wetlands across China. In addition, C stocks of different wetlands in China were estimated using unbiased data from the China Second Wetlands Survey. The results showed that China's wetlands sequestered 16.87 Pg C (315.76 Mg C/ha), accounting for about 3.8% of C stocks in global wetlands. Net ecosystem productivity, jointly determined by gross primary productivity and ecosystem respiration, exhibited annual C sequestration of 120.23 Tg C. China's wetlands had a total gaseous C loss of 173.20 Tg C per year from soils, including 154.26 Tg CO_2-C and 18.94 Tg CH_4-C emissions. Moreover, C stocks, uptakes and gaseous losses varied with wetland types, and were affected by geographic location and climatic factors (precipitation and temperature). Our results provide better estimation of the C budget in China's wetlands and improve understanding of their contribution to the global C cycle in the context of global climate change.

KEYWORDS
carbon emission, carbon sequestration, carbon stock, carbon uptake, climate, wetland

1 | INTRODUCTION

Wetlands cover only 5%–8% of the global land surface (Mitsch et al., 2012) but are considered important sinks of carbon (C) in the biosphere (Nahlik & Fennessy, 2016), because wetland soil C accounts for 20%–30% of the estimated 2,500 Pg C of global soil C stocks (Lal, 2008). While wetlands act as large C sinks, they are also important sources of greenhouse gases (GHGs) like carbon dioxide (CO_2) and methane (CH_4) (Bloom et al., 2017; Bousquet et al., 2011; Ma, Zhang, Tang, & Liu, 2016). As a result, wetlands play an important role in regulating the C cycle at a global scale and quantifying the C budgets of wetlands including C stores, uptake and emission, has become a major topic in global change research (Lu et al., 2017).

Estimating C stocks in wetland ecosystems and understanding the controlling factors have increasingly attracted worldwide attention (Lu et al., 2017; Nahlik & Fennessy, 2016; Turetsky et al., 2014). Generally, temperate and tropical wetlands sequester 3–4 times more C than boreal wetlands (Mitsch et al., 2012); freshwater inland wetlands hold greater soil C than tidal saltwater sites (Nahlik & Fennessy, 2016); and coastal wetlands act as large C sinks, but inland wetlands are small C sinks or nearly C neutral (Lu et al., 2017). The C stocks in wetlands can be decreased by anthropogenic disturbance such as hydrologic modification and agricultural use (Nahlik & Fennessy, 2016). In addition, C uptake/emission also differs remarkably among wetlands. Coastal wetlands can take up great amounts of CO_2 (Chmura, 2013) but have negligible CH_4 emissions (Chmura, Anisfeld, Cahoon, & Lynch, 2003), and some inland wetlands are thought to have a small atmospheric C uptake or be “neutral pipes” merely conveying terrestrial C to oceans (Chu et al., 2015). Tropical wetlands are the world's largest natural source of CH_4 (Bousquet et al., 2011), accounting for 52%–58% of global wetland CH_4 emissions (Bloom, Palmer, Fraser, Reay, & Frankenberg, 2010). The CO_2 and CH_4 emission and uptake are generally dependent on C availability and decomposition rate (Miyajima, Wada, Hanba, & Vijarnsorn, 1997), presence of macrophytes (Laanbroek, 2010), inundation extent and temperature (Elberling et al., 2011; Yvon-Durocher et al., 2014), as well as soil pH (Singh, Kulshreshtha, & Agnihotri, 2000). Therefore, wetlands are sometimes sink of atmospheric C, but can function as C sources due to change in environmental conditions (Mitsch et al., 2012; Pester, Knorr, & Friedrich, 2012). Moreover, gross primary production (GPP), ecosystem respiration (RE) and net ecosystem production (NEP) differ with latitude, largely due to complex distributions of vegetation and the controlling factors (e.g., precipitation, temperature, topography and soil properties) (Lu et al., 2017) that affect the uptake/emission of carbon in wetland ecosystems. Because of significant differences in wetland type, land cover and environmental conditions, quantifying the C budget in different wetland ecosystems at the national scale can help in understanding their contribution to mitigating climate change, and adopting corresponding strategies to protect wetlands.

China contains Asia's largest wetlands (Wang, Wu, Madden, & Mao, 2012), accounting for about 10% of the global wetland area (Niu et al., 2012). In the past 20 years, many studies have been carried out on C budgets in China's wetland ecosystems to determine whether they are a net C sink or source, and their contribution to mitigating GHG emissions (Chen et al., 2013a; Jin, Zhuang, He, Zhu, & Song, 2015; Wang et al., 2014; Xu et al., 2018; Zheng, Niu, Gong, Dai, & Shangguan, 2013). Wetlands function as important C stocks in China, but estimations differ among studies due to different measurement methods and the estimated wetland areas (Xu et al., 2018; Zheng et al., 2013). For example, Zheng et al. (2013) estimated that total C stocks ranged within 5.39–7.25 Pg C using inventory approach, but Xu et al. (2018) obtained 3.62 ± 0.80 Pg C from a synthesis. Moreover, China's wetlands play a large role in C intake and emission, which not only take up CO_2 from the atmosphere but also release CH_4 into the atmosphere, especially in the Qinghai-Tibetan Plateau and Northeast China (Chen et al., 2013a; Wang et al., 2014). Jin et al. (2015) predicted that China's wetlands will change from C sources to sinks after 2030. In fact, C budgets in China's wetlands are associated with wetland type (Li et al., 2017), growing period from May to September (Zhang et al., 2008) and standing-water depth (Chen et al., 2013a), leading to high uncertainty in estimating ecosystem C budgets. Exploring the general patterns and C budgets in China's wetland ecosystems is necessary to quantify their contributions to C budgets of global wetlands.

Although studies have attempted to accurately estimate C budgets in China's wetland ecosystems, many uncertainties remain. First, China's wetland area data used to estimate C budgets came from multisources, and there is a lack of authoritative data from national departments. Current data sources include national soil census data (Zheng et al., 2013), remote sensing interpretation data (Ma et al., 2015) and local investigation reports (Ding, Cai, & Wang, 2004). Thus, areas used for calculation of C budgets in wetlands are inconsistent, resulting in large differences in estimates. Second, China's wetland types are diverse, but estimation of C budgets have been mainly on a single type, for example, marsh (Chen et al., 2013b; Ma et al., 2015), leading to differences in results. Consequently, current estimations are unlikely to reflect the C budgets of all of China's wetlands. Third, there are insufficient field measurements of C budgets in wetlands across all of China. Previous studies mainly measured local data on C budgets focused on different wetland types (Jin et al., 2015; Liu et al., 2014; Ma, Liu, et al., 2016; Ma, Zhang, et al., 2016) and there is a lack of integrated studies in different regions of China, resulting in inaccurate estimates. Fourth, most previous wetland C budgets studies were not comprehensive and lacked integrated analysis on C cycles of wetland ecosystems, especially at the national scale. Previous studies focused on estimation of either soil C stocks (e.g., Ma et al., 2015) or CO_2 and CH_4 emissions (e.g., Chen et al., 2013a) in wetlands. These results did not accurately reflect the overall situation of C budgets in China's wetland ecosystems. To update C budgets of wetlands at the national scale, and better understand their contribution to C cycle for mitigating global climate change, will require applying a more advanced method to a large amount of authoritative cross-site field data (Nahlik & Fennessy, 2016).

In order to strengthen protection of wetlands, the China State Forestry Administration carried out the Second National Wetlands Survey during 2009–2013, which showed that China's wetlands covered 53.42×10^6 ha and consisted of four major natural types: coastal, river, lake and marsh wetlands (State Forestry Administration, 2015). This provided one unbiased dataset of the distribution area of China's wetlands. In recent years, field-monitored data of C stocks, uptakes and emissions of different wetland types and regions greatly increased in China (Li et al., 2017; Ma, Liu, et al., 2016), and the results provide a large detailed dataset for comprehensive research on the C budget of wetlands. Therefore, a synthesis study to estimate C budgets is now feasible and timely. In this study, we used the wetland datasets of distribution area and type issued by China State Forestry Administration, combined with a national database compiled from literature across China.

The specific objectives of the study were to (a) estimate C stocks (soil and plant biomass C), C uptake/emission (GPP, NEP, RE and soil CO_2 and CH_4 emission) in different wetlands; (b) explore the relationships among C budgets and the factors regulating them; and (c) determine net C budgets in China. This study will provide essential information to strengthen understanding of the contribution of wetlands to the C cycle, and improve wetland management strategies and climate change policy related to wetlands at the national scale.

2 | MATERIALS AND METHODS

2.1 | Data preparation

All available peer-reviewed publications concerning China's wetlands were used in our synthesis. Publications were searched through the Web of Science, Google Scholar and China National Knowledge Infrastructure for studies published between December 1999 and May 2018. Key words used to search the studies were soil C, C flux/eddy, CH_4 emission, biomass, GPP, NEP, RE, coastal, river, lake, marsh, wetland and China. The following criteria were used to select publications for analysis: (a) sites having at least 1 year of continuous flux measurements via open-path or closed-path eddy covariance (EC) methods that could be used for calculating annual CO_2 fluxes (GPP, RE and NEP) when the data were available; (b) soil C stocks were provided or were calculated based on soil organic C (SOC) concentration, bulk density (BD) and soil depth; (c) location, mean annual temperature (MAT, °C) and precipitation (MAP, mm), and wetland types were clearly given.

Raw data were obtained from tables or, when not available in tables, extracted by digitizing graphs using GetData Graph Digitizer (version 2.24, Russian Federation), which provided high accuracy of data extraction (>99%) for various meta-analyses or synthesis (e.g., Yang, Luo, & Finzi, 2011, Deng, Liu, & Shangguan, 2014). For each paper, the following information was compiled: sources, location (longitude, latitude and altitude), climatic data (MAP and MAT), above- and below-ground biomass, soil depth from soil surface, BD and amount of SOC in each soil depth (Appendix Dataset S1). Additionally, many studies of wetlands worldwide have taken 100 cm as a suitable soil depth to estimate the soil C stocks (Xu et al., 2018; Zheng et al., 2013), which includes most soil organic material (Carnell et al., 2018). So, for comparison with previous studies, we only chose data for 0–100 cm soil depth to estimate soil C stocks from the collected papers.

In total, the final dataset comprised 193 studies mostly published during 2005–2018 (Appendix Word S1). This included 370 sites with 600 observations across China, which covered most of the area of China's wetlands (Figure 1). All wetlands were broadly classified into coastal, river, lake and marsh based on China's Wetland Classification adopted from the Ramsar Convention (State Forestry Administration, 2015). The compiled dataset consisted of 171, 72, 154 and 203 observations in coastal, river, lake and marsh wetlands respectively (Figure 1; Appendix Dataset S1).

2.2 | Data calculation

Vegetation C stock (C_B)

$$C_B = B \times C_f \tag{1}$$

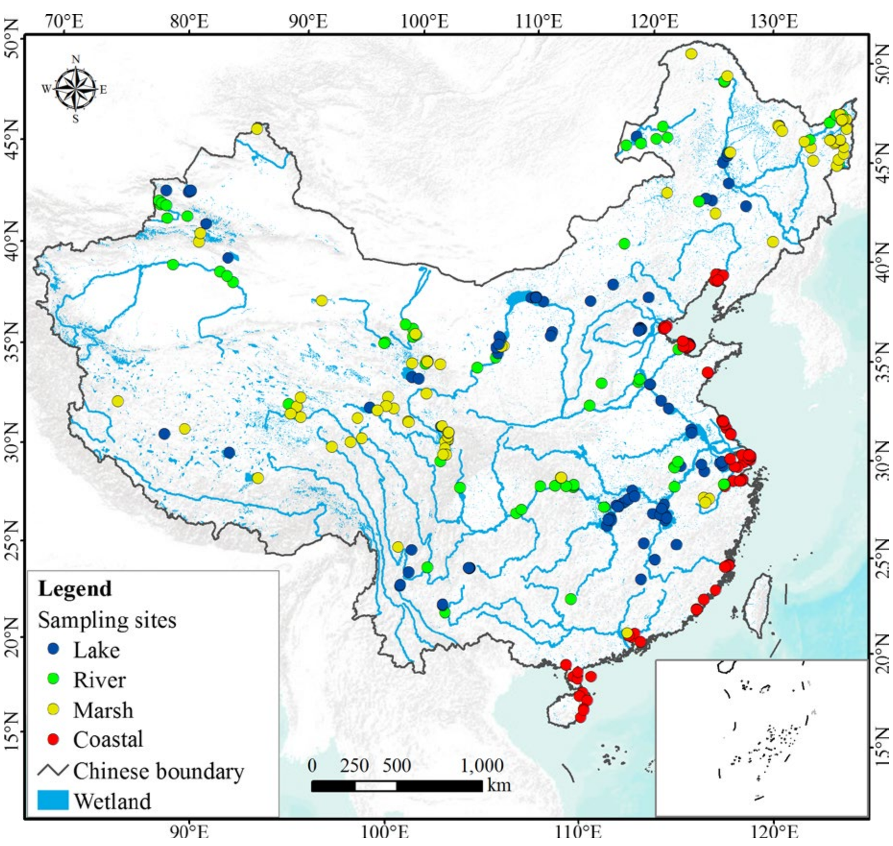

FIGURE 1 Distributions of sampling sites of China's wetlands in the dataset [Colour figure can be viewed at wileyonlinelibrary.com]

where, C_B is vegetation C stock, B is vegetation biomass and C_f is plant biomass C coefficient. It is worth noting that in China, marsh, meadow and river wetlands are dominated by herbaceous plants such as submerged, floating-leaf and emergent vegetation, whereas coastal wetlands are dominated by mangroves. Previous studies showed that the C_f of Chinese herbaceous plants is approximately 0.45 (Fang, Guo, Piao, & Chen, 2007; Fang, Yang, Ma, Anwar, & Shen, 2010) and of trees is about 0.50 (Fang, Chen, Peng, Zhao, & Ci, 2001). The C_f of different dominant species did not significantly differ according to the State Forestry Administration (Li & Lei, 2010). Therefore, based on previous studies, we adopted the C_f value of 0.45 in wetlands with herbaceous plants as dominant species, and of 0.50 in wetlands with tree-dominant species, for estimating respective C stocks.

Soil C stocks (C_s)

$$C_s = \frac{SOC \times BD \times D}{10} \tag{2}$$

where, C_s is SOC stock (Mg/ha), with SOC in g/kg; BD is in g/cm^3, and D is soil thickness (cm).

Soil BD values are critical for calculations of C_s, but many studies did not measure this. Alternatively, we established an empirical relationship between SOC concentration and soil BD using the collected values from wetlands (Appendix Dataset S1). The formula for the calculation follows:

$$BD = 0.8572e^{-0.0331SOC} + 0.7446e^{-0.0027SOC}$$

$$R^2 = 0.72, p < 0.0001, n = 105 \tag{3}$$

The missing values of soil BD were interpolated using predicted values from the empirical functions (*4-paramter double exponential decay model*) in Figure 2. To increase comparability of data derived from different studies, original soil C data were converted to soil C stocks in the top 100 cm using the depth functions of Jobbágy and Jackson (2000) according to the following equations:

$$Y = 1 - \beta^d \tag{4}$$

$$X_{100} = \frac{1-\beta^{100}}{1-\beta^{d0}} \times X_{d0} \tag{5}$$

For observations that only had 0–100 cm soil C stocks, we derived (Deng et al., 2014; Deng, Shangguan, Wu, & Chang, 2017):

$$X_{d0} = \frac{1-\beta^{d0}}{1-\beta^{100}} \times X_{100} \tag{6}$$

where Y represents the cumulative proportion of the soil C stock from the soil surface to depth d (cm), β is the relative rate of decrease in the soil C stock with soil depth, X_{100} denotes soil C stock in the upper 100 cm, d_0 denotes the original soil depth available in individual studies (cm) and X_{d0} is original soil C stock.

Although Jobbágy and Jackson (2000) provided the depth distribution of soil C for 11 biome types globally, there was no significant difference in the depth distribution among biome types or between individual biomes and the global average. Therefore, in the present study, the global average depth distributions for C were adopted to calculate β (i.e., 0.9786) in equations.

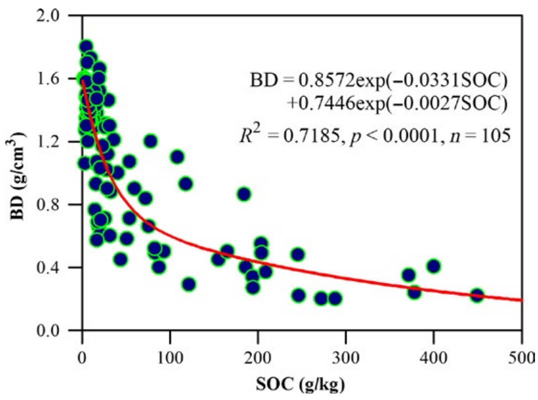

FIGURE 2 Relationship between bulk density (BD) and soil organic carbon (SOC) in wetlands of China [Colour figure can be viewed at wileyonlinelibrary.com]

2.3 | GPP, NEP and RE

In our dataset, not all sites reported the GPP, NEP and RE. For many studies that did not measure GPP and NEP, we adopted the methods of Lu et al. (2017). Temperature and precipitation are both important controlling factors of GPP, NEP and RE (Lu et al., 2017). Because MAT and MAP together explained higher variances in GPP, NEP and RE than MAT or MAP alone, Lu et al. (2017) developed the following two regression equations (Equations 7 and 8), which can best describe the spatial patterns of GPP and NEP (both in g C m^{-2} year^{-1}) in global inland and coastal wetland ecosystems:

$$GPP = 416.61 + (41.91 \times MAT) + (0.21 \times MAP) + (0.01 \times MAT \times MAP)$$

$$R^2 = 0.71, p < 0.001 \tag{7}$$

$$NEP = 159.29 + (7.75 \times MAT) - (0.24 \times MAP) + (0.02 \times MAT \times MAP)$$

$$R^2 = 0.57, p < 0.001 \tag{8}$$

For each site, if RE was not reported, it was calculated as the difference of GPP and NEP:

$$RE = GPP - NEP \tag{9}$$

2.4 | Data analysis

One-way analysis of variance (ANOVA) was used to test the significance of differences in plant and soil C stocks, GPP, NEP, RE and soil CO_2 and CH_4 emissions among the four wetlands. The significance was at p < 0.05. Given the large differences in sample size among different wetland types, the assumptions of the one-way ANOVA, including normal distribution and homogenous variance of the data, were first tested. When a significance difference among wetland types was detected, the least significant difference (LSD) post-hoc test was used for multiple comparisons. The generalized linear model (GLM) was used to conduct multiple linear regression analysis. Pearson correlations were calculated between location, climate and plant and soil C stocks, soil CO_2 and CH_4 emissions, GPP, RE and NEP. To explore the effects of location, climate, plant and soil on soil C stocks, soil CO_2 and CH_4 emissions, GPP, RE and NEP, multiple regression using the "Enter" method was used to analyse the multiple relationships between location, climate, SOC (0–20 cm), plant biomass and soil C stocks, soil CO_2 and CH_4 emissions, GPP, RE and NEP. Due to that GPP and NEP at some sites were estimated from MAP and MAT, we did not include MAP and MAT in the multiple regression analysis, although they definitely had good relationships (Appendix Table S1). All statistical analyses were performed with SPSS version 18.0 (SPSS Inc., Chicago, IL).

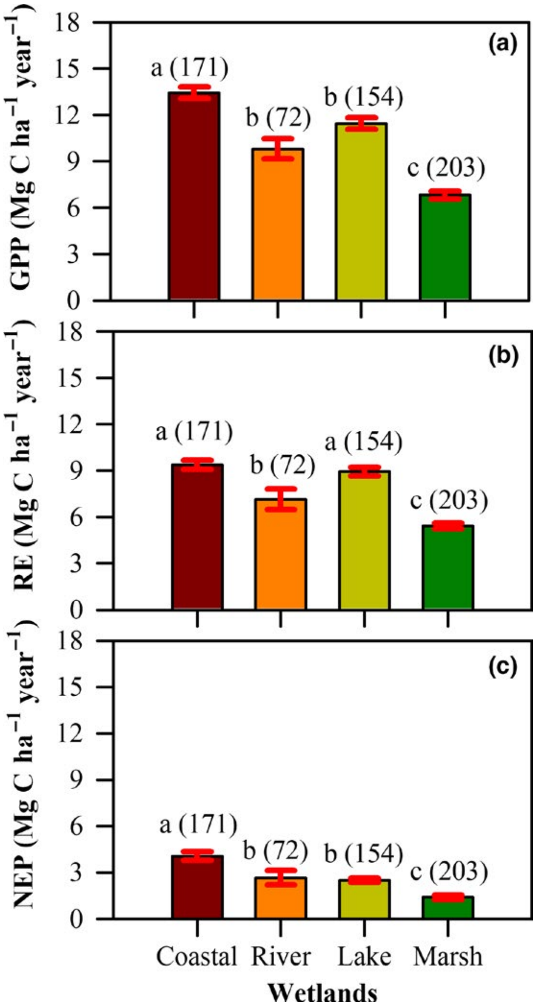

FIGURE 3 Gross primary production (GPP) (a), ecosystem respiration (RE) (b) and net ecosystem production (NEP) (c) in the four typical wetlands of China. Note: Plant biomass includes shoot and root biomasses. Data are means ± *SE*. Different letters above error bars indicate significant difference among different wetlands at $p < 0.05$. Values in parentheses are the number of observations [Colour figure can be viewed at wileyonlinelibrary.com]

3 | RESULTS

3.1 | C uptake and gaseous C losses

We compared GPP, RE and NEP for coastal, river, lake and marsh wetlands (Figure 3). The GPP in coastal wetland (13.44 ± 0.36 Mg ha^{-1} $year^{-1}$; mean ± standard error is used here and onward) was significantly higher than in river, lake and marsh (9.81 ± 0.65, 11.44 ± 0.36 and 6.83 ± 0.25 Mg C ha^{-1} $year^{-1}$ respectively). There was no significant difference in GPP between river and lake; however, GPP in both river and lake were significantly higher than that in marsh (Figure 3a). The RE was 9.37 ± 0.29, 7.15 ± 0.66, 8.94 ± 0.27 and 5.43 ± 0.19 Mg C ha^{-1} $year^{-1}$ in coastal, river, lake and marsh wetlands, respectively (Figure 3b). There was no significant difference in RE between coastal wetland and lake ($p > 0.05$); however, both of these were significantly higher than for river and marsh ($p < 0.05$) and the RE in the river was significantly higher than for marsh ($p < 0.05$) (Figure 3b). Coastal wetland had the highest NEP (4.06 ± 0.27 Mg C ha^{-1} $year^{-1}$), followed by river and lake (2.66 ± 0.47 2.50 ± 0.13 Mg C ha^{-1} $year^{-1}$, respectively), which did not significantly differ from each other and marsh had the smallest RE of 1.40 ± 0.15 Mg C ha^{-1} $year^{-1}$ (Figure 3c).

Overall, soil CO_2 and CH_4 emissions were related to wetland types, reflecting a high degree of diversity in wetlands in China (Figure 4). Soil CO_2 emission in coastal wetland was highest (5.98 ± 0.60 Mg CO_2-C ha^{-1} $year^{-1}$) and was significantly higher than for river, lake and marsh (1.76 ± 0.19, 1.62 ± 0.15 and 3.49 ± 0.51 Mg CO_2-C ha^{-1} $year^{-1}$ respectively). There was no significant difference in emissions between river and lake; however, both were significantly lower than for marsh (Figure 4a). Similarly, soil CH_4 emission in coastal wetland was highest but not significantly different to marsh (0.60 ± 0.26 and 0.19 ± 0.08 Mg CH_4-C ha^{-1} $year^{-1}$ respectively). Soil CH_4 emissions in coastal wetland and marsh were significantly higher than for river and lake (0.15 ± 0.04 and 0.51 ± 0.13 Mg CH_4-C ha^{-1} $year^{-1}$, respectively) but the latter two did not significantly differ from each other (Figure 4b).

3.2 | C stocks

The C stocks of ecosystem, plant biomass and soil varied as a function of wetland type and soil depth (Figure 5). When grouped by type, coastal wetland stored the highest plant biomass C (9.60 ± 1.84 Mg/ha) and was significantly higher than for river, lake and marsh (3.70 ± 1.45, 4.49 ± 1.06 and 2.31 ± 0.66 Mg/ha, respectively); however, the latter three did not significantly differ (Figure 5a). Marsh stored the highest soil C, with 467.05 ± 43.18 Mg/ha, followed by lake, river and coastal wetland (269.79 ± 38.95, 177.96 ± 43.98 and 84.08 ± 8.08 Mg/ha, respectively; Figure 5b). Overall, total C stocks (plant biomass C and soil C) of marsh was 469.36 Mg/ha, which was significantly ($p < 0.05$) higher than for lake, river and coastal wetland (274.28, 181.66 and 93.68 Mg/ha, respectively); plant biomass C stocks only accounted for a small proportion of wetland ecosystem, with 1.4%–2.7%.

When grouped by soil depth, overall soil C stocks in all four wetlands decreased with soil depth within 0–100 cm (Figure 5b). Soil C

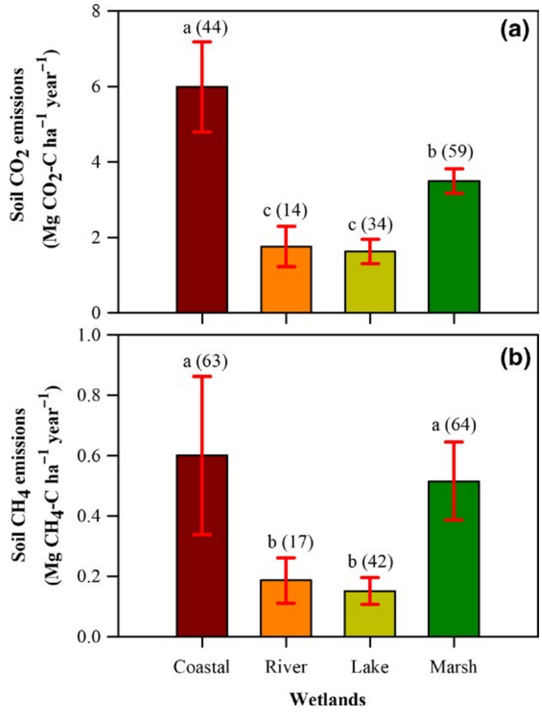

FIGURE 4 Soil carbon dioxide (CO_2) (a) and methane (CH_4) emissions (b) in the four typical wetlands of China. Note: Plant biomass include shoot and root biomasses. Data are means ± *SE*. Different letters above error bars indicate significant difference among the different wetlands at $p < 0.05$. Values in parentheses are the number of observations [Colour figure can be viewed at wileyonlinelibrary.com]

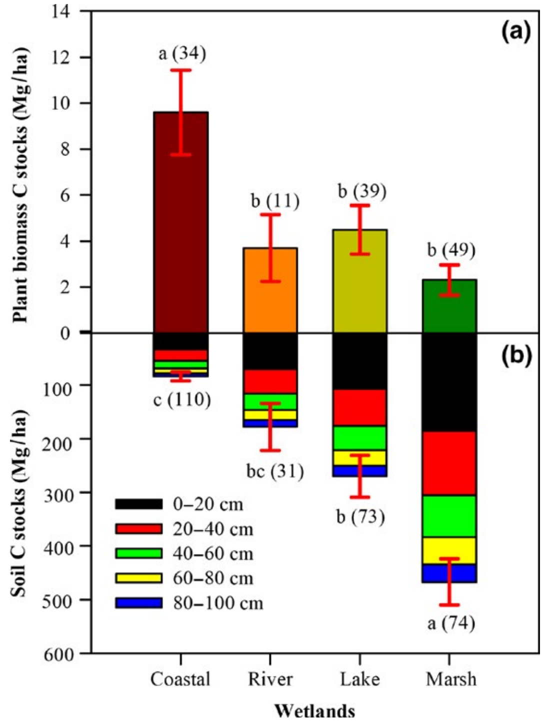

FIGURE 5 Plant biomass (a) and soil carbon (C) stocks (b) in the four types of wetlands in China. Note: Plant biomass include shoot and root biomasses. Data are means ± *SE*. Different letters above error bars indicate significant difference among the different wetlands at $p < 0.05$. Values in parentheses are the number of observations. Error bars in (b) are the mean of 0–100 cm soil C stocks [Colour figure can be viewed at wileyonlinelibrary.com]

stocks in 0–20 cm in coastal, river, lake and marsh were 33.37 ± 6.35, 70.62 ± 35.64, 107.06 ± 30.81 and 185.34 ± 34.15 Mg/ha, respectively; correspondingly accounting for 35.6, 38.9, 39.0 and 39.5% of soil C stocks in 0–100 cm. They were significantly higher than soil C stocks in 20–100 cm ($p < 0.05$).

3.3 | Net C budgets in China's wetlands

Using area data issued by the State Forestry Administration in 2013, we estimated the C budgets in China's ecosystems (Table 1). Results showed that China's wetlands totally sequestered 14.77–18.97 Pg C: 0.16–0.28 Pg C in plant biomass and 14.16–18.69 Pg C in soil (Table 1). Among wetland types, marsh had the largest C stocks, followed by lake and river, and coastal wetland had the least (Table 1).

Soil C total annual emissions of China's wetlands were 154.26 Tg CO_2-C, 18.94 Tg CH_4-C and 173.20 Tg C (Table 1). Such C emissions differed among wetlands: marsh was the largest C source and accounted for 50.4% of the total soil C emission, followed by coastal (22.0%), lake (16.0%) and river (11.9%; Table 1).

Of the four wetlands, marsh had the largest GPP and RE, followed by lake and river, and coastal wetland had the least (Table 1). In the whole of China's wetland, although 76.2% of total GPP (505.19 Tg/year) was lost through RE (384.94 Tg/year), the NEP was positive (120.23 Tg/year) indicating that they acted as a net C sink (Table 1).

3.4 | Relationships among C budgets

Annual GPP, RE and NEP exhibited significant linear correlations with SOC and plant biomass (shoot and root) respectively (Figure 6). Annual GPP, RE and NEP had exponential declining relationships with SOC (Figure 6a–c), but linear increasing relationships with plant biomass (Figure 6d–f). Moreover, the GPP, RE and NEP were significantly positively correlated among each other (Figure 7c). Except for the negative linear relationships of CH_4 emissions with GPP and NEP (Figure 8g and h), the relationships of CO_2 emission with plant biomass, SOC, GPP and NEP (Figure 8a–d), as well as CH_4 emission with plant biomass and SOC (Figure 8e and f) were all nonsignificant.

3.5 | Factors regulating C budgets

We examined effects of location (longitude, latitude and altitude), climate (MAT and MAP) on C stocks, C uptake and emission in China's wetlands (Table 2). Plant biomass C was mainly determined by MAT and MAP, and both 0–20 and 0–100 cm soil C were mainly determined by latitude and altitude (Table 2). In addition, both 0–20 and 0–100 cm soil C were negatively associated with MAP, which had a stronger effect than latitude and altitude (Table 2). Soil CO_2 emissions were significantly positively correlated with longitude and MAT (Table 2). Soil CH_4 emissions were significantly positively

TABLE 1 Estimates of net carbon (C) budgets in China's wetlands

Wetland type	Area[a] ($\times 10^6$ ha)	C stocks			Soil C emission			GPP (Tg/year)	RE (Tg/year)	NEP (Tg/year)
		Plant biomass (Pg)	Soil[b] (Pg)	Total (Pg)	CO_2-C (Tg/year)	CH_4-C (Tg/year)	Total (Tg/year)			
Coastal	5.79	0.06 ± 0.01	0.49 ± 0.05	0.54 ± 0.06	34.68 ± 6.91	3.47 ± 1.52	38.15 ± 8.43	77.82 ± 2.08	54.27 ± 1.67	23.54 ± 1.56
River	10.55	0.04 ± 0.02	1.88 ± 0.46	1.92 ± 0.48	18.60 ± 5.64	1.97 ± 0.79	20.57 ± 6.44	103.50 ± 6.86	75.47 ± 6.91	28.03 ± 4.94
Lake	15.33 (6.75)[c]	0.07 ± 0.02	4.14 ± 0.59	4.20 ± 0.61	24.97 ± 4.96	2.31 ± 0.68	27.79 ± 5.64	175.45 ± 5.52	137.15 ± 4.17	38.29 ± 2.05
Marsh	21.73	0.05 ± 0.01	10.15 ± 0.94	10.20 ± 0.95	76.01 ± 6.94	11.18 ± 2.81	87.20 ± 9.75	148.42 ± 5.43	118.05 ± 4.12	30.37 ± 3.20
Total	53.42	0.22 ± 0.06	16.65 ± 2.04	16.87 ± 2.10	154.26 ± 24.45	18.94 ± 5.80	173.20 ± 30.26	505.19 ± 19.89	384.94 ± 16.88	120.23 ± 11.75

Note. Tg = 10^{12} g, Pg = 10^{15} g.
[a]Data from State Forestry Administration (2015). [b]Soil C stocks are of 0–100 cm. [c]Indicates the area, including artificial wetlands in parentheses.

correlated with latitude, but negatively with MAP and MAT (Table 2). Moreover, longitude had a more important effect on soil CO_2 emission than MAT; and MAT and MAP had a greater effect on soil CH_4 emission than longitude (Table 2). At ecosystem scale, both location and climate factors significantly affected GPP, RE and NEP (Table 2). There were negative correlations of latitude and altitude with GPP, RE and NEP, but longitude was positively correlated with GPP, RE and NEP (Table 2). Compared with coefficients, climatic factors MAP and MAT had stronger correlations than location factors (latitude, altitude and longitude) with GPP, RE and NEP (Table 2).

We conducted linear regression analysis to assess the combined effects of location (longitude, latitude and altitude), climate (MAP and MAT), surface SOC (0–20 cm), plant biomass on 0–20 and 0–100 cm soil C stocks, soil CO_2 and CH_4 emissions, GPP, RE and NEP (Table 3). Factors of location, climate, SOC and plant biomass together explained 83.2% and 81.3% of variation for 0–20 and 0–100 cm soil C stocks, respectively (Table 3). But these factors only explained 49.3% and 38.9% of the variations in soil CO_2 and CH_4 emission ($p > 0.05$) (Table 3). In addition, location, SOC and plant biomass together explained 96.2% and 90.9% of variation for GPP and NEP, respectively, but location and plant biomass together explained 90.2% of variation in RE (Table 3). Based on these analyses, soil C stocks, GPP, NEP and RE were well described using the regression equations (Table 3).

4 | DISCUSSION

4.1 | C stocks in China's wetland ecosystems

China's wetlands stored 16.78 Pg C, 1.3% in biomass and 98.7% in soil (to depth of 100 cm), indicating that soils were the largest sink in wetland ecosystems (Table 1). Considering soil C stocks only, China's wetlands accounted for about 3.8% of the estimated 450 Pg C in global wetlands (Gorham, 1991; Warner, Clymo, & Tolonen, 1993), or close to 1.0% of the world's total soil C (Mitsch et al., 2012). Considering a price of \$4.6–23.0 per Mg C (Liu et al., 2014), C trade potentials of the C stocks in China's wetland ecosystems are substantial, in the range of \$77.2–\$385.9 billion. Our estimate of C stored in China's wetlands was much larger than previous estimates of 5.39–7.25 Pg C (Zheng et al., 2013) and 3.62 ± 0.80 Pg C (Xu et al., 2018). The difference is mainly attributed to the much larger area of wetlands (53.42 × 10^6 ha; State Forestry Administration, 2015) compared to previous studies—22.49 × 10^6 ha (Zheng et al., 2013) and 14.46 × 10^6 ha (Xu et al., 2018). An estimated 7.0 Mg C ha^{-1} is stored in biomass of the world's wetlands (Aselmann & Crutzen, 1989), which is slightly lower compared with that of China (7.12 Mg C/ha) (Table 4). The soil C pool (16.65 Pg C in 53.42 × 10^6 ha) in China's wetlands (Table 3) is larger than that of the USA with 11.52 Pg C in 38.4 × 10^6 ha (Nahlik & Fennessy, 2016). On an area basis, the soil C in China's wetlands (315.76 Mg/ha) are also a little higher than that of the USA (300.00 Mg/ha).

The C stocks had a high degree of variability among the different wetland types. In our study, marsh wetland stored the most C and accounted for 60.2% of the total, and coastal wetland stored the least C with only 3.3% of the total. Marsh wetland contained the largest

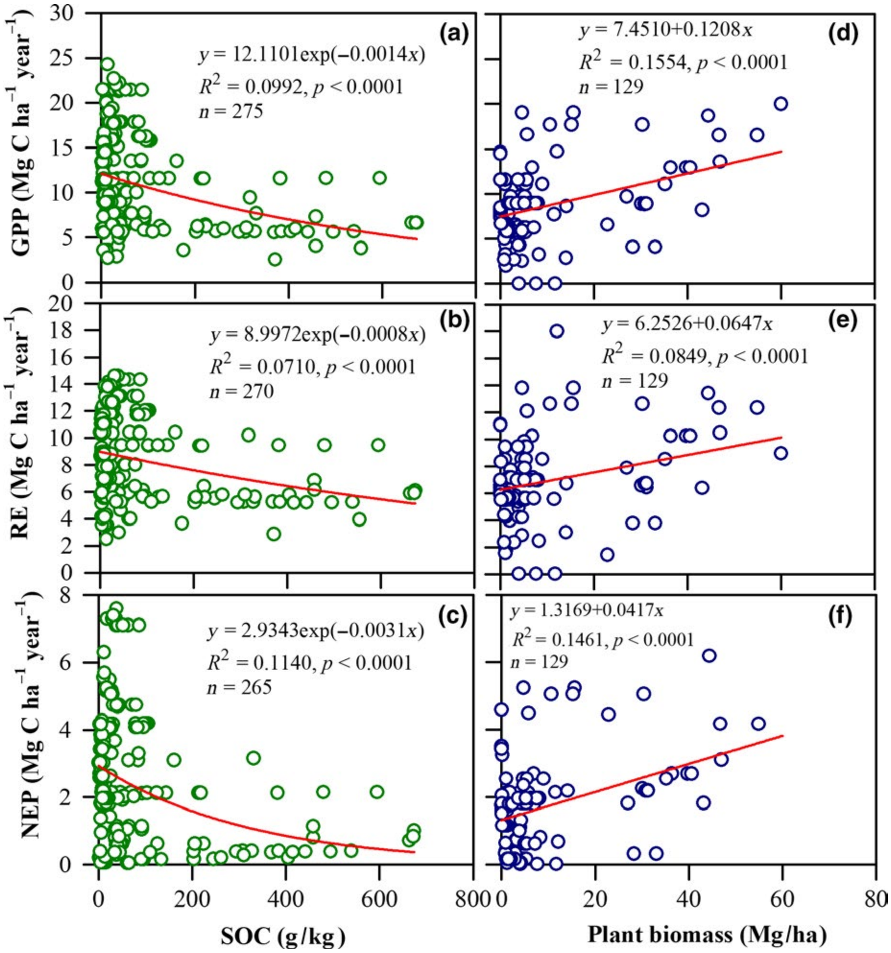

FIGURE 6 Relationships of soil organic carbon (SOC) and plant biomass with GPP (a, d), RE (b, e), NEP (c, f) in China's wetlands. Note: SOC is 0–20 cm soil organic carbon content (g/kg); plant biomass includes shoot and root biomasses [Colour figure can be viewed at wileyonlinelibrary.com]

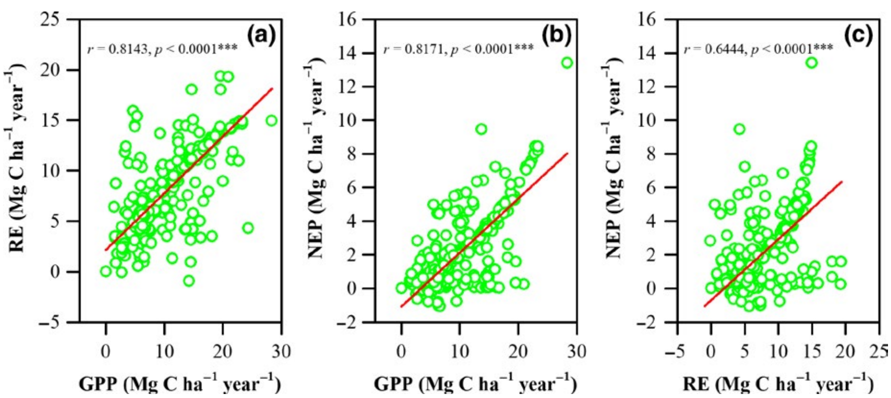

FIGURE 7 Relationships among GPP, RE and NEP in China's wetlands. ***indicates correlation coefficients (r) are significant at $p < 0.001$, $n = 600$ [Colour figure can be viewed at wileyonlinelibrary.com]

amount of C (10.20 Pg) due to its largest area (21.73×10^6 ha), accounting for 40.7% of the total wetland area and is mainly distributed in northeast China (high latitude) and the Qing-Tibetan Plateau (State Forestry Administration, 2015) (Figure 5). Previous studies reported that C stocks were highly correlated with wetland types (Lu et al., 2017), location (Nahlik & Fennessy, 2016), cool temperature and anaerobic conditions (Nahlik & Fennessy, 2016). The current study showed that C stocks were overall positively correlated with latitude and altitude, reflecting an indirect effect of location links to C accumulation in China's wetlands (Table 2). Wetlands at high latitude and altitude stored the most C, related to the abundance of wetlands in northeast China and the Qing-Tibetan Plateau where cool temperatures and anaerobic conditions provide conditions that enhance C accumulation (State Forestry Administration, 2015). By contrast, coastal wetlands cover the smallest area along the coast of the South and East China Seas to the Bohai Sea (Figure 1), where mild climatic conditions enhance decomposition of organic matter and diminish C accumulation (Moinet et al., 2018), although they are generally believed to have the highest C stored of any ecosystem worldwide (Lu et al., 2017). In addition, some studies reported that river and lake wetlands close to coastal regions are “neutral pipes” for C accumulation that merely convey terrestrial C to oceans (Aufdenkampe et al., 2011; Cole et al., 2007), whereas inland wetlands

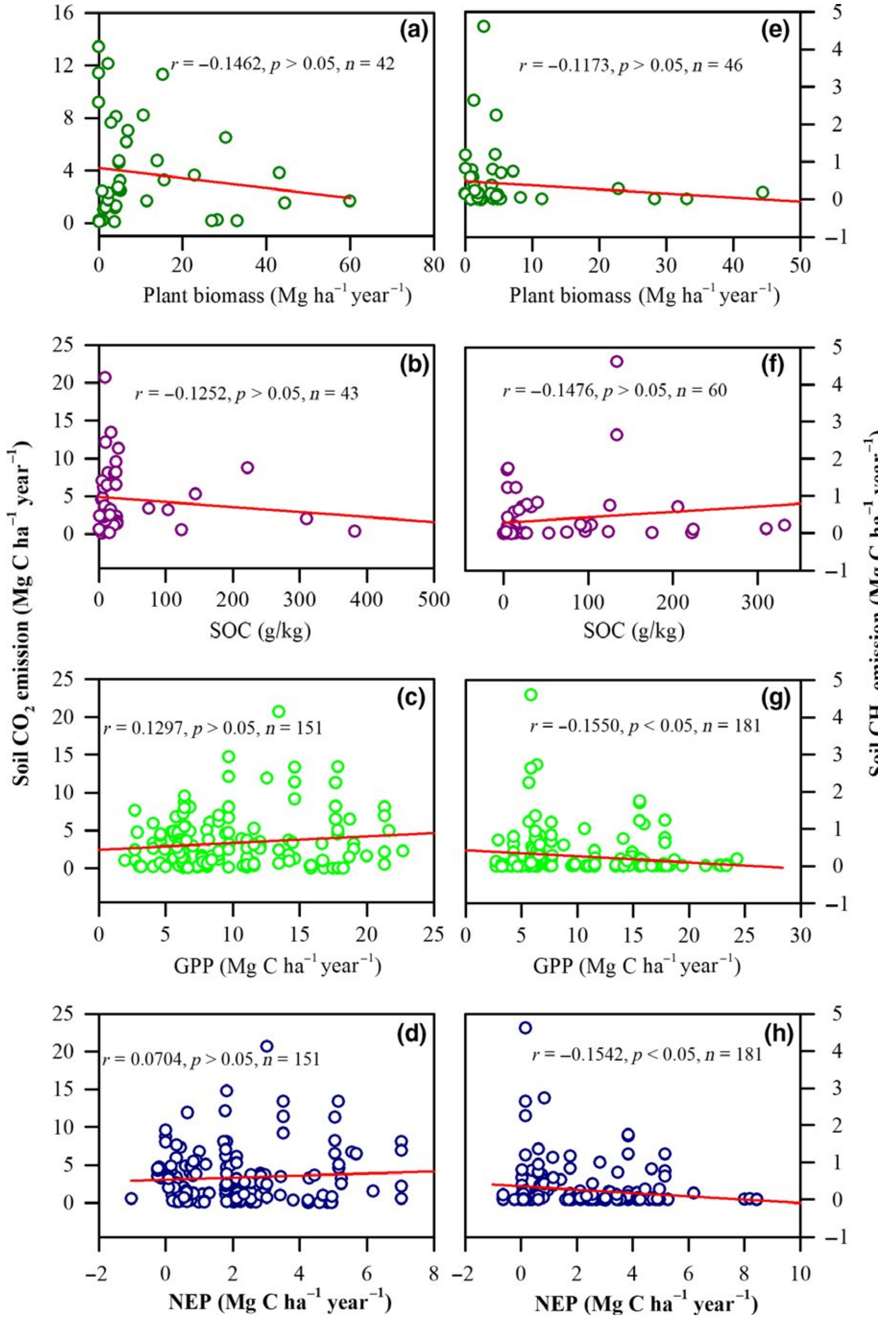

FIGURE 8 Relationships between plant biomass (a, e), soil organic carbon (SOC) (b, f), gross primary production (GPP) (c, g) and net ecosystem production (NEP) (d, h) and soil carbon dioxide (CO_2) and methane (CH_4) emissions in China's wetlands. Note: SOC is of 0–20 cm soil (g/kg); plant biomass includes shoot and root biomasses. $p < 0.05$ indicates correlations are significant at $p < 0.05$ [Colour figure can be viewed at wileyonlinelibrary.com]

are usually closed basins that cannot transport C to oceans (Euliss et al., 2014; Pennock et al., 2010; Tangen, Finocchiaro, & Gleason, 2015). China has inland wetlands mainly in the Qing-Tibetan Plateau with closed-basin ecosystems, and has coastal wetlands close to oceans with open ecosystems—the different roles of inland and coastal wetlands in C accumulation are interesting and need further study.

In addition, precipitation is the most important factor affecting soil C stocks (Tuo et al., 2018), which largely depends on both vegetation productivity (Iglesias, Barchuk, & Grilli, 2012) and organic matter decomposition (Campo & Merino, 2016; Carvalhais et al., 2014). However, the effect of precipitation on the C cycle is highly variable among studies. Some studies indicated that soil C stock is positively correlated with precipitation (Satrio, Gandaseca, Ahmed, & Majid, 2009; Shen, Jenerette, Hui, Phillips, & Ren, 2008), mainly due to larger above-ground biomass and thus more litter production with higher precipitation (Banning, Grant, Jones, & Murphy, 2008; Campo & Merino, 2016; Iglesias et al., 2012). Other studies argued that increased precipitation would decrease soil C stock, mainly due to higher decomposition of organic matter (Campo & Merino, 2016) and increases in soil water availability (Wan, Norby, Ledford, & Weltzin, 2007). In our study, the negative correlation between soil C and precipitation may be explained from higher precipitation leading to higher decomposition of litter and soil C. Furthermore, precipitation is positively correlated with temperature in China (Miao, Sun, Borthwick, & Duan, 2016)—high precipitation is generally coupled with high temperature in south China, where high temperature in turn increases soil C deposition despite the

TABLE 2 Pearson correlations between location (longitude, latitude and altitude), climate (MAP, mean annual precipitation, and MAT, mean annual temperature) and plant and soil carbon (C) stocks, soil carbon dioxide (CO_2) and methane (CH_4) emissions, gross primary production (GPP), ecosystem respiration (RE) and net ecosystem production (NEP)

	Longitude (E)	Latitude (N)	Altitude (m)	MAP (mm)	MAT (°C)
Plant biomass C (Mg/ha)	0.098 (*n* = 133)	−0.159 (*n* = 133)	−0.101 (*n* = 133)	0.198* (*n* = 133)	0.249** (*n* = 133)
0–20 cm Soil C stock (Mg/ha)	0.070 (*n* = 286)	0.181** (*n* = 286)	0.494*** (*n* = 286)	−0.327*** (*n* = 286)	−0.055 (*n* = 286)
0–100 cm Soil C stock (Mg/ha)	0.060 (*n* = 286)	0.152** (*n* = 286)	0.284*** (*n* = 286)	−0.385*** (*n* = 286)	−0.104 (*n* = 286)
Soil CO_2 emission (Mg CO_2-C/ha)	0.204* (*n* = 151)	−0.100 (*n* = 151)	−0.075 (*n* = 151)	0.06 (*n* = 151)	0.172* (*n* = 151)
Soil CH_4 emission (Mg CH_4-C/ha)	0.047 (*n* = 181)	0.152* (*n* = 181)	0.052 (*n* = 181)	0.151* (*n* = 181)	0.169* (*n* = 181)
GPP (Mg/ha)	0.150*** (*n* = 595)	−0.678*** (*n* = 595)	−0.474*** (*n* = 595)	0.839*** (*n* = 595)	0.776*** (*n* = 595)
RE (Mg/ha)	0.201*** (*n* = 590)	−0.620*** (*n* = 590)	−0.491*** (*n* = 590)	0.804*** (*n* = 590)	0.760*** (*n* = 590)
NEP (Mg/ha)	0.139** (*n* = 582)	−0.575*** (*n* = 582)	−0.472*** (*n* = 582)	0.774*** (*n* = 582)	0.672*** (*n* = 582)

*Correlation significant at $p < 0.05$ (two-tailed); **Correlation significant at $p < 0.01$ (two-tailed); ***Correlation significant at $p < 0.001$ (two-tailed).

TABLE 3 Linear regressions between geographic location (longitude, latitude and altitude), climate (MAP, mean annual precipitation, and MAT, mean annual temperature), 0–20 cm SOC, plant biomass and 0–20 and 0–100 cm soil carbon (C) stocks, soil CO_2 and CH_4 emissions, GPP, RE and NEP

Variables (*y*)	Models	R^2	Sig. (*p*)	*n*
0–20 cm Soil C stock (Mg/ha)	$y = -207.091 + 5.259x1 - 0.206x2 + 0.014x3 + 2.615x4 - 0.050x5 + 0.627x6 - 0.061x7$	0.832	0.000***	72
0–100 cm Soil C stock (Mg/ha)	$y = -521.893 + 13.253x1 - 0.520x2 + 0.035x3 + 6.589x4 + 0.125x5 + 1.581x6 - 0.153x7$	0.813	0.000***	72
Soil CO_2 emission (Mg CO_2-C/ha)	$y = -90.955 + 0.806x1 + 0.426x2 + 0.005x3 + 2.007x4 - 0.008x5 + 0.058x6 - 0.074x7$	0.493	0.209	21
Soil CH_4 emission (Mg CH_4-C/ha)	$y = -4.4 + 0.001x3 + 0.385x4 + 0.002x5 + 0.018x6 - 0.012x7$	0.389	0.403	16
GPP (Mg/ha)	$y^a = 34.748 - 0.516x1 - 0.025x2 - 0.002x3 - 0.072x6^a + 0.006x7$	0.962	0.000***	72
RE (Mg/ha)	$y^b = 17.308 - 0.328x1 - 0.032x2 - 0.001x3 + 0.001x7$	0.902	0.000***	72
NEP (Mg/ha)	$y^c = 13.960 - 0.159x1 - 0.046x2 - 0.001x3 + 0.013x6^a + 0.005x7$	0.909	0.000***	72

Note. *x*1, latitude (N); *x*2, longitude (E), *x*3, altitude (m); *x*4, MAP (mm); *x*5, MAT (°C), *x*6, 0–20 cm SOC (g/kg), *x*7, plant biomass (Mg/ha).
n is the number of observations entered into the models.
a, b and c indicate that *x*4 and *x*5 are removed from models due to some data of GPP, RE and NEP are estimated by MAP and MAT using models of Lu et al. (2017).
[a]Indicate *x*6 is 12.1101exp(−0.0014SOC) and 2.9343exp(−0.0031SOC) in the models a and b, also, SOC is 0–20 cm soils. ***$p < 0.001$.

high primary productivity. Because of the complex processes involved in forming soil C stocks under different precipitation conditions, further studies on the trade-off between primary productivity and soil C decomposition are needed.

4.2 | C uptake in China's wetland ecosystems

Our results suggested that China's wetlands were an important net CO_2 sink at the national scale, but annual NEP exhibited large variability among wetland types (Figure 3, Table 1). Coastal wetlands had highest NEP (Figure 3) due to higher productivity (Lu et al., 2017), indicating that coastal wetlands had higher CO_2 sequestration capacity than river, lake and marsh wetlands in China. It should be noted that land cover of coastal wetlands represented only 10.8% of the total wetland (Table 1), which was much lower than that of river, lake and marsh (89.2%). When annual NEP rates were applied at the national scale, annual net CO_2 uptake in marsh was the largest, followed by lake, river and coastal wetlands (Table 1). Therefore, when taking NEP and distribution area together, marsh, lake and river wetlands represented a larger CO_2 sink than coastal wetlands (Table 1). Compared with other regions, the NEP in China's wetlands (2.23 Mg ha^{-1} $year^{-1}$) was higher than that of the USA, Canada, Europe and Russia, as well as the global average level (1.79 Mg ha^{-1} $year^{-1}$) (Table 4). This can be attributed to distribution area, geographic location and climatic conditions of China's wetlands.

Lu et al. (2017) found that GPP, RE and NEP of inland and coastal wetlands globally were determined by MAP and MAT. In addition to the positive correlations with MAP and MAT, we also found that

GPP, RE and NEP were significantly correlated with factors of location (latitude, longitude and altitude), SOC and plant biomass. Our linear regression analyses showed that factors of geographic location, SOC and plant biomass explained the variation of GPP, RE and NEP in China's wetlands, and latitude was the most influential factor (Table 3). Geographic location governs not only MAP and MAT (Yao, Yang, Mao, Zhao, & Xu, 2016), but other environmental factors such as radiation (Yu, Wang, & Yang, 2018). Since geographic location reflects the MAP and MAT to some degree, it can be used to indirectly explain the variation of these C budgets, partly reflecting relationships of GPP, RE and NEP with MAT and MAP (Lu et al., 2017). It should be noted that compared with Lu et al. (2017), we improved the models explaining the variation of GPP, RE and NEP through (a) including more wetland types in equations, particularly river and lake wetlands, indicating that the models could be better applied to different wetlands and (b) adding plant and soil factors into the models as well as climate factors, thus significantly improving the accuracy of models in wetlands in the context of global climate change. As a result, compared with Lu et al. (2017), the coefficient of determination (R^2) of the models increased from 0.71, 0.54 and 0.57 to 0.93, 0.90 and 0.91 for GPP, RE and NEP respectively.

4.3 | Gaseous C losses in China's wetlands

Although functioning as an overall net C sink, China's wetlands were also an atmospheric C source by emitting CO_2 and CH_4 from soil, but the magnitude of emissions substantially differed with wetland type (Figure 4, Table 1). Coastal wetlands had the largest C source with highest soil CO_2 and CH_4 emissions in China, followed by marsh wetlands, and lake and river wetlands which had the smallest soil CO_2 and CH_4 emissions (Figure 4, Table 1). The largest C emissions in coastal wetlands can be attributed to their tropical location, widespread presence of mangrove vegetation, dynamic flooding along the coast line and mild conditions. Locations with high altitude and latitude (Bousquet et al., 2011), low annual temperature and a short growing season (Zhang et al., 2008) for meadow species might result in low soil CO_2 and CH_4 emissions in marsh wetlands. Additionally, the low CO_2 and CH_4 emissions in river and lake wetlands could be associated with their long-term inundation conditions and lack of macrophytes (Laanbroek, 2010). At the national scale, marsh wetland had the largest annual C emission, coastal wetland was the second largest and river and lake were the smallest C sources (Table 1), and were associated with wetland types with different distribution areas. Therefore, marsh is likely the most important source of soil C emission in China's wetlands. Soil CO_2 emission in China's wetlands was lower than that of the USA, but higher than Canada, Europe and Russia, as well as the global average (Table 4). Moreover, soil CH_4-C emission (0.29 Mg C ha^{-1} year^{-1}) in China's wetlands was lower than that of the USA, Canada and South America, as well as the average world level (0.48 Mg C ha^{-1} year^{-1}), but higher than that of Europe (Table 4). The discrepancy might be because China has different climatic and soil conditions to these other countries, leading to differences in distribution areas of each wetland type.

TABLE 4 Summary of carbon (C) budgets of wetlands in the world

	Plant C (Mg/ha)	Soil C (0–100 cm) (Mg/ha)	Ecosystem C (Mg/ha)	GPP (Mg ha^{-1} year^{-1})	RE (Mg ha^{-1} year^{-1})	NEP (Mg ha^{-1} year^{-1})	Soil CO_2-C emission (Mg ha^{-1} year^{-1})	Soil CH_4-C emission (Mg ha^{-1} year^{-1})
China[a]	7.12	225.13	232.25	10.26	7.93	2.23	3.38	0.29
United States[b]	–	–	–	0.34–20.93 (9.17)	2.19–20.82 (8.82)	0.13–9.49 (2.22)	2.28–4.57 (3.43)	0.01–1.65 (0.61)
Canada[b]	–	–	–	3.77–19.91 (7.88)	3.55–17.37 (6.31)	0.39–6.06 (1.58)	0.01–0.09 (0.05)	0.07–0.74 (0.37)
South America[b]	–	234	344	–	–	–	–	0.3–2.63 (1.36)
Europe[b]	–	720–1,900 (1,049)	1,319.2[§]	2.01–12.15 (5.57)	1.29–8.86 (4.49)	0.05–6.36 (1.17)	0.95–1.25 (1.10)	0.03–0.47 (0.12)
Russia[b]	–	–	–	1.17–2.32 (1.74)	0.99–1.41 (1.20)	0.19–0.92 (0.42)	0.60–4.13 (2.67)	0.02–0.16 (0.07)
Australia[b]	–	–	220–550 (395)	–	–	3.11	–	0.2
Global[c]	0.5–13.5 (7.0)	–	–	6.92	5.57	1.79	2.26	0.48

Note. "–" indicate no data collected in Appendix Dataset S2.
The interval data represents the maximum and minimum values in the Appendix Dataset S2 of each category, and mean values indicated in the "()".
[a]Data estimation from this study. [b]Data sources of each region listed in Appendix Dataset S2. [c]Mean of dataset.

Soil CO_2 and CH_4 emissions are associated with the types of wetlands (Mitsch et al., 2012), geographic location (Nisbet, Dlugokencky, & Bousquet, 2014), together with other controlling factors such as temperature, precipitation, topography and soil properties (Lu et al., 2017). Our results showed that longitude and MAT were positively correlated with soil CO_2 emission; and latitude, MAP and MAT were positively correlated with soil CH_4 emission. The results suggest that effects of climate factors on C emissions were controlled by geographic locations that differed for the types of wetlands. The highest soil CH_4 emission in coastal wetlands in China could be well explained by their geographic location, in which most were situated in tropical regions (Bousquet et al., 2011) along the coastal zone of the South and East China Seas with mild climate, together with mangrove vegetation of high GPP (Lu et al., 2017) and daily changes in flooding conditions (Elberling et al., 2011). The positive effect of latitude on CH_4 emission indicates that marsh mainly distributed in north and highland China was also an important source of CH_4 to the atmosphere (Panneer, Natchimuthu, Arunachalam, & Bastviken, 2014).

Linear regressions showed that 49.3 and 38.9% of variation in soil CO_2 and CH_4 emissions, respectively, were explained by controlling factors of geographic location (longitude, latitude and altitude), climate (MAT and MAP), as well as soil (SOC) and plant biomass. However, regression models were not significant. The insignificant relationships linking soil CO_2 and CH_4 emissions with factors of the geographic location (longitude, latitude and altitude), climate (MAT and MAP), as well as soil (SOC) and plant biomass, suggesting that these factors did not provide enough information to simulate the two emissions. We suspect that this was likely because datasets in microbial communities were unavailable in the models, and these are generally considered to be another factor affecting rates of C decomposition and the balance of released CO_2 versus CH_4 (Mann et al., 2015; Vonk et al., 2015). Therefore, in an effort to better simulate variation in soil CO_2 and CH_4 emissions, further incorporating soil microbial processes into models is needed in China's wetlands. Particularly, it is crucial to improve our understanding of the contribution of the microbial community to controlling the soil C emission combined with related biophysical factors at the national scale.

4.4 | Implication for wetland C management in China

Overall, we found that C budgets differed remarkably as a function of wetland type, together with environmental conditions related to geographic location, climate, soil and vegetation. This provides important insights into C management practice for different wetlands in the face of different anthropogenic disturbances in China. Marsh, mainly distributed in the Qing-Tibetan Plateau and northeast China (Figure 1), contains the largest net C pool in China. Therefore, preserving marshes in the Tibetan Plateau and northeast China are the two main areas of C management in China. Since overgrazing mainly threatens C budget on the Qing-Tibetan Plateau (Chen et al., 2013b), scientifically determining an optimum grazing intensity according to carrying capacity, is a key to maintaining marsh C on this plateau. Agricultural encroachment has caused a large loss of marsh in northeast China (Meng et al., 2017), and preventing further encroachment by agriculture, as well as adopting a strategy of turning reclaimed wetland to natural wetland, are important ways to protect marsh C in this region. Reclamation of coastal wetlands for agricultural and industrial uses over the past several centuries has caused a great loss of coastal wetlands (Ma et al., 2014), leading to changes in C budgets, particularly CH_4 emissions (Nisbet et al., 2014). Therefore, comprehensive measures focused on wetland management are required to achieve a target of "no net loss" and sustainable use of coastal wetlands at both national and local levels. Pollution and eutrophication are the main factors threatening the environments of river and lake wetlands (Ministry of Environmental Protection of the People's Republic of China, 2016). To prevent pollution and eutrophication, comprehensive protection measures are needed to strengthen supervision and administration of environmental protection in river and lake basins, promote the reduction of total discharge amount of pollutants from industrial and domestic wastewater, and protect and improve the environment to maintain C budgets of river and lake wetlands. In addition, climate warming is an important factor affecting wetlands. It causes melting glaciers in western China, permafrost thawing in northeast China and the Qing-Tibet Plateau, rising sea levels in coastal wetland regions (Meng et al., 2017), changing conditions such as hydroperiod, vegetation and soil that regulate C intake and emission (Jin et al., 2015), and finally the net C stocks in China's wetlands. Mitigating anthropogenic disturbance to achieve a target of "no net loss" and restoring degraded wetlands can help maintain C budgets and contribute to global efforts for mitigating global warming and climate change.

4.5 | Uncertainty analysis

Compared to previous studies (Chen et al., 2013a; Jin et al., 2015; Wang et al., 2014; Xu et al., 2018; Zheng et al., 2013), we offered the most accurate estimate on overall C budgets related to C stocks, uptakes and emissions based on unbiased wetland types with distribution area data issued by the Chinese State Forestry Administration (2015) and results from a relatively large number of published field studies across the whole areas of China (Appendix Word S1). However, some factors, including calculation methods for C stocks, data collected in current studies and models used to obtain ecosystem C fluxes, might introduce uncertainties into the results. The main potential uncertainties are discussed as follows.

4.5.1 | Uncertainty in C stock estimations

Uncertainty in estimation of ecosystem C stocks can be derived from estimation of plant C, from estimation of soil C or both. Current estimations of plant C stocks mainly involve grassland (Deng et al., 2017; Fang et al., 2010; Piao, Fang, Zhou, Tan, & Tao, 2007) and forest ecosystems (Deng & Shangguan, 2017; Guo, Hu, Pan, Birdsey, & Fang, 2014) in China, and methods for estimating wetland plant C stock are still lacking, despite the important role of wetland plants in the C cycle. In our study, based on dominant plants in four wetland types, we divided the species into herbaceous (submerged, floating-leaved, emergent species and other grasses) and woody plants (shrub and forest species), and set two respective coefficients for estimating their C stocks. Because wetlands in China have a large amount of different dominant plants, even within the same wetland type, setting two coefficients for estimating plant C stock will increase uncertainties in the results, particularly when applying them to a national scale. Therefore, a fine-grained coefficient incorporating dominant plant types in wetlands across China is needed for more accurate estimation of plant C stock. Furthermore, the insufficient dataset of below-ground biomass, obtained from both direct and indirect measurement using different root:shoot ratios, as well as linkages between above- and below-ground biomass despite their isometric relationship (Yang, Fang, Ma, Guo, & Mohammat, 2010), also increase uncertainty of estimates of plant C. Additionally, depth of wetland soils is known to be crucial in estimation accuracy for soil C stock (Nahlik & Fennessy, 2016). Depths of wetland soils vary significantly in different conditions, and are usually more or less than 100 cm across China's wetlands (State Forestry Administration, 2015). However, data for soil depths are unavailable in previous publications (Appendix dataset S1). Due to the lack of reliable soil depths, our results of soil C stocks (0–100 cm) in China's wetlands might be over- or underestimated. Simultaneously considering both plant and soil C estimations, the uncertainties in our results of ecosystem C stocks in China's wetlands would be further increased. The accuracy of the current study is limited due to the uneven distribution of data collected in current studies. Relatively fewer field study data on rivers may add uncertainty at the national scale, despite representing only 19.8% of the total wetland area.

4.5.2 | Uncertainty in estimation of ecosystem C fluxes

We were unable to obtain sufficient data on China's wetland ecosystem C fluxes, including GPP, RE and NEP, due to limited numbers of studies and EC study sites compared to those for terrestrial ecosystems across China. We used some data of GPP, RE and NEP estimated by Lu et al. (2017) (Appendix dataset S1) to estimate ecosystem C fluxes in China's wetlands. Although their model could be well applied to inland wetlands including marsh (freshwater swamp marsh and shrub swamp, peatland and alpine tundra wetlands), and coastal wetlands (intertidal marshes and forested wetlands, coastal freshwater marshes and forests, and anthropogenic perturbations coastal wetlands) worldwide, it might be inadequate for estimating ecosystem C fluxes of river and lake wetlands, resulting from the different environmental conditions of marsh and coastal wetlands. Therefore, considerable knowledge gaps concerning ecosystem C uptakes and emissions in China's wetlands exist and suggests that further research is needed in major wetland types in different regions, particularly river wetlands like the Yellow and Yangtze Rivers, and lake wetlands of the Qing-Tibet Plateau.

4.5.3 | Uncertainty in factors explaining C budgets

In our study, geographic location was the most important factor explaining GPP, RE and NEP; the R^2 of the models were much higher than that using only MAP and MAT (Lu et al., 2017). Although the models we adopted improved explanation of the variation in GPP, RE and NEP, there is still uncertainty in predictions. Factors of geographic location are stable, and cannot be used as changing environment variables, like MAP and MAT, to predict changes in C budgets of China's wetlands, particularly in the context of global climate change, because geographic location not only governs MAP and MAT, but also other environmental factors like radiation and atmospheric pressure. In this respect, geographic location is only an indirect factor explaining C budgets, and further studies on direct environmental factors controlling these variations are needed to improve accuracy of predicting C budgets in the context of ongoing climate change.

Despite these limitations, the updated estimation we provided here is informative for understanding the disproportionately large and regionally variable C stores in China's wetlands, as well as the important role in regulating global C cycle. Additionally, our results can help government adopt corresponding strategies to improve wetland C management practice in the context of global climate change.

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CONFLICTS OF INTEREST

The authors declare no competing interests.

AUTHOR CONTRIBUTIONS

X.D. and L.D. designed research; X.D. collected data; X.D. and L.D. performed research; C.H. and L.D. contributed new analytic tools; L.D. analyzed data; X.D., L.D., D.-G.K., and K.T. wrote the paper.

ORCID

Lei Deng https://orcid.org/0000-0002-5898-5100

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