Received: 28 September 2019 | Accepted: 13 November 2019
DOI: 10.1111/gcb.14933

PRIMARY RESEARCH ARTICLE

Global Change Biology WILEY

Conversion of coastal wetlands, riparian wetlands, and peatlands increases greenhouse gas emissions: A global meta-analysis

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Funding information
Chinese Ministry of Science and Technology, Grant/Award Number: 2016YFE0133700; Royal Netherlands Academy of Arts and Sciences; National Natural Science Foundation of China, Grant/Award Number: 41871088 and 41571083; National Key R&D Program of China, Grant/Award Number: 2017YFC0506001; Key Project of Shanghai Science & Technology Committee, Grant/Award Number: 19DZ2210200

Abstract

Land-use/land-cover change (LULCC) often results in degradation of natural wetlands and affects the dynamics of greenhouse gases (GHGs). However, the magnitude of changes in GHG emissions from wetlands undergoing various LULCC types remains unclear. We conducted a global meta-analysis with a database of 209 sites to examine the effects of LULCC types of constructed wetlands (CWs), croplands (CLs), aquaculture ponds (APs), drained wetlands (DWs), and pastures (PASs) on the variability in CO_2, CH_4, and N_2O emissions from the natural coastal wetlands, riparian wetlands, and peatlands. Our results showed that the natural wetlands were net sinks of atmospheric CO_2 and net sources of CH_4 and N_2O, exhibiting the capacity to mitigate greenhouse effects due to negative comprehensive global warming potentials (GWPs; −0.9 to −8.7 t CO_2-eq ha^{-1} year^{-1}). Relative to the natural wetlands, all LULCC types (except CWs from coastal wetlands) decreased the net CO_2 uptake by 69.7%–456.6%, due to a higher increase in ecosystem respiration relative to slight changes in gross primary production. The CWs and APs significantly increased the CH_4 emissions compared to those of the coastal wetlands. All LULCC types associated with the riparian wetlands significantly decreased the CH_4 emissions. When the peatlands were converted to the PASs, the CH_4 emissions significantly increased. The CLs, as well as DWs from peatlands, significantly increased the N_2O emissions in the natural wetlands. As a result, all LULCC types (except PASs from riparian wetlands) led to remarkably higher GWPs by 65.4%–2,948.8%, compared to those of the natural wetlands. The variability in GHG fluxes with LULCC was mainly sensitive to changes in soil water content, water table, salinity, soil nitrogen content, soil pH, and bulk density. This study highlights the significant role of LULCC in increasing comprehensive GHG emissions from global natural wetlands, and our results are useful for improving future models and manipulative experiments.

KEYWORDS
coastal wetland, global warming potential, greenhouse gas emission, inland wetland, land-use change, meta-analysis

1 | INTRODUCTION

Currently, global climate warming is one of the most serious environmental problems due to increasing emissions of greenhouse gases (GHGs) into the atmosphere. The concentrations of atmospheric carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) reached new highs in 2017, at 405.5 ppm, 1,859.0 ppb, and 329.9 ppb, respectively (Le Quéré et al., 2018; World Meteorological Organization, 2018). The increase of CH_4 and N_2O concentration has more severe impact than CO_2 because the global warming potentials (GWPs) of CH_4 and N_2O are 34 and 298 times greater than the contribution of CO_2 to global warming over a 100 year scale, respectively (IPCC, 2013). In addition to fossil fuel combustion, land-use/land-cover change (LULCC) has been identified as the second largest source of anthropogenic emissions of GHGs due to its impacts on the global biogeochemical cycle and hydrological properties of terrestrial and aquatic ecosystems (Arneth et al., 2017; Houghton et al., 2012; IPCC, 2013).

Although natural wetlands (mainly including coastal wetlands, riparian wetlands, and peatlands) only account for 5%–8% of the earth's land area, they store approximately 20%–30% of the soil carbon (C) on earth and thus play a crucial role in mitigating global climate change (Chmura, Anisfeld, Cahoon, & Lynch, 2003; Lu et al., 2017; McLeod et al., 2011; Mitsch et al., 2013; Nahlik & Fennessy, 2016). However, the area of global natural wetlands has been decreasing rapidly in recent decades due to anthropogenic activities (e.g., LULCC), which potentially cause C and nitrogen (N) losses in the forms of CO_2, CH_4, and N_2O into the atmosphere (Duke et al., 2007; MacKinnon, Verkuil, & Murray, 2012; Pendleton et al., 2012; Waycott et al., 2009). Coastal hardening and land development resulted in the loss of more than 16% of coastal wetlands worldwide from 1984 to 2016 (Murray et al., 2019). Approximately 26% of the area of global riparian wetlands and peatlands has been drained for agriculture (e.g., croplands [CLs] and pastures [PASs]) since 1985 (Zedler & Kercher, 2005), and the annual rate of loss of natural inland wetlands has been approximately 1.2% since the beginning of the 20th century, mainly due to LULCC (Davidson, 2014).

GHG emissions in the natural wetlands are complicated by plant type (Ge, Guo, Zhao, & Zhang, 2015; Inglett, Inglett, Reddy, & Osborne, 2012; Ström, Mastepanov, & Christensen, 2005; Tong et al., 2012), primary productivity (Knox et al., 2014; Saunders, Kansiime, & Jones, 2012), water table (WT; Furukawa, Inubushi, Ali, Itang, & Tsuruta, 2005; Hatala et al., 2012; Tong, Huang, Hu, & Jin, 2013), and saline level (Hu, Ren, et al., 2017; Krauss et al., 2016). The WT is high and the sediment is anoxic in the natural wetlands, inhibiting decomposition of litters; therefore, a large amount of organic matter is sequestered (Mitsch et al., 2013; Nahlik & Fennessy, 2016). LULCC usually alters the balance of net ecosystem CO_2 exchange (NEE) and the emissions of CH_4 and N_2O by directly involving simultaneous changes in multiple environmental factors and biotic processes of plants and hydrological regime and soil microbial structures (Eickenscheidt, Heinichen, & Droesler, 2015; Kandel, Lærke, & Elsgaard, 2018; Kasimir, He, Coria, & Nordén, 2018; Kløve, Sveistrup, & Hauge, 2010; Tangen, Finocchiaro, & Gleason, 2015). In the natural coastal wetlands, tidal inundation and high salinity (Sal) levels are the key environmental factors that impact the dynamics of GHGs (Poffenbarger, Needelman, & Megonigal, 2011). Enclosure and drainage induced by LULCC reduce or interrupt the frequency of flooding and salt water and then affect the production and emissions of CO_2 and CH_4; changes in tides also lower nitrification and denitrification processes associated with N_2O emissions (Fernández, Santín, Marquínez, & Álvarez, 2010; Pendleton et al., 2012). In contrast, Yang et al. (2017) reported that the conversion of natural coastal wetlands to aquaculture ponds (APs) led to a significant increase in CH_4 emissions by 10 times, with a 25% decrease in N_2O emissions due to the higher WT and lower Sal and dissolved nitrogen concentration in the soil. Compared to that in terrestrial ecosystems, the soil sulfate concentration in coastal wetlands is generally higher, and CH_4 production is restrained (Chmura et al., 2003; Hadi et al., 2005). When the coastal wetlands were enclosed and converted to the CLs, decreased Sal and increased aboveground plant biomass enhanced CH_4 and N_2O emissions and the ecosystem respiration (ER) of CO_2 efflux (Olsson et al., 2015). Moreover, the replacement of dominant vegetation and biomass removal changed a CO_2 sink into a strong carbon source (Han et al., 2014). Some natural wetlands are reconstructed through hydrological engineering and vegetation establishment, and these processes are becoming a new type of disturbance to coastal wetlands (Craft et al., 2003; Crooks, Herr, Tamelander, Laffoley, & Vandever, 2011). The type of constructed wetland (CW) with permanent flooding is generally considered to increase CO_2 uptake while stimulating CH_4 release (Hu et al., 2016; Krauss, Whitbeck, & Howard, 2012). Zhong et al. (2016) found that CWs still functioned as CO_2 sinks, but the CO_2 sequestration capacity was lower than that of natural coastal wetlands.

The drainage of natural wetlands will alter the water regime and nutrient availability (Audet, Elsgaard, Kjaergaard, Larsen, & Hoffmann, 2013; Krauss et al., 2012), and a reduced WT after drainage generally leads to aerobic conditions and stimulates the decomposition of soil organic carbon (SOC) and nitrogen (or peat oxidation), thus enhancing CO_2 and N_2O emissions (Berglund & Berglund, 2011; Gleason, Tangen, Browne, & Eliss, 2009; Huang et al., 2010; Salm et al., 2012) while decreasing CH_4 emissions (Knox et al., 2014; Tangen et al., 2015). Phillips and Beeri (2008) investigated the spatial variability in emission of GHGs in the hydrophyte zone of a riparian wetland and found that ER was higher in the drained zone (including prairies, PASs, and CLs) than in the natural wetland zone; however, CH_4 emissions showed the opposite trend, and N_2O emissions were highly variable. Although some measurements showed that the CLs (especially rice paddies) and PASs still had the capacity for CO_2 sequestration, these land-use types clearly acted a GHG sources during the harvest period in the CLs and the removal of aboveground biomass (AGB) through grazing in the PASs (Frank & Dugas, 2001; Knox et al., 2014; Saunders et al., 2012). Knox et al. (2014) and Saunders et al. (2012) also found that the drained peatlands for agricultural land use (e.g., PASs and CLs) turned CO_2 sinks into net emission sources, with significantly decreased CH_4 emissions. Agricultural management practices (e.g., tillage, fertilization, and irrigation) also had a large impact on GHG emissions in the drained wetlands (DWs). For instance, the application of nitrate fertilizers in CLs increased the ER of CO_2 efflux and N_2O emissions caused by the high availability of inorganic nitrogen (Snyder, Bruulsema, Jensen, & Fixen, 2009) while inhibiting CH_4 production (Han et al., 2014; Jiang, Wang, Hao, & Song, 2009). Similar to the PASs developed from the natural wetlands, the application of ammonia-based fertilizers (e.g., dung and urine patches) decreased CH_4 oxidation (Mosier, Schimel, Valentine, Bronson, & Parton, 1991; Saggar, Hedley, Giltrap, & Lambie, 2007) and enhanced N_2O emissions (Pärn et al., 2018; van Groenigen, Velthof, Bolt, Vos, & Kuikman, 2005).

With an abundance of research achievements to draw upon, we conducted a global meta-analysis that synthesized 209 site-based studies. We focused on the variability in GHG emissions of CO_2, CH_4, and N_2O from the natural coastal wetlands, natural riparian wetlands (NRWs), and natural peatlands (NPLs) to five worldwide popular LULCC types across the globe. Changes in the GWPs of GHG emissions in the natural wetlands and converted man-made landscapes were also determined. We hypothesized that the conversion of natural wetlands into other land-use types would increase the regional greenhouse effect by changing the hydrological and edaphic conditions, primary production, and management practices. The main objectives of this study were to quantify the effect of LULCC on GHG emissions in natural wetland ecosystems and to detect the key environmental variables in relation to changes in GHG emissions and the potential resulting GWP effects since LULCC events.

2 | MATERIALS AND METHODS

2.1 | Data sources and processing

The data used in this study were drawn from peer-reviewed articles collected from the Web of Science (http://www.isiknowledge.com/) with the following keywords: TS = (NEE * OR CO_2 * OR CH_4 * OR N_2O) AND TS = (coastal wetland * OR marsh * OR mangrove * OR freshwater wetland * OR riparian wetland * OR peatland * OR fen * OR bog) AND TS = (land use * OR agriculture * OR cropland * OR constructed wetland * OR pasture * OR aquaculture) with time span of 1950–2018. A total of 887 papers matched the keywords. To avoid bias in publication selection, four basic criteria were followed to select appropriate studies. (a) The selected experiments were conducted in the field and included at least one type of natural wetland that was classified as NCW, NRW, or NPL. (b) All converted land-use types were directly transformed from the natural wetlands, and land-use types that had undergone multiple land-use transitions, such as restored wetlands, were excluded. (c) The selected experiments were conducted at the same temporal and spatial scales in both the natural wetlands and converted land-use types to avoid short-term noise, and the measurement period of the experiment exceeded one growing season. (d) The selected studies provided data for at least one type of GHG flux. In total, 90 papers with 209 study sites were used for the meta-analysis in this study (depicted in Figure 1, listed in Tables S1–S3), including 66 NCW sites, 65 NRW sites, and 78 NPL sites, all with comparison pairs of LULCC types. A total of five converted land-use

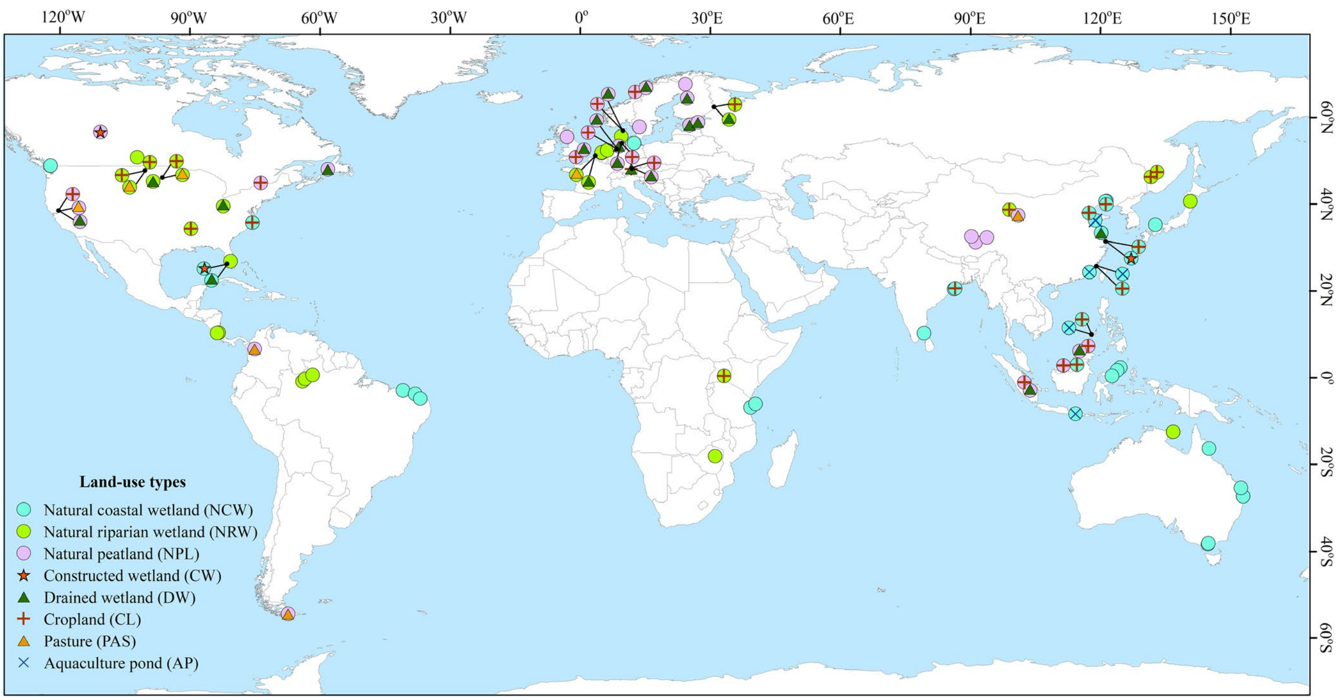

FIGURE 1 Location of the study sites. Natural coastal wetlands (NCWs) include tidal salt marshes, mangroves, and tidal freshwater marshes; natural riparian wetlands (NRWs) include undisturbed freshwater marshes and swamps in the riparian zones of rivers or shallow lakes; and natural peatlands (NPLs) include undisturbed fens, bogs, and swamps. The symbols of LULCC types of constructed wetland (CW), cropland (CL), aquaculture pond (AP), drained wetland (DW), and pastures (PAS) were overlapped on the symbols of natural wetlands (details in Tables S1–S3)

types were identified and investigated from the land-use transitions that occurred in the natural wetlands, including the CWs, CLs, and APs from NCWs, and the DWs, CLs, and PASs from NRWs and NPLs (Figure 1). The fluxes of CO_2, CH_4, and N_2O in all of the collected studies were measured using the eddy covariance technique or the static chamber technique, which are two commonly used observation methods for GHG emissions. The mean values and standard deviations/errors of the chosen variables were directly provided or were extracted directly from the text, tables, or digitized graphs using the program GetData Graph Digitizer 2.25 (http://getdata-graph-digitizer.com/) when necessary.

The compiled database (Tables S1–S3) comprised dominant vegetation, location, GHG fluxes, and environmental variables. The GHG flux data included gross primary productivity (GPP), ER, NEE, methane fluxes (FCH_4), and nitrous oxide fluxes (FN_2O). The environment factors (EFs) included mean annual temperature (MAT), mean annual precipitation (MAP), AGB, mean soil temperature (MST), soil water (SW) content, WT, bulk density (BD), soil pH (pH), SOC, Sal, soil total carbon, soil total nitrogen (TN), and soil NO_3^- and soil NH_4^+. The longitude, latitude, MAT, and MAP data were directly obtained from the published papers; if this information was not provided, we checked Google Earth using the location (longitude and latitude) of the study sites at http://www.worldclim.org/. Because the GHG flux and EF data were incomplete for some studies, the amount of data varied with the land-use types.

In this study, positive NEE values indicated CO_2 emissions to the atmosphere, and negative values indicated CO_2 uptake from the atmosphere. The studies with measurements of CO_2 fluxes by the eddy covariance method usually provided the data of GPP, NEE, and ER. Some studies that obtained measurements of CO_2 fluxes using the static opaque chamber-based technique for enclosed plants and soil can be considered ER, and CO_2 fluxes measured using static transparent chambers can be considered NEE. CH_4 fluxes were measured with both the eddy covariance method and the static opaque chamber-based technique, and all measurements of N_2O fluxes were conducted with the chamber-based technique.

According to IPCC (2013), the radiative forcing constants of CH_4 and N_2O are 34 and 298 times that of CO_2 at the 100 year time horizon, respectively. The fractions 44/12, 16/12, and 44/28 were used to convert the mass of carbon for CO_2 and CH_4 and nitrogen for the N_2O to CO_2 equivalent, respectively. To estimate the greenhouse effect of GHG emissions in the natural wetlands and different land-use types, the value of the comprehensive GWP (t CO_2-eq ha^{-1} year^{-1}) from CO_2, CH_4, and N_2O fluxes (CO_2, CH_4: g C m^{-2} year^{-1}; N_2O: g N m^{-2} year^{-1}) was calculated with the following equation:

$$\mathrm{GWP}=100\times\left(\frac{44}{12}\mathrm{NEE}+34\times\frac{16}{12}F\mathrm{CH_4}+298\times\frac{44}{28}F\mathrm{N_2O}\right). \quad (1)$$

The change rates of GHG (ΔGHGs) and environmental factors (ΔEFs) can reflect the synchronous responses of GHG emissions and environmental variables to LULCC. The relationship between ΔGHGs and ΔEFs can be further utilized to test the role of changed EFs in regulating the variability in GHG emissions to reveal the key EFs affecting greenhouse effects during land-use conversion. The ΔGHGs and ΔEFs were calculated by the following equations:

$$\Delta\mathrm{GHGs}=\frac{\left(\mathrm{GHG_{LULCC}}-\mathrm{GHG_N}\right)}{\left(\mathrm{GHG_N}\right)}, \quad (2)$$

$$\Delta\mathrm{EFs}=\frac{\left(\mathrm{EF_{LULCC}}-\mathrm{EF_N}\right)}{\left(\mathrm{EF_N}\right)}, \quad (3)$$

where GHG_N is the GHG flux in natural wetlands, GHG_{LULCC} is the GHG flux after land-use conversions, EF_N is the unit value of the EFs in natural wetlands, and EF_{LULCC} is the unit value of EFs after land-use conversions.

2.2 | Statistical analysis

First, all GHG fluxes from the 209 sites were calculated as the means and standard errors. The differences in GHG fluxes (GPP, NEE, ER, FCH_4, and FN_2O) and EFs (AGB, SW, WT, BD, pH, SOC, Sal, TN, NO_3^-, and NH_4^+) between the natural wetlands and the converted land-use types were tested with one-way analysis of variance (ANOVA).

Second, the response ratio (RR) was used to evaluate the responses of GHG fluxes to LULCC following Hedges, Gurevitch, and Curtis (1999) and Luo, Hui, and Zhang (2006). A total of 178 sites with data of the treatment (LULCC types) and control (natural wetlands) groups in the same study area within the same measurement time or with similar observation years (less than 3 years) were used to calculate the RR, which is defined as the natural logarithm of the ratio of the GHG mean values in the treatment groups to that in the control groups (Equation 4):

$$\mathrm{RR}=\ln\left(\frac{\overline{X_t}}{\overline{X_c}}\right)=\ln\left(\overline{X_t}\right)-\ln\left(\overline{X_c}\right), \quad (4)$$

where $\overline{X_t}$ and $\overline{X_c}$ are the means of the concerned variable in the treatment and control groups, respectively. The variance (v) was estimated with the following equation:

$$v=\frac{S_t^2}{n_t\overline{X_t}^2}+\frac{S_c^2}{n_c\overline{X_c}^2}, \quad (5)$$

where n_t and n_c are the sample sizes of the treatment and control groups, respectively, and S_t and S_c are the standard deviations in the treatment and control groups, respectively. The weighting function was calculated based on the reciprocal of the variance (Equation 6) in individual RRs (Hedges et al., 1999; Luo et al., 2006).

$$w_{ij}=\frac{1}{v}. \quad (6)$$

The weighted RR (RR_{++}) was calculated from the individual RR_{ij} (i = 1, 2,..., m; j = 1, 2,..., k) pairwise comparison between the treatment and control groups (Equation 7), where m is the number of groups and k is the number of comparisons in the ith group:

$$RR_{++} = \frac{\sum_{i=1}^{m} \sum_{j=1}^{ki} w_{ij} RR_{ij}}{\sum_{i=1}^{m} \sum_{j=1}^{ki} w_{ij}}. \tag{7}$$

The standard error of RR_{++} was estimated as follows:

$$S\left(RR_{++}\right) = \sqrt{\frac{1}{\sum_{i=1}^{m} \sum_{j=1}^{ki} w_{ij}}}. \tag{8}$$

The 95% confidence interval (95% CI) was $RR_{++} \pm 1.96\ S(RR_{++})$. If the 95% CI did not overlap with zero, the LULCC response was considered significant.

With the database of all 209 sites, redundancy analysis (RDA) was used to test the relationship between ΔGHGs and ΔEFs. Furthermore, a two-layer neural network model of multilayer perceptron (MLP) by artificial neural networks (ANNs) was performed to evaluate the relative influence (referring to the fraction of variation explained) of each ΔEF to target each ΔGHG. The network was set two hidden layers with three to five nodes, and the fraction of the samples into the training, validation, and testing was 50%, 30%, and 20%, respectively. MLP was based on Garson's (1991) algorithm which uses partitioning connection weights of the neural network to determine the explanatory variable importance of the ANN. Min–max normalization was performed to normalize the values of GHGs, ΔGHGs, EF, and ΔEF into [0, 1] before the RDA and ANN analyses were conducted:

$$y = \frac{(x - \min)}{(\max - \min)}, \tag{9}$$

where y is the normalized data and min and max are the minimum and maximum values of the variables, respectively.

ANOVA was performed using the SPSS statistics 23.0 package (SPSS Inc.). RDA was performed using the CANOCO 4.5 package (Microcomputer Power). The ANN was performed using MLP module in SPSS statistics 23.0 package. The best ANN model was selected based on the lowest relative error value in the holdout group, and the model summary of selected model was shown in Table S4. In all statistical tests, a significance level of $p < .05$ was used.

3 | RESULTS

3.1 | Variability in GHG fluxes

Across all study sites, the natural wetlands showed a capacity for CO_2 sequestration, with NEE values of −208.1, −779.9, and −135.0 g C m^{-2} $year^{-1}$ on average for NCWs, NRWs, and NPLs, respectively, which acted as emission sources of CH_4 and N_2O (positive FCH_4 and FN_2O values, Figure 2).

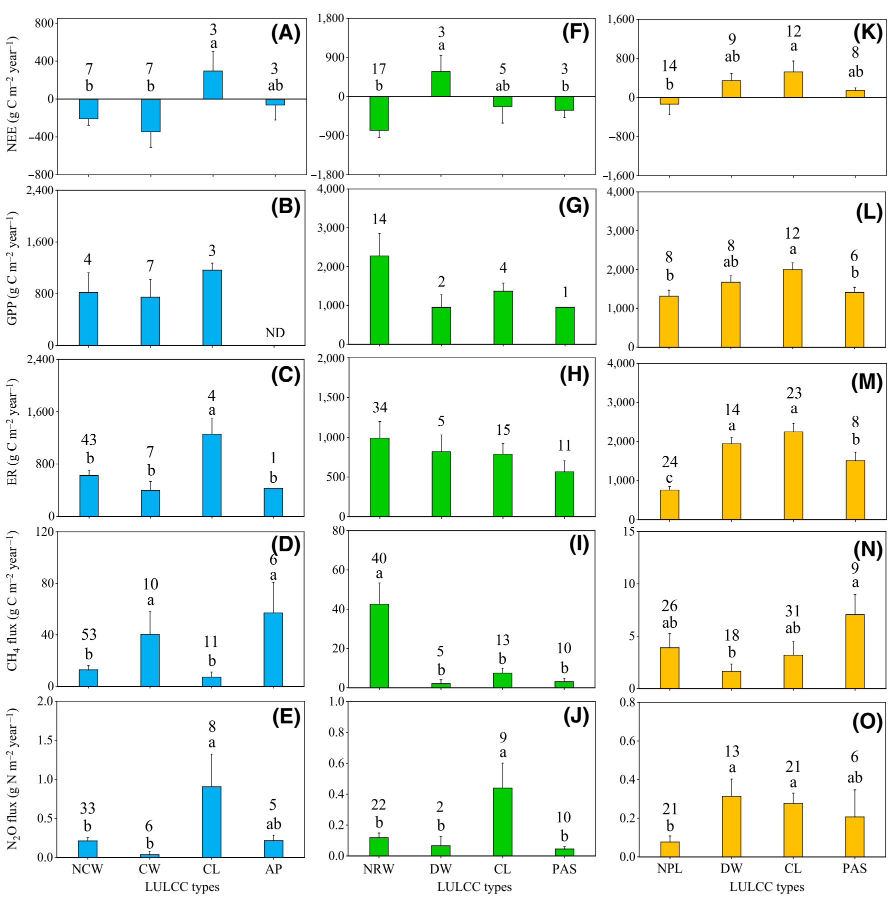

FIGURE 2 Changes in greenhouse gas fluxes in the natural coastal wetlands (NCWs) and the associated LULCC types (A–E), the natural riparian wetlands (NRWs) and the associated LULCC types (F–J), and the natural peatlands (NPLs) and the associated LULCC types (K–O). The sample size for each variable is shown next to the bar, and different lowercase letters denote significance at $p < .05$. ND, no data [Colour figure can be viewed at wileyonlinelibrary.com]

When NCWs were converted to the LULCC types of CL and AP, the capacity of CO_2 sequestration decreased, even showing net CO_2 emissions (positive NEE values) for CLs (Figure 2A) due to a greater increase in ER (101.5% on average) relative to slight changes (42.3% on average) in GPP. Compared to that in NCWs, the net CO_2 uptake in CWs increased. The FCH_4 values in CWs and APs were significantly higher (p < .05), by 216.7% and 346.9% on average, respectively, than that in NCWs, and the changes in FCH_4 in CLs were insignificant (Figure 2D). In contrast, the FN_2O in CLs was significantly higher (p < .05), by on average 328.5%, relative to that in NCWs, and the changes in FN_2O in CWs and APs were insignificant (Figure 2E).

The LULCC types of CL and PAS decreased the CO_2 sequestration capacity by 70.5% and 59.5% on average compared to the value in NRWs, respectively, and the DW conversion turned the NEE into positive values (net CO_2 emissions; Figure 2F). The decreases in NEE under LULCC were attributed to the greater extent of decreases in GPP than the decline in ER. All LULCC types significantly decreased (p < .05) the FCH_4 by 82.5%–95.0% on average compared to that in NRWs (Figure 2I). The DW and PAS types did not significantly change the FN_2O, although the value significantly increased by an average of 269.1% in CLs relative to that in NRWs (Figure 2J).

When NPLs were converted to the LULCC types of DW, CL, and PAS, the net CO_2 sequestration sink turned into a net CO_2 emission source, and the NEE in CLs significantly decreased (p < .05) by 486.6% (Figure 2K). Although the GPP increased in the DWs, CLs, and PASs compared to that in NPLs, the degrees of increases in ER (by 155.1%, 195.1%, and 98.1% on average, respectively) were significantly higher (p < .05) than that in GPP (by 27.4%, 51.9%, and 7.3% on average, respectively). FCH_4 slightly decreased in DWs and CLs, but significantly increased (p < .05) by an average of 81.1%, in PASs compared to that in NPLs (Figure 2N). The FN_2O values in DWs and CLs were significantly higher (p < .05), by 308.1% and 260.3% on average, than that in NPLs, respectively, with unremarkable increases in the FN_2O in PASs (Figure 2O).

3.2 | Responses of GHG fluxes to LULCC

Based on the analysis of weighted RRs for the strict pairwise sites, the LULCC types across all natural wetlands (except for DWs relative to NPLs) decreased NEE by 78.0% on average (Figure 3a), with significant (p < .05) effects of CWs relative to NCWs, DWs, and PASs relative to NRWs, and PASs relative to NPLs. The dataset referring to the effect of LULCC on GPP was unavailable for NCWs. Regarding NRWs and NPLs, the LULCC types increased the GPP by an average of 50.0%, with a higher increase of 63.2%, in ER (Figure 3b,c). All LULCC types (except for APs relative to NCWs) decreased the FCH_4 by 51.1% on average across the natural wetlands, with significant (p < .05) effects of DWs and CLs relative to NRWs (Figure 3d). The LULCC types (except for PASs relative to NRWs)

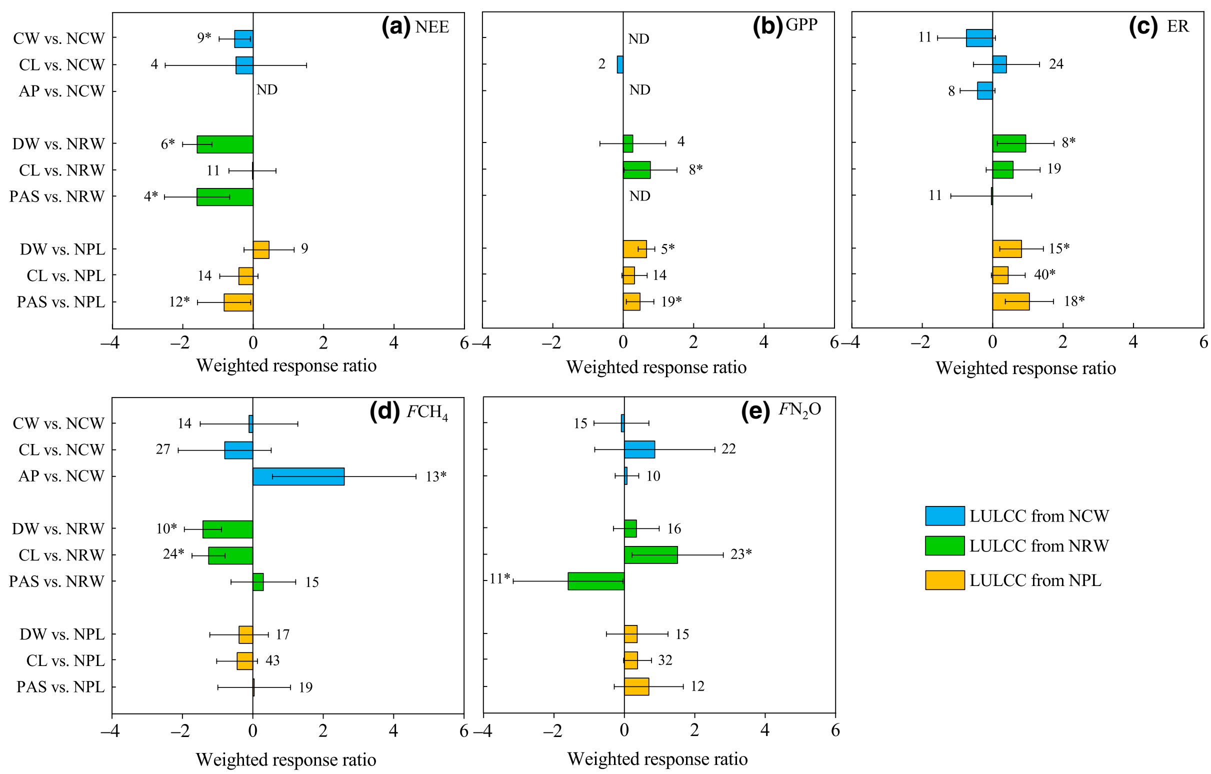

FIGURE 3 Weighted response ratios (RR_{++}) of (a) net ecosystem CO_2 exchange (NEE), (b) gross primary productivity (GPP), (c) ecosystem respiration (ER), (d) methane fluxes (FCH_4), and (e) nitrous oxide fluxes (FN_2O) to the land-use/land-cover change (LULCC) types from natural wetlands. Bars represent the RR_{++} values and 95% CIs. If the 95% CI value of RR_{++} did not overlap with zero, the LULCC response was considered significant. Vertical lines were drawn at RR = 0. The sample size for each variable is shown next to the bar, and asterisks (*) denote significance at p < .05. ND, no data [Colour figure can be viewed at wileyonlinelibrary.com]

stimulated FN_2O by an average of 51.7% across the natural wetlands, with a significant ($p < .05$) effect of CLs relative to NRWs (Figure 3e).

3.3 | Changes in GWP and GHG emissions due to LULCC

The comprehensive GWPs from CO_2, CH_4, and N_2O fluxes were −0.9, −8.7, and −2.9 t CO_2-eq ha^{-1} $year^{-1}$ on average for NCWs, NRWs, and NPLs (Table 1), respectively, exhibiting the capacity to mitigate greenhouse effects. However, the LULCC types of CW, CL, and AP increased the GWPs by 7.8, 22.3, and 29.5 times on average that in NCWs, respectively, turning to a strong greenhouse effect. The LULCC types of DW and CL enhanced the GWPs, and the value remained negative and was lower by 13.85% on average in PASs than in NRWs. Compared to NPLs, all LULCC types enhanced the GWPs by 4.0–8.7 times on average (Table 1), increasing the greenhouse effect.

According to the annual area loss rate of natural wetlands through LULCC, the annual GHG emissions from LULCC were at least 0.96 ± 0.22 Gt CO_2-eq, which is comparable to the result of LULCC in forests in the Amazonia region and is approximately 8.0–9.6% of annual global GHG emissions (Table 2).

3.4 | Variability in the EFs

Relative to NCWs, none of the LULCC types significantly changed the SW, soil NO_3^-, NH_4^+, or AGB (Figure 4A; Figure S2). The LULCC type of CW significantly decreased ($p < .05$) the SOC and Sal compared to the levels in NCWs but had no appreciable effect on the WT. CLs significantly decreased ($p < .05$) the pH, Sal, and WT while significantly increased ($p < .05$) the BD compared to the levels in NCWs. APs significantly decreased ($p < .05$) the SOC but significantly increased ($p < .05$) the BD and WT compared to those in NCW (Figure 4A; Figure S2).

Relative to NRWs, all LULCC types significantly decreased ($p < .05$) the SW, WT, SOC, and soil NH_4^+ (Figure 4B), except for the effect of DWs on SOC (Figure S2). The DWs did not significantly change the AGB, pH, and soil NO_3^- and TN but significantly decreased ($p < .05$) the BD compared to the levels in NRWs. CLs had no appreciable effect on the AGB, pH, and BD and TN but significantly increased ($p < .05$) the soil NO_3^- compared to the levels in NRWs (Figure 4B; Figure S2).

Compared to NPLs, none of the LULCC types significantly changed the SW, TN, and soil NO_3^- or NH_4^+ (Figure 4C; Figure S2). The LULCC type of DW only significantly decreased ($p < .05$) the WT. CLs significantly increased ($p < .05$) the AGB but decreased the pH, WT, and SOC compared to the levels in NPLs. PASs significantly decreased ($p < .05$) the WT and SOC but significantly increased ($p < .05$) the BD compared to those in NPLs (Figure 4C; Figure S2).

3.5 | Sensitivity of ΔGHG fluxes to ΔEFs

With RDA, the changes in EFs (ΔEFs) presented in the ordination explained 71.0%, 56.1%, and 40.6% of the variability in ΔGHGs

TABLE 1 Changes in comprehensive GWP from the NCWs, NRWs, and NPLs to the associated LULCC types

Land-use type	GWP (t CO_2-eq ha^{-1} $year^{-1}$)			Comprehensive GWP	Rate of change (%)[a]
	CO_2	CH_4	N_2O		
LULCC from NCWs					
NCW	−7.63 ± 2.56	5.78 ± 3.32	0.99 ± 0.20	−0.86 ± 3.92	—
CW	−12.66 ± 6.05	18.31 ± 8.15	0.17 ± 0.08	5.82 ± 8.98	+776.01
CL	10.85 ± 7.52	3.23 ± 1.79	4.24 ± 1.95	18.32 ± 2.39	+2,226.71
AP	−2.31 ± 2.13	25.84 ± 10.82	1.01 ± 0.31	24.54 ± 8.01	+2,948.80
LULCC from NRWs					
NRW	−28.60 ± 5.93	19.30 ± 4.88	0.56 ± 0.14	−8.74 ± 13.94	—
DW	21.23 ± 13.63	0.97 ± 0.88	0.31 ± 0.29	22.51 ± 6.87	+357.61
CL	−8.45 ± 13.92	3.37 ± 0.88	2.06 ± 0.76	−3.03 ± 3.74	+65.38
PAS	−11.57 ± 6.27	1.40 ± 1.16	0.22 ± 0.08	−9.95 ± 4.14	−13.85
LULCC from NPLs					
NPL	−4.98 ± 7.99	1.77 ± 0.61	0.36 ± 0.15	−2.85 ± 2.06	—
DW	12.74 ± 5.40	0.75 ± 0.70	1.47 ± 0.42	14.95 ± 3.88	+641.453
CL	19.26 ± 8.15	1.44 ± 0.61	1.29 ± 0.25	22.00 ± 5.96	+870.82
PAS	5.26 ± 2.01	3.20 ± 0.88	0.97 ± 0.66	9.43 ± 1.24	+395.35

Abbreviations: AP, aquaculture pond; CL, cropland; CW, constructed wetland; DW, drained wetland; GWP, global warming potential; LULCC, land-use/land-cover change; NCW, natural coastal wetlands; NPL, natural peatlands; NRW, natural riparian wetlands; PAS, pasture.
[a]Relative changes in GWP of LUCCC types against NCW, NRW, and NPL.

TABLE 2 Greenhouse gas (GHG) emissions due to land-use/land-cover change (LULCC; or degradation) from natural ecosystems

Emission source	Region	Years of observation	Annual area loss rate (%)	GHGs types	GHG emission ($Gt\ CO_2$-eq/year)	Source
LULCC in forests and wetlands	Global	1989–1998	1.84	CO_2, CH_4	5.86 ± 2.93	IPCC (2000)
LULCC in forests and wetlands	Global	2000–2010	—	CO_2, CH_4, N_2O	10.0–12.0	IPCC (2014)
LULCC in forests	Tropical area	1990–1997	0.43	CO_2, CO, CH_4	2.34 ± 0.77	Achard et al. (2002)
LULCC in forests	Amazonia	1981–1990	0.81	CO_2, CO, CH_4, N_2O	0.98–1.02	Fearnside (2000)
LULCC in peatlands	Southeast Asia	2000–2006	1.30	CO_2	0.36–0.86	Hooijer et al. (2010)
LULCC in coastal ecosystems	Global	1997–2005	1.50 (tidal marsh) 1.90 (mangroves) 1.50 (seagrass)	CO_2, CH_4	0.15–1.02	Pendleton et al. (2012)
LULCC in wetlands	Global	1990–2016	0.98 (coastal wetlands) 0.48 (inland wetlands)	CO_2, CH_4, N_2O	0.96 ± 0.22	This study

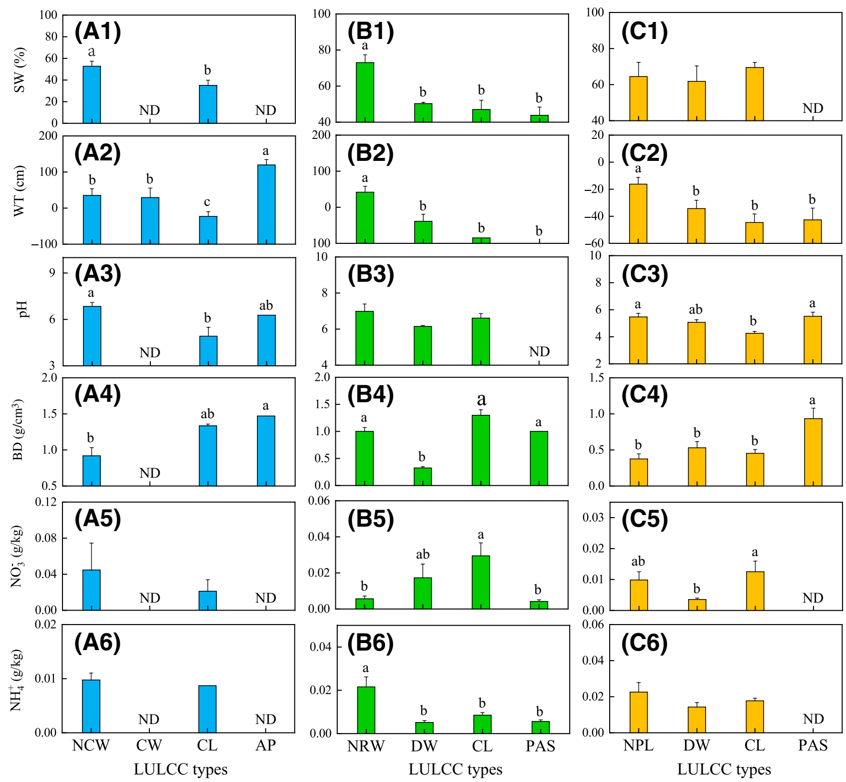

FIGURE 4 Changes in the environmental factors (EFs) in the natural wetlands (NCWs: A1–A6), riparian wetlands (NRWs: B1–B6), and peatlands (NPLs: C1–C6) and the associated land-use/land-cover change (LULCC) types. Different letters indicate significant differences (p < .05). ND, no data. The changes in AGB, SOC, TN, and Sal are presented in Figure S2 [Colour figure can be viewed at wileyonlinelibrary.com]

(Figure 5) regarding the LULCC types in NCWs, NRWs, and NPLs, respectively.

When NCWs were converted to the associated LULCC types, the ΔNEE was positively correlated with ΔAGB and ΔTN, and negatively correlated with ΔSW, ΔBD, ΔSal, and ΔSOC (Figure 5). The ΔFCH_4 was positively correlated with ΔWT and ΔNO_3^-. The ΔFN_2O was positively correlated with ΔNO_3^- and ΔWT, and negatively correlated with ΔSal and ΔpH. By ranking the relative influences of ΔEFs on ΔGHGs, the combination of ΔSW, ΔpH, ΔWT, and ΔSal explained 68.1% of the variability in ΔNEE (Figure 6a), and ΔWT and ΔNO_3^- were the dominant factors affecting the variation in ΔFCH_4, explaining 49.5% of the relative influence (Figure 6b). The combination of ΔSal, ΔSW, and ΔNO_3^- explained 59.7% of the variability in ΔFN_2O (Figure 6c).

Regarding NRWs and the associated LULCC types, the ΔNEE was positively correlated with ΔNO_3^-, ΔTN, and ΔSOC, but negatively

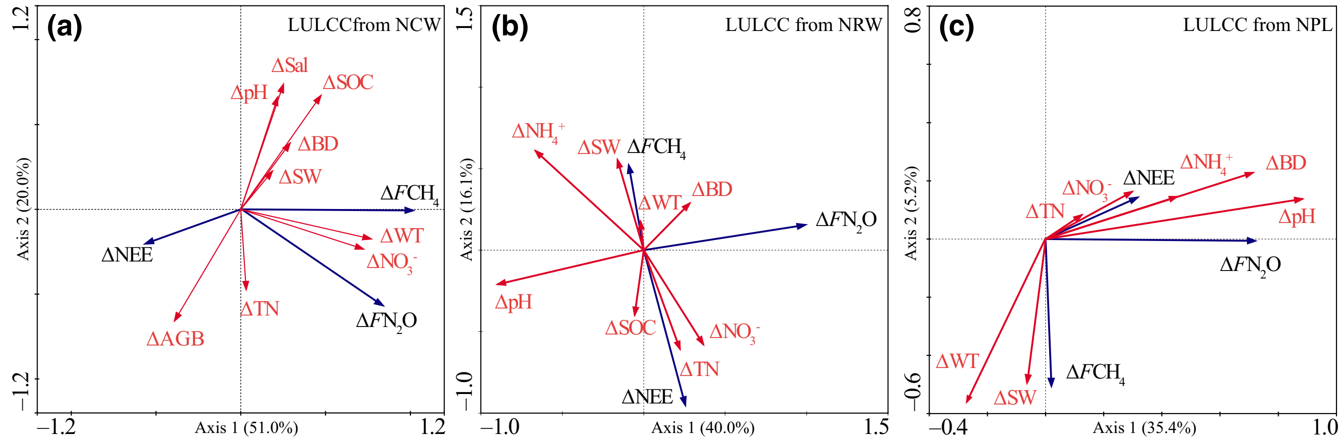

FIGURE 5 Redundancy analysis of the relationship between ΔGHG (greenhouse gas) fluxes and ΔEFs (environment factors) when the natural coastal wetlands (NCWs: a), riparian wetlands (NRWs: b), and peatlands (NPLs: c) were converted to the associated land-use/land-cover change (LULCC) types [Colour figure can be viewed at wileyonlinelibrary.com]

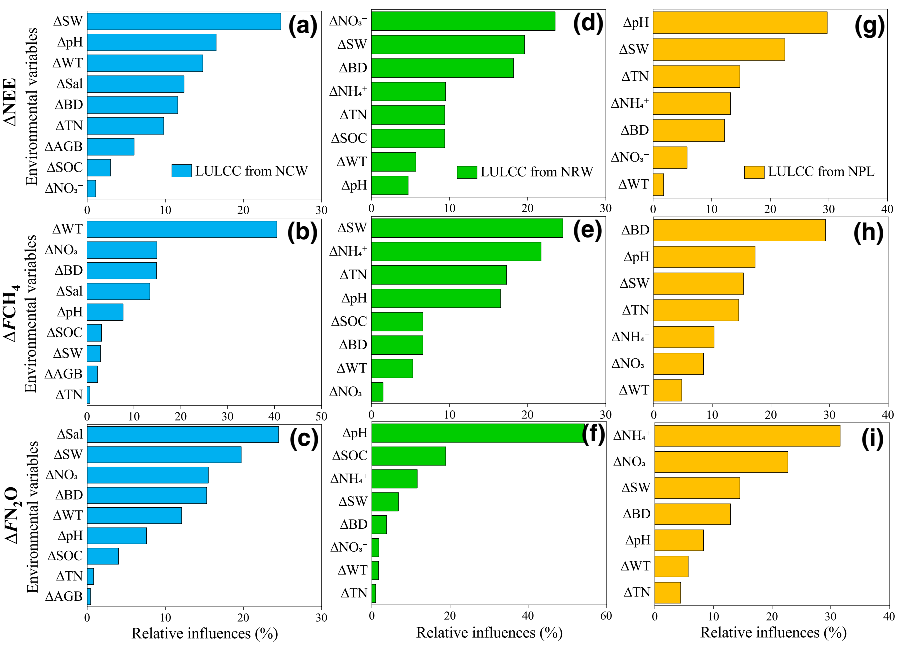

FIGURE 6 Relative influences of ΔEFs (environment factors) to ΔGHG (greenhouse gas) fluxes when the natural coastal wetlands (NCWs: a–c), riparian wetlands (NRWs: d–f), and peatlands (NPLs: g–i) were converted to the associated land-use/land-cover change (LULCC) types [Colour figure can be viewed at wileyonlinelibrary.com]

correlated with ΔSW, ΔWT, and $ΔNH_4^+$ (Figure 5). The $ΔFCH_4$ was positively correlated with ΔWT, ΔSW, and $ΔNH_4^+$, and negatively correlated with ΔTN, ΔSOC, and $ΔNO_3^-$. The $ΔFN_2O$ was positively correlated with $ΔNO_3^-$ and ΔBD, and negatively correlated with ΔpH. As shown in Figure 6d–f, the combination of $ΔNO_3^-$, ΔSW, and ΔBD explained 61.3% of the variability in ΔNEE, and the combination of ΔSW, $ΔNH_4^+$, and ΔTN explained 80.0% of the variability in $ΔFCH_4$. The ΔpH was the dominant factor affecting $ΔFN_2O$, with 54.4% of the relative influence, when NRWs were converted to the associated LULCC types.

When comparing NPLs and the associated LULCC types, the ΔNEE was positively correlated with $ΔNO_3^-$, $ΔNH_4^+$, and ΔBD but negatively correlated with ΔWT and ΔSW (Figure 5). The $ΔFCH_4$ was positively correlated with ΔWT and ΔSW, but negatively correlated with ΔTN and $ΔNO_3^-$. The $ΔFN_2O$ was positively correlated with ΔpH, ΔBD, and $ΔNH_4^+$ but negatively correlated with ΔWT. By ranking the relative influences of ΔEFs on ΔGHGs, ΔpH and ΔSW together contributed 52.2% of the relative influence to ΔNEE (Figure 6g), and the combination of ΔBD, ΔpH, and ΔSW explained 61.6% of the variability in $ΔFCH_4$ (Figure 6h). Together, $ΔNH_4^+$ and $ΔNO_3^-$ contributed 54.3% of the relative influence on $ΔFN_2O$ (Figure 6i).

4 | DISCUSSION

4.1 | GHG budget of LULCC from natural wetlands

This study confirmed that the natural wetlands act as GHG sinks or are neutral. Considering the global area of coastal wetlands (1.28×10^5 km^2; Murray et al., 2019), riparian wetlands (3.69×10^6 km^2; Lehner & Döll, 2004), and peatlands (4.23×10^6 km^2; Xu, Morris, Liu, & Holden, 2018), the annual GHG uptake by the three types of natural wetlands would be around 0.01, 3.23, and 1.21 Gt CO_2-eq/year, respectively, which are comparable to the results of Lu et al. (2017). However, both analytical methods of data (with total 209 sites or 178 strictly controlled sites) showed that LULCC would turn GHG sinks into sources. By combining CO_2, CH_4, and N_2O emissions, the comprehensive GWPs of various LULCC types from the natural coastal wetlands, riparian wetlands, and peatlands increased to 16.22 ± 5.50, 3.17 ± 9.87, and 15.46 ± 3.64 t CO_2-eq ha^{-1} year^{-1} on average, respectively.

At the global scale, it is generally believed that the loss of natural wetlands caused by LULCC is approximately 33%–57% (Davidson, 2014; Hu, Niu, Chen, Li, & Zhang, 2017). Based on the annual loss rates of different natural wetland types determined by Murray et al. (2019) and Davidson (2014), we estimated that LULCC (mainly from 1990 to 2018) would result in at least 0.96 ± 0.22 Gt CO_2-eq of GHGs being released into atmosphere per year, accounting for 8.0%–9.6% of the annual global GHG emissions estimated by the IPCC in 2014 (Table 2). This result indicated that LULCC might offset the GHG mitigation potential of the original natural wetlands and make the developed ecosystems anthropogenic GHG sources with enhanced greenhouse effects.

Previous studies indicated that the GHGs dynamics were sensitive to latitudinal variations of climate (e.g., temperature and precipitation; Inglett et al., 2012; Lu et al., 2017). However, the latitudinal distribution of annual GHG fluxes in the wetlands undergoing LULCC showed that there were no significant latitude patterns (Figure S1), indicating that land use change and human disturbance probably led to reduction of spatial heterogeneity of GHG emissions in relation to climate conditions.

4.2 | GHG emissions in coastal wetlands and the associated LULCC types

Tidal cycling in coastal zones can affect GHG production via various hydrological processes of flood inundation, saltwater inpouring, and wave action (Guo et al., 2009; Li et al., 2018). For instance, inflowing water at flood tides restricts gas diffusion from soil and dilutes the dissolved GHGs in porewater, resulting in low GHG emission levels (Yamamoto, Hirota, Suzuki, Zhang, & Mariko, 2011). Tides may depress plant-mediated GHG emissions when plant stems are submerged, and a portion of diffused CH_4 is oxidized by inflowing tidal water (De Angelis & Scranton, 1993; Tong et al., 2013). Moreover, various alternative electron acceptors (e.g., SO_4^{2-}) in saltwater inhibit the activity of methanogens through substrate competition (Chambers, Osborne, & Reddy, 2013; Neubauer, Franklin, & Berrier, 2013; Poffenbarger et al., 2011; Sun et al., 2013; Vizza, West, Jones, Hart, & Lamberti, 2017; Wilson, Mortazavi, & Kiene, 2015).

When the natural coastal wetlands are enclosed, the tidal influences by dykes (or seawalls) and the exchange of water and nutrient matter from offshore seas are blocked, causing decreases in soil Sal and the WT (Fernández et al., 2010; Zhong et al., 2016). The LULCC type of CW is generally identified as a seminatural ecosystem with little tidal fluctuation and a low Sal level, which might increase the GPP of plants and obstruct CO_2 diffusion from soil, thus resulting in a relatively higher NEE compared to that in the natural wetlands (Yamamoto et al., 2011). However, the FCH_4 in the CWs was significantly higher (~2.2 times) than that in the natural wetlands, mainly due to a reduction in inhibition due to Sal and the influence of tides on CH_4 emissions (Hu, Ren, et al., 2017).

Our results showed that the CLs generally increased the CO_2 and N_2O emissions while decreasing FCH_4. Drainage and agricultural management practices (e.g., tillage, fertilization, and irrigation) profoundly alter hydrological and nutrient conditions (Hirata et al., 2013; Verhoeven & Setter, 2009). A lowered WT due to enclosure changed the soil environment from anaerobic to aerobic, which stimulated existing enzyme activities to enhance soil mineralization and soil respiration (Freeman et al., 1996; Gleason et al., 2009; Huang et al., 2010; Salm et al., 2012; Yamamoto et al., 2011). Additionally, a lowered water table promoted CH_4-oxidizing bacteria but inhibited the activity of methanogens, resulting in a decrease in FCH_4 (Yrjälä et al., 2011). The intensive application of fertilizer in the CLs enhanced the concentrations of NH_4^+, NO_3^-, and organic matter in soil, which stimulated the activities of nitrifying and denitrifying bacteria and soil C conversion and the consequent N_2O and CO_2 emissions (Alm et al., 1999; Baggs, 2008; Klotz & Stein, 2008; Morse, Ardón, & Bernhardt, 2012).

Compared to the other LULCC types associated with the coastal wetlands, the APs had the highest FCH_4 and comprehensive GWP, which were 3.5 and 29.5 times that of the natural wetlands, respectively, indicating a hot spot of GHG emissions. The APs are constructed as artificial aquatic ecosystems with closure and long-term waterlogging conditions and the absence of vegetation. In all kinds of APs studied, GPP was remarkably low due to the harvest of macrophytes for the construction of open water, and respiration from aquatic animals increases the CO_2 emissions, leading to a reduction in NEE (Chen, Dong, Wang, Gao, & Tian, 2016; Kosten et al., 2010). Long-term waterlogging during the cultivation period will reduce oxygen penetration into sediments and decrease the redox potential of sediment, making the conditions favorable for anaerobic methanogen activities and the consequent CH_4 production (Yang et al., 2017). In addition, cultured aquatic animals consume oxygen on the surface of sediments, resulting in partial anaerobic conditions (Liu et al., 2016). Ebullition is the main CH_4 emission pathway of non-vegetated shallow ponds, such as APs (Delsontro, Boutet, St-Pierre, Giorgio, & Prairie, 2016; Keller & Stallard, 1994). Compared to those in the natural wetlands, the sediments in APs, with a low Sal, anaerobic conditions, and the enrichment of organic matter (mainly animal excrement and forage) that is favorable for methanogens, can enhance the methane production rate and number of CH_4 bubbles (Yang, He, Huang, & Tong, 2015). As a result, CH_4 bubbles can quickly pass through sediments and the water column and be released into the atmosphere, avoiding CH_4 consumption by aerobic methane-oxidizing microorganisms (Keller & Stallard, 1994). Moreover, a frequent forage supply for aquatic animals will accumulate in the APs, probably providing abundant labile

C and N for algae growth and microbial activities (e.g., nitrifying and denitrifying bacteria) that subsequently stimulate the CO_2 and N_2O emissions (Chen et al., 2016; Yang et al., 2018).

4.3 | GHG emissions in riparian wetlands and the associated LULCC types

Of the natural wetland types, the riparian wetlands had the highest FCH_4, probably due to higher SW content (long-term inundation) and NH_4^+ levels available adjacent to water bodies. Long-term inundation can provide stable anaerobic conditions and reduce soil oxygen concentrations, thus stimulating methanogen activities. Previous studies indicated that NH_4^+ can promote CH_4 production, mainly by competing with methane monooxygenase for O_2 to reduce CH_4 oxidation (Hu, Ren, et al., 2017; King & Schnell, 1998).

Drainage in the riparian wetlands alters the water regime and nutrient availability (Audet et al., 2013; Krauss et al., 2012), and previous studies have shown that drainage activities could increase the CO_2 emissions (Finocchiaro, Tangen, & Gleason, 2014; Hendriks, Huissteden, Dolman, & Molen, 2007; Musarika et al., 2017; Olsson et al., 2015). Taking DWs as an example, this LULCC type had higher CO_2 emissions, while lower CH_4 and N_2O emissions were mainly due to reductions in the WT, soil BD, and available NH_4^+. Drying environments most likely decrease plant growth and GPP, thereby greatly enhancing CO_2 emissions in the DWs (Berglund & Berglund, 2011; Musarika et al., 2017). However, a lower WT would decrease CH_4 production in soil, and restricted plants would block the CH_4 transport pathway via roots and stems bypassing the oxidized soil layer (Salm et al., 2012). In addition, the DWs had the lowest soil BD, which would contribute to a higher soil porosity and higher oxygen availability, in turn probably promoting the oxidation of CH_4, leading to a decrease in FCH_4 (Chen, Wang, Han, Wan, & Li, 2010; Yrjälä et al., 2011). Compared to the conditions in natural wetlands, the DWs had lower available NH_4^+ with aerobic conditions, which limit both nitrification and denitrification processes and further decrease the N_2O emissions.

The CL and PAS are the LULCC types for agricultural production that generally has higher plant production than that in the natural wetlands. Therefore, the CLs and PASs can act as CO_2 sinks, while the both LULCC types may also result in net C losses during the field management practices of harvesting and grazing (Frank & Dugas, 2001; Graham, Haynes, & Meyer, 2002; Knox et al., 2014; Saunders et al., 2012). The CLs significantly increased FN_2O while decreasing FCH_4 because the CLs had the highest available NO_3^- and low-level BD, which are associated with fertilization and tillage. NO_3^- is a substrate of denitrifying bacteria, and an adequate supply of NO_3^-, lowered BD, and improved anaerobic condition after drainage would enhance denitrification and the consequent N_2O emissions (Baggs, 2008; Klotz & Stein, 2008). However, some studies have reported that NO_3^- is negatively correlated with CH_4 production due to NO_2^- production by denitrification, which is toxic to the methanogens responsible for CH_4 production (Patra & Yu, 2014; Reay & Nedwell, 2004). In addition, drainage and fertilization could reduce soil pH in the CLs (also the DWs; Hadi et al., 2005), and soil pH had a negative relationship with FN_2O. As soil pH decreases, nitric oxide reductase is inhibited, resulting in increased N_2O accumulation (Denmead et al., 2010; Mørkved, Dörsch, & Bakken, 2007; Obia, Cornelissen, Mulder, & Dörsch, 2015; Petersen et al., 2012).

Relative to the conditions in NRWs, the PASs had the lowest comprehensive GWP due to increased NEE and a relatively lower FCH_4 and FN_2O. Schaufler et al. (2010) and Bateman and Baggs (2005) found that the soil moisture optima for denitrification and N_2O production ranged between 60% and 80% of the water-filled pore space. Similar to the DWs from the natural wetlands, the decreased SW content and available N in the PASs would enhance the soil oxygen availability and then inhibit denitrification and CH_4 production. However, as mentioned above, various and extensive field management practices in the agricultural lands probably turn C sinks into sources by stimulating the GHG emissions.

4.4 | GHG emissions in peatlands and the associated LULCC types

Water table of all LULCC types (DWs, CLs, and PASs) from the NPLs were lowered below the soil surface, and higher rates of peat soil oxidation caused by drainage activity might significantly increase ER in the LULCC types compared to the levels in peatlands. As a result, soil organic C was greatly reduced in the LULCC types, which indicated that LULCC resulted in soil C loss by releasing CO_2 to the atmosphere (Don, Schumacher, & Freibauer, 2011). The peatlands and its LULCC types (DWs, CLs, and PASs) had lower FCH_4 values, possibly due to the relatively dry conditions. When the WT was lowered to a level that was below the soil surface, the upper peat layer acted as an efficient oxidative methanotrophic barrier for diffusive CH_4 from the subsoil and decreased the CH_4 emissions (Kandel et al., 2018). However, the contents of soil total N in the LULCC types were higher than that in the peatlands due to drainage and fertilization, which improve N mineralization for nitrification and denitrification, further resulting in higher FN_2O levels (Martikainen, Nykänen, Crill, & Silvola, 1993; Pärn et al., 2018). Yang et al. (2013) also suggested that a lowered WT could increase N_2O production in either nitrification or denitrification by increasing the volume of modestly aerated soil.

Similar to the CLs described from the NRWs, the reduced soil pH could be attributed to drainage and fertilization from the CLs to the peatlands (Denmead et al., 2010; Hadi et al., 2005; Petersen et al., 2012). Consequently, nitric oxide reductase would also be inhibited for transforming N_2O to N_2, resulting in increased N_2O accumulation (Mørkved et al., 2007; Obia et al., 2015).

FCH_4 and FN_2O in the PASs were higher than those in the peatlands. Intensive trampling by livestock will compact the surface soil and increase the BD, which will result in less water infiltration and partial anaerobiosis, which are favorable for anaerobic methanogenic processes (MacDonald et al., 1996; Schaufler et al., 2010; Tangen et al., 2015). However, root development will be restricted by compressed soil, probably leading to decreased soil organic C stocks from plant litters (Lugo & Brown, 1993; Rasse, Rumpel, & Dignac, 2005). At same time, plant growth and the CO_2 sequestration capacity will be limited. When the peatlands were converted to the PASs, the yield of fresh dung (ammonia fertilizer) by livestock in intensively grazed PASs increased the available NH_4^+, which significantly promoted the CH_4 and N_2O emissions (Gregorich, Rochette, VandenBygaart, & Angers, 2005; Lin et al., 2009; Sherlock et al., 2002).

4.5 | Implications for GHG modeling and manipulative experiments

This study provides an understanding of variations in GHG emissions from the natural wetlands to the worldwide popular LULCC types, offering hints for the development of land surface GHG models and the improvement of experimental manipulations. Current land surface models, for instance, CSIRO's Atmosphere Biosphere Land Exchange (CABLE) model, mainly identify precipitation and temperature as driving parameters when modeling the C cycle (Xia, Luo, Wang, & Hararuk, 2013), which might simplify the effects of environmental change due to LULCC. Regarding wetland ecosystems, GHG emissions are regulated by multiple abiotic and biotic factors, such as wetland hydrology, soil properties, and plant physiological processes. This study revealed a wide range of changes in environmental factors during LULCC conversion, which were critical for variations in GHG emissions. Furthermore, the responses of GHG emissions in the different types of natural wetlands to LULCC were inconsistent due to site-specific hydrological and soil characteristics, saline environments, and vegetation types (as discussed above). For instance, LULCC in the coastal wetlands will stimulate a large amount of C to be released into the atmosphere with a larger GWP value mainly due to the inhibition of tides and decrease in Sal. Waterlogging and relatively higher oxygen consumption due to high stocking densities in the APs led to the highest CH_4 emissions and highest GWP. The development and management intensity of LULCC types associated with the riparian wetlands and peatlands will also result in the notable promotion of GHG emissions in terms of higher CO_2 and N_2O emissions under fertilization and harvest practices. This information might be helpful for modifying GHG exchange models by using process-based mechanisms to predict the effects of LULCC.

In addition, the sites used in this study were mainly located in regions with large populations that are currently hot spots for LULCC, but data from Africa, South America, and Siberia were often lacking. The main natural wetland types in Africa and South America are salt marshes and mangroves, which have high rates of C sequestration (Pendleton et al., 2012). However, the natural wetlands in those regions also experience degradation and loss that are attributed to LULCC, showing loss ratios of 16% and 32% in Africa and South America, respectively (Davidson, 2014; Hu, Niu, et al., 2017). Peatlands cover over 8% of Russia's territory with an extensive area of 1.18×10^6 km^2 (Xu et al., 2018), whereas the effects of land use on Russian peatlands is negligible compared to that in more densely populated countries (Robarts, Zhulidov, & Pavlov, 2013). The investigation of GHG emissions in relation to LULCC from natural wetlands in those critical regions is urgently needed for global-scale estimations of GHG variation. Furthermore, in global LULCC cases, simultaneous conversion for multiple land-use types frequently occurs rather than a single land-use type (Hooijer et al., 2010; Knox et al., 2014). Therefore, future manipulative experiments and GHG models should consider the simultaneous observations of multiple LULCC types and field management activities for the evaluation of integrated effects on GHG emissions.

5 | SUMMARY

The natural wetlands can be identified as the net CO_2 sink and the net sources of CH_4 and N_2O, exhibiting the capacity of mitigation of greenhouse effect due to their negative values of GWP. However, we found that all LULCC types decreased the net CO_2 uptake mainly due to relatively slight changes in gross primary production and higher increase in ER. When the natural coastal wetlands were enclosed for LULCC types, the CWs and APs significantly increased the CH_4 emission, and the CLs significantly increased the N_2O emission. All LULCC types associated with the NRWs significantly decreased the CH_4 emissions, while the CLs significantly increased the N_2O emission. Compared to the NPLs, the PASs significantly increased the CH_4 emission, and the DWs and CLs significantly increased the N_2O emission. Compared to the natural wetlands, the associated LULCC types (except for PASs from riparian wetlands) resulted in the remarkable higher GWPs, enhancing the greenhouse effect. The variability in GHG fluxes with LULCC was mainly sensitive to changes in SW content, WT, Sal (for coastal wetlands), soil N content, soil pH, and BD. This study highlights the significant role of LULCC in stimulating the GHGs emission and suggests a possibility of shifting the mitigation of global warming by the natural wetlands to a notable greenhouse effect due to LULCC, which deserves more attention for land-use policy and land surface GHG models at global scale.

ACKNOWLEDGEMENTS

This paper is a product of the project "Coping with deltas in transition" within the Programme of Strategic Scientific Alliances between China and the Netherlands, financed by the Chinese Ministry of Science and Technology (2016YFE0133700) and Royal Netherlands Academy of Arts and Sciences. This work was also supported by the National Natural Science Foundation of China (41871088 and 41571083), the National Key R&D Program of China (2017YFC0506001), and the Key Project of Shanghai Science & Technology Committee (19DZ2210200).

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

How to cite this article: Tan L, Ge Z, Zhou X, Li S, Li X, Tang J. Conversion of coastal wetlands, riparian wetlands, and peatlands increases greenhouse gas emissions: A global meta-analysis. *Glob Change Biol*. 2020;26:1638–1653. https://doi.org/10.1111/gcb.14933