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Annual carbon sequestration and loss rates under altered hydrology and fire regimes in southeastern USA pocosin peatlands

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Abstract

Peatlands drained for agriculture or forestry are susceptible to the rapid release of greenhouse gases (GHGs) through enhanced microbial decomposition and increased frequency of deep peat fires. We present evidence that rewetting drained subtropical wooded peatlands (STWPs) along the southeastern USA coast, primarily pocosin bogs, could prevent significant carbon (C) losses. To quantify GHG emissions and storage from drained and rewetted pocosin we used eddy covariance techniques, the first such estimates that have been applied to this major bog type, on a private drained (PD) site supplemented by static chamber measurements at PD and Pocosin Lakes National Wildlife Refuge. Net ecosystem exchange measurements showed that the loss was 21.2 Mg CO_2 ha^{-1} year^{-1} (1 Mg = 10^6 g) in the drained pocosin. Under a rewetted scenario, where the annual mean water table depth (WTD) decreased from 60 to 30 cm, the C loss was projected to fall to 2 Mg CO_2 ha^{-1} year^{-1}, a 94% reduction. If the WTD was 20 cm, the peatlands became a net carbon sink (−3.3 Mg CO_2 ha^{-1} year^{-1}). Hence, net C reductions could reach 24.5 Mg CO_2 ha^{-1} year^{-1}, and when scaled up to the 4000 ha PD site nearly 100,000 Mg CO_2 year^{-1} of creditable C could be amassed. We conservatively estimate among the 0.75 million ha of southeastern STWPs, between 450 and 770 km^2 could be rewet, reducing annual GHG emissions by 0.96–1.6 Tg (1 Tg = 10^{12} g) of CO_2, through suppressed microbial decomposition and 1.7–2.8 Tg via fire prevention, respectively. Despite covering <0.01% of US land area, rewetting drained pocosin can potentially provide 2.4% of the annual CO_2 nationwide reduction target of 0.18 Pg (1 Pg = 10^{15} g). Suggesting pocosin restoration can contribute disproportionately to the US goal of achieving net-zero emission by 2050.

KEYWORDS
carbon dioxide, carbon sequestration, eddy covariance, greenhouse gas, methane, net ecosystem exchange, peatlands, pocosin

1 | INTRODUCTION

1.1 | Global carbon and peatlands

Peatlands are wetlands with organic soils that comprise about 3% of land cover globally but contains 600–700 Pg (1 Pg = 10^{15} g) of carbon (C), which exceeds total vegetation carbon stocks on Earth with an amount close to atmospheric CO_2–C (Dargie et al., 2017; Page et al., 2011; Xu et al., 2018; Yu et al., 2010). Hence, both C stores in peat and C pathways in peatlands, particularly their sequestration/release processes and responses to drought/drainage and fire, can drastically affect future greenhouse gas (GHG) releases to the atmosphere. For example, drained peatlands emit ~2 Pg CO_2 year^{-1} through microbial respiration and peat fire, translating to ~5% of all anthropogenic GHG emissions (Joosten et al., 2016). Furthermore, deep peat fires occurring mostly on drained peatlands may account for as much as 15% of annual global GHG emissions in some years (Rein, 2015). A more recent study implies that GHG emissions from drained peatlands by 2100 may comprise 12%–41% of the remaining GHG emission budget; thus, rewetting of drained peatlands is a key component to keeping global warming below +1.5 to +2°C (Leifeld et al., 2019).

A recent process-based land surface model, including both natural and cultivated peatlands, estimated northern hemisphere emissions alone from all land use activities was ~60 Pg C during 1750–2018, which surpassed peatland-excluded estimates of historical CO_2 emissions by 67% (Qiu et al., 2021). This finding suggests a serious underestimate of CO_2 emissions from cultivated northern peatlands and that terrestrial C loss estimates since 1750 were undervalued by 18%, a value which helps close the historical global C budget and demonstrates the global impact of draining peatlands. Unfortunately, development trends and C losses on subtropical wooded peatlands (STWPs) and tropical wooded peatlands, which store about 30% of peat globally (Farmer et al., 2011; Page et al., 2011), are almost non-existent in the literature, especially from Africa and South America. Thus, the effects of deforestation, drainage for agriculture, extraction of peat for fuel, and coastal development on reductions of C stocks and increased GHG emissions in many regions of the world are unknown (Cooper et al., 2020; Richardson et al., 2014; Silvestri et al., 2019).

1.2 | US subtropical coastal peatlands

The STWPs occur within a latitude band of 35° N–35° S and are differentiated from other peatlands by climate and C source, that is, wood-derived peat in STWPs as opposed to *Sphagnum*/*Carex* in northern peatlands (Andriesse, 1988; Joosten et al., 2016; Wang et al., 2015; Winton et al., 2017; Zinck, 2011). Millions of hectares of STWPs are found along the Atlantic coast in the USA from Virginia to Georgia. They have persisted through changing climate and fluctuating sea levels for the last 10,000 years. If they are not drained, these STWPs likely continue to accrete peat in warmer temperatures, even under increased droughts or light fires (Flanagan et al., 2020; Hodgkins et al., 2018; Richardson, 2012; Sharitz & Gibbons, 1982). Importantly, in the continental United States, 20% of the peat is stored in these bog areas known as pocosin (Fargione et al., 2018; Flanagan et al., 2020).

The southeastern evergreen shrub bogs, or pocosins, are a major STWP formation, which features a dense growth of evergreen broadleaf shrubs with scattered pond pine (Richardson, 2012). The typically thick (1–5 m) peat soils (Histosols) underlying pocosins act as chemical sponges over geologic time, locking up C and nutrients in vegetation (<5%), but mostly by storing elements in the accreting organic peat layer. Under normal saturated hydrologic conditions, decomposition in these organic soils is minimal due to a lack of oxygen, low pH, and high aromatics, allowing for a long-term rate of C accumulation of ~0.43 Mg C ha^{-1} year^{-1} over the past millennia (Hodgkins et al., 2018; Richardson et al., 2014). However, more than 100,000 ha of pocosins, mainly located in coastal North Carolina, Georgia and Virginia, drained for now-defunct peat mining operations, marginal farming, and/or forestry (Richardson, 1981, 1983, 2012) now lie in a fallow state. As a result, their C, and nutrient retention functions are diminished, and a large volume of C is continually released to the atmosphere as CO_2 and, to a much lesser degree, as dissolved organic carbon (DOC) in adjacent waters (Gregory et al., 1984; Richardson, 1983; Richardson et al., 2014; Wang et al., 2016).

Earlier small-scale studies have shown that hydrologic restoration, paired with the unique chemistries of pocosin peat soils, resulted in increased C sequestration and reduced GHG losses (Richardson, 2012; Wang et al., 2015). However, what remains unknown is the magnitude of chronic GHG losses that occur each year due to the slow oxidation of vast drained peatlands and rewetting results in significant reductions in GHG losses due to reductions in both peat oxidation and fire as well as an increase in C sequestration rates in rewet areas. These data are critical to quantifying the importance of restored pocosin peatlands on sequestering and preserving C on the southeastern coastal landscape.

1.3 | Objectives

The main objectives of this study were to (i) predict the annual C storage and GHG fluxes for drained versus rewetted (rehydrated) pocosin peatlands using the eddy covariance (EC) method in conjunction with multiple soil chamber measurements of soil respiration; (ii) develop a proxy model from environmental attributes that could be used to predict and quantify net ecosystem exchange (NEE), R_{eco} (ecosystem respiration), and the annual addition of C stored as peat resulting from the rewetting drained pocosins; (iii) provide an assessment of the C stocks at risk on drained pocosin due to peat oxidation or fire; and (iv) convey a scientific basis for forecasting GHG reductions in pocosin peatlands to facilitate sound management decisions regarding C sequestration potential under managed versus unmanaged hydrologic conditions along the southeastern USA coastal region.

2 | METHODS

2.1 | Site descriptions

To address our objectives, we monitored C flux and storage across a vast (>40,000 ha) peatland in coastal North Carolina in various stages of drainage, restoration, and natural conditions to assess the key hydrologic conditions and soil properties controlling soil C stabilization, GHG fluxes, C accumulation, and long-term C storage.

The initial study sites (35°37′–35°44′ N, 76°27′–76°35′ W) were in the U.S. Fish and Wildlife Service (USFWS) Pocosin Lakes National Wildlife Refuge (PLNWR) and on adjoining private drained (PD) but unmanaged peatlands located adjacent to the southern boundaries of PLNWR along the coast of North Carolina (Figure 1a). This paper will designate phase-one study sites (Phase I) as PLNWR and phase-two sites (Phase II) as PD. Natural, drained, and restored subsites were selected at PLNWR and labeled in Figure 1a (details in Wang et al., 2015). Both drained sites at PLNWR and PD sites have been drained for over three decades and have not received any past fertilizer applications. Large drainage canals (>50 km in total length) encircle and cut through both properties with major north to south canals found through the PD property, which creates four major blocks (each bounded by drainage canals) that comprise an experimental 4000 ha carbon farm (Figure 1a).

The coastal pocosin region of North Carolina has a warm, humid climate with an average temperature of 16.8°C annually (January average 6.7°C, July average 26.2°C). Precipitation is uniformly distributed all year, but water tables recede in the summer when evapotranspiration (ET) typically exceeds precipitation and rebounds in the winter when ET substantially declines (Richardson & McCarthy, 1994). Annual rainfall compared with ET shows that 65% (800 mm) of the annual precipitation input of 1230 mm leaves as ET (Richardson & McCarthy, 1994), with groundwater losses less than 1% of rainfall (Heath, 1975). The near-surface layer of natural pocosin peats is thus typically unsaturated, with the water table rarely rising above the ground surface throughout the year and often found 20–30 cm below the surface in the winter, and regularly falling to >100 cm below the surface during summer months of dry years (Richardson, 2012).

Regional pocosin vegetation is typically characterized by a dense growth of broadleaf evergreen shrubs (<2 m high), with scattered pond pine trees (*Pinus serotina* Michx.) (Richardson, 2012; Sharitz & Gibbons, 1982). The dominant

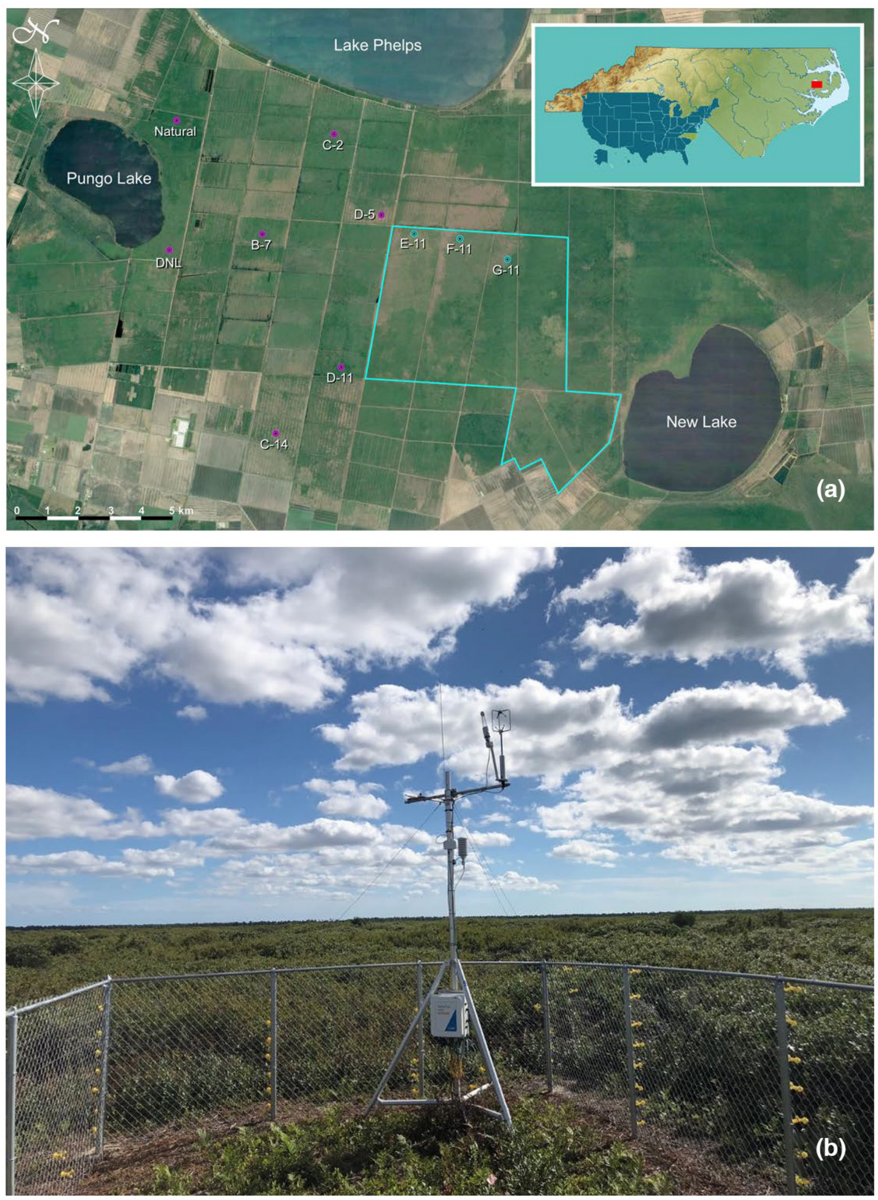

FIGURE 1 (a) Map of the 44,500 ha Pocosin Lakes National Wildlife Refuge (PLNWR) bounded by Pungo Lake to the west, Phelps Lake to the north and New Lake to the east (latitude 35°42′ N, longitude 76°27′ W). In Phase I, C2, B7, and D11 were restored sites in 2015, C14 was a drained site, and natural site was used as a control site in PLNWR. E11, DNL, F11, and G11 are automated water level recording stations as well as D-5. The 4000 ha private drained (PD) farm in Phase 2 is outlined in blue–green in Hyde County, North Carolina, USA. E11, F11, and G11 are all drained sites. The fully functional eddy covariance (EC) tower is located at G11 on PD land. A second EC tower at F11 was used as a comparison check of net ecosystem exchange (NEE) values. Note the extensive canals and ditches throughout the peatlands. (b) Eddy flux tower in a pocosin shrub bog site at the PD F11 site (Phase II). An electric fence was erected to deter bears and wildlife destruction or disturbance.

species are native shrubs, including fetterbush lyonia [*Lyonia lucida* (Lam.) K. Koch], inkberry [*Ilex glabra* (L.) A. Gray], large gallberry [*Ilex coriacea* (Pursh) Chapm.], honeycup [*Zenobia pulverulenta* (W. Bartram ex Willd.) Pollard], and laurel greenbrier (*Smilax laurifolia* L.) with some smaller trees of loblolly bay [*Gordonia lasianthus* (L.) Ellis]. The species composition at both PD and PLNWR systems changed because of drainage and fire. Western brakenfern [*Pteridium aquilinum* (L.) Kuhn] covers large parts of the ground area that have more recently experienced a fire with the post-fire shrub community dominated by titi (*Cyrilla racemiflora* L.), fetterbush, inkberry, and wax myrtle [*Morella cerifera* (L.) Small] with dispersed pond pine and loblolly bay throughout the blocks.

Starting 3–5 cm below the litter surface, the peat layer (up to 5 m deep) throughout the region is black, fine-grained, and highly decomposed with ash content less than 3% (Ingram & Otte, 1982; Wang et al., 2015). Chemical characteristics of peat soil in our study sites at PLNWR and PD displayed no substantive differences in carbon content (52%–56%), nitrogen (1.2%–1.6%), or phosphorus (327–395 $\mu g\, g^{-1}$) with pH ranging 3.8–4.0 on drained PD lands; values slightly higher than the 3.3 pH found on an undrained natural area at (PLNWR). Similar soil characteristics were expected as PD and PLNWR are all part of the same large regional Scuppernong bog system with similar development and drainage history, but PLNWR is government-owned and PD a private landholding.

Fire is a recurring and expected disturbance in these peatlands. These sites' typical long-term fire patterns fit the criteria of a Class 2 fire regime (Heinselman, 1981), with frequent, light surface fires having a 1- to 25-year return interval (Christensen, 1988). Surface fire recurrence under wet or moist soil conditions is important for not only maintaining the native shrub/tree community, but also for producing substantial black carbon (mostly partly burned wood material), which also contains high amounts of aromatic and phenolic compounds (Flanagan et al., 2020; Knicker, 2007; Wang et al., 2015), thus allowing for peat accumulation. The seasonal timing of surface fires in the late winter and spring, when soils are wetter, reduces the likelihood of destructive ground fires. Additionally, destructive ground fires are much less likely to occur in saturated (undrained) peatlands (Flanagan et al., 2020; Turetsky et al., 2011). Thus, management of hydrology and fire intensity in natural and degraded shrub/tree peatlands will be principal to maintaining peat/litter quality (phenolic/black carbon), enhancing long-term carbon accumulation, and preventing downstream DOC losses to coastal waters.

2.2 | Background soil and water analysis

We measured physical and chemistry peat soil characteristics along with collecting data on DOC at both the PD and PLNWR sites. Detailed methods for these measurements along with our approach to water level monitoring, DRAINMOD modeling, and C outflow are presented in the supplemental material.

2.3 | Static soil chamber gas measurements for CO_2, CH_4, and N_2O

In Phase I, the static chamber technique was used to collect gas samples monthly in PLNWR peatlands from in August 2011 to August 2012 to assess the importance of CO_2, CH_4, and N_2O releases to the atmosphere under drained, restored, and natural conditions in pocosin soils (Wang et al., 2015; Wang, Ho, et al., 2021). Drained, restored, and natural sites are shown in Figure 1a, in which drained site (C14) had three subplots, restored sites (C2, D11, and B7) had one plot, and the natural site had one plot. At each plot (50–100 m^2), three metal soil collars (28 cm diameter and 13 cm height) were permanently installed to a depth of 10 cm into the soil at each plot in May 2011. The upper rim of soil collar had a 2 cm by 1.5 cm gutter, which was filled with water to make an airtight seal with the upper chambers when collecting gas samples. The upper chamber (28 cm diameter and 21 cm height) was equipped with a temperature probe and a battery-driven fan for air mixing. Vegetation inside these collars was removed. At each plot, water level was automatically recorded every 60 min by water level data loggers (Solinst Levelogger model 3001). During the gas sampling period, the lower edge of the top chamber was placed into the water-filled gutter of the soil collar. Four gas samples were taken at 10–15 min intervals from the headspace through polytetrafluoroethylene tubes into the 100 ml gas sampling bags (multilayer polymer with aluminum foil) by syringe. Simultaneously, chamber air temperature, soil temperature (Thermocouple Thermometer), and soil moisture (FieldScout TDR 150; Spectrum Technologies, Inc.) were monitored. CO_2, CH_4, and N_2O concentrations were determined within 3 days after sampling by a gas chromatograph (GC; Varian 450) equipped with a flame ionization detector. To keep the accuracy of the analyses, we calibrated the GC against two standard gas mixtures after every eight samples. Soil respiration was calculated from the linear change of its concentrations in the chamber as a function of time, base area, chamber volume, and the molar volume of CO_2 at the air temperature inside the chamber.

2.4 | LI-COR Smart Chamber for CO_2 and CH_4

In Phase II, we used an EC system to estimate the ecosystem-scale gas flux of CO_2 and to further verify CH_4 fluxes, we established soil gas flux plots within the footprint of the EC tower utilizing a LI-COR Smart Chamber portable unit (LI-8200-01S, Smart Chamber; LI-COR Biosciences) system. Eight collars in two clusters along the axis of prevailing wind were established around each tower. Four collars were located NE of the EC tower; another four, SW of the tower. Each cluster was kept at a distance from the adjacent eddy flux station by 25–55 m. A total of 16 collars (4 collars × 2 clusters × 2 towers) were installed for the Phase II gas measurement.

An important part of measurement by the portable soil gas unit was a collar permanently anchored in the soil for the Smart Chamber to fasten on. We selected metal instead of PVC collars to avoid frequent disturbance by wildlife on-site. Aluminum pipe (6061-T6 schedule 40) with 21.9 cm O.D. was fashioned into 11.4 cm length sections. One end of the section was polished to smoothness so to allow for an effective seal with the gas chamber system. The other end was inserted into the peat soil so that a 3–5 cm collar extended above the surface. Any aboveground living plant parts inside the collar were removed to ensure leaf aboveground respiration was excluded.

A laser-based analyzer (LI-7810, $CO_2/CH_4/H_2O$ Trace Gas Analyzer; LI-COR Biosciences) in conjunction with the gas flux survey chamber was used to sample and analyze the soil gas fluxes. In addition to the soil gas fluxes that the core instrument measures, an auxiliary HydraProbe (Stevens Water Monitoring System) provided simultaneous monitoring of soil temperature, moisture, and electrical conductivity. Barring inclement weather, site access denial, or instrument maintenance, we took soil gas flux measurements monthly for nearly 2 years (2019–2021).

2.5 | EC methods

Fluxes of CO_2 and water vapor were quantified with the EC method. At the G11 tower location (Figure 1a), the primary hardware was a SmartFlux 3 data logger (LI-COR, Inc.), Windmaster sonic anemometer (Gill Instruments), and LI-7500DS Open Path CO_2/H_2O Analyzer (LI-COR, Inc.). These flux sensors were installed at a 4.5 m height on an electrically grounded 5 m metal tripod, surrounded by evergreen shrubs with a maximum canopy height of 2.0 m (Figure 1b). In addition, the EC system included the Biomet suite of sensors (LI-COR, Inc.) for energy balance computations. Soil sensors included three HFP01 soil heat flux sensors (Hukseflux Thermal Sensors), installed 5 cm below the soil surface; and three Stevens Hydra Probe II soil moisture and soil temperature probes installed 5, 15, and 30 cm below the soil surface (Stevens, Inc.). All soil sensors were located within 3 m of the EC tripod. Meteorological data were collected using the following sensors: humidity and air temperature with the Vaisala HMP155 Probe installed 3 m above the ground (Vaisala, Inc.); an NR-Lite 2 net radiometer (Kipp & Zonen BV), and an LI-200R Pyranometer (LI-COR, Inc.) both mounted 4 m aboveground (Figure S1a); a TR-525M Tipping Bucket Rain Gauge (Texas Electronics, Inc.), mounted 2 m above ground. The EC data were processed in the field using EddyFlux software embedded within the Smartflux 3 system. Further data processing was done within TOVI software, including QA/QC filtering Foken Flags (Burba, 2013; Burba & Anderson, 2010; Mauder & Foken, 2015), Friction [Shear] Velocity (U*), Gap Filling, and Night-time Partitioning (Reichstein et al., 2005), and footprint estimation (Kljun et al., 2004) systems downloaded from the data loggers and archived. The archive includes concentration, atmospheric pressure, wind speed, moisture, and temperature measurements for the 1-year period of observations. High-frequency data (10 Hz) were used to calculate CO_2, H_2O, and heat fluxes. The EC fluxes were calculated from the raw data averaged over 30 min intervals using EddyPro software (LI-COR Biosciences); EddyPro also conducts statistical tests and corrections, and data processing was performed using conventional methods (Burba, 2013). An identical EC tower (F11) was set up in similar pocosin vegetation 1.7 km away from G11 for C flux comparisons, but electrical issues causing missed data collections only allowing for direct differentiation in some months as well as an estimated net annual NEE value for comparison.

3 | RESULTS AND DISCUSSION

3.1 | Water table patterns

The seasonal hydroperiod of three drained blocks (E11, F11, and G11) at the PD site (Phase II), a restored block (D5), and a natural site (PLNWR DNL) in the adjacent PLNWR (Phase I) is illustrated in Figure 2. All drained sites except E11 in the summers of 2019

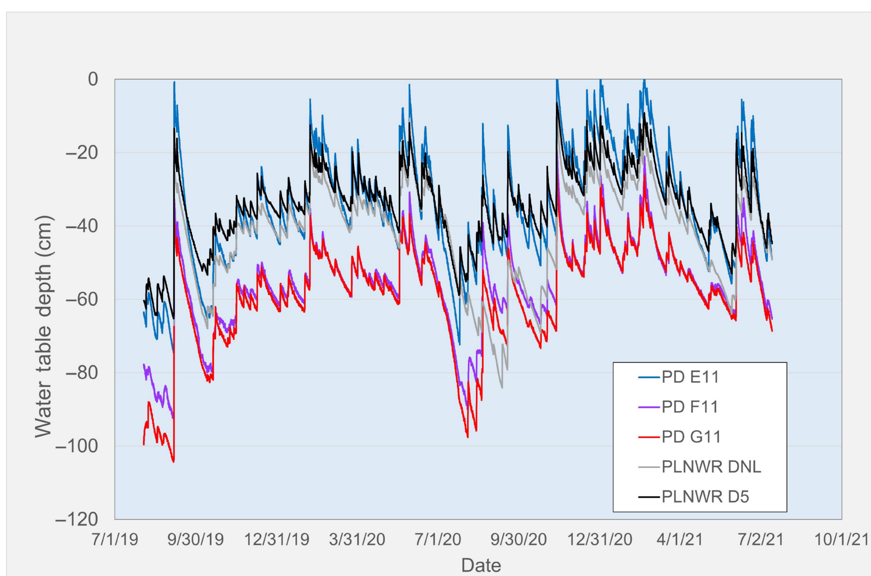

FIGURE 2 Water table depths (WTDs) over a 2-year period in a restored pocosin block in Pocosin Lakes National Wildlife Refuge (PLNWR; D5 block, Phase I) compared with drained pocosin blocks (G11, F11 and E11 blocks, Phase II) at the private drained (PD) site. PLNWR DNL a natural site was also included for comparison. Data from the D5 block are from the USGS well number 354216076271201 located at latitude 35°42′09.8′′ N, longitude 76°27′16.9′′ W. Data available online from the USGS National Water Information System: Mapper, https://maps.waterdata.usgs.gov/mapper/index.html.

and 2020 display increasing water table depth (WTD), indicating dryer soil conditions due to higher solar irradiation and evaporation rates, and a resulting deeper water table, a typical seasonal water deficit observed in many southeastern peatlands (Richardson & McCarthy, 1994; Skaggs et al., 1991; Wang et al., 2015). The northern quarter of the E11 block maintained higher water tables due to the fact this site has lost >0.3 m of peat due to a 2008 fire, thus lowering its peat surface. By August of 2019, WTDs at the drained sites F11 and G11 fell to nearly 100 cm below the surface, while the WTD in the restored site never exceeded 75 cm. A similar pattern occurred in 2020 at all sites. Higher water tables (lesser WTDs) in the non-growing season indicate a rebound of the water table when ET is low. The restored block in PLNWR consistently shows a higher water table than the drained sites at PD, with water level differences averaging nearly 25 cm lower in the drained sites (Figure 2). All sites show a rapid increase in water levels after a rainfall event, for example, in the late summer and early fall of 2019. During the winter of 2019, the drained blocks typically had the water table located at 60 cm below the surface, compared with 35 cm in the restored PLNWR D5 block. A well located in a minimally disturbed area of the PLNWR (DNL) showed a similar pattern to D5 block. The water tables at these sites are highly responsive to rainfall, with water table elevations within the drained sites sometimes increasing by 20 cm in a matter of hours and sometimes declining to pre-storm levels in a matter of days during the growing season. These differences in WTD allowed us to compare GHG fluxes across a range of hydrologic conditions, including soil moisture, and develop models that can be used to relate GHG fluxes to hydrologic conditions.

3.2 | Carbon outflow leakage in canals

The average annual depth-based canal outflow estimates using the DRAINMOD model (Richardson & McCarthy, 1994; Skaggs et al., 1991) for the reference (no disturbance over 30 years) and restored sites in PLNWR were 34 and 36.5 cm year^{-1}, respectively (Table 1). These flows were lower than the 41.6 cm year^{-1} outflow found at the drained site in PLNWR (Table 1). Depth-based outflow estimates were converted to a volumetric equivalent and multiplied by average total organic carbon (TOC) concentrations in the outflow of our drained, reference, and restored sites and yielded an annual estimate of C leakage of pocosin under each treatment condition (Table 1). The TOC concentrations were high but had consistent TOC, as noted by their low standard errors. The drained site exported 0.39 Mg C ha^{-1} year^{-1} of TOC, a value of 15.3% higher than either the restored or natural site. However, compared with soil respiration (5–12 Mg C ha^{-1} year^{-1}, Wang et al., 2015), all losses via water outflow were one order of magnitude lower. As a result, the very low C leakage via runoff can be considered a negligible contributor to C loss from the site. Low drainage exports for C were also reported for similar coastal wetland systems (Amatya et al., 1998).

3.3 | Comparison of CO_2, CH_4, and N_2O fluxes

In Phase I of our study at PLNWR (Richardson et al., 2014; Wang et al., 2015), we determined that CO_2 comprised the overwhelming proportion of GHG emitted from the shrub-bog soil under natural, restored, and drained conditions (Figure 3). Our findings showed that CH_4 and N_2O emissions at PLNWR make an insignificant contribution (<1.5%) to the radiative balance of these peatlands since values were extremely low under all treatment and reference conditions (Figure 3). CO_2 equivalent estimates in the restored, drained, and reference areas in PLNWR showed that CO_2 contributed over 98% of the CO_2-equivalent GHG fluxes for all pocosin sites. Even when CH_4 contributions increased with raised water levels (mostly below 20 cm WTD) in the restored sites, its values were still less than 1.5% of GHG warming potential in the restored site. Methane concentrations were also barely detectable at all tested sites at PD (Phase II) during most months. LI-COR Smart soil chamber data at PD sites between March of 2019 and July of 2021 (Figure 4a,b) further verified the low level of losses of methane from the PD pocosin blocks at E11 (0.023 ± 0.121 nmol m^{-2} s^{-1}), and G11 (-0.227 ± 0.126 nmol m^{-2} s^{-1}) and demonstrated again that soil CH_4 fluxes under a range of soil moisture and temperature conditions were extremely low and supported our earlier static chamber findings at PLNWR (Figure 3). Importantly, most measurements were negative as readings shown below the zero-line on Figure 4a,b indicate no CH_4 efflux but rather an absorption of CH_4 by the soils (median at E11: −0.18, G11: −0.31 nmol m^{-2} s^{-1}). Therefore, in this study, CO_2 is the only significant GHG considered for GHG analysis long-term in pocosins; this was the same finding on pine plantations grown on similar nearby pocosin peat soils (Noormets et al., 2010).

TABLE 1 DRAINMOD estimates of total organic carbon (TOC) export from drained, reference, and restored sites at Pocosin Lakes National Wildlife Refuge in North Carolina

Treatment	Outflow (cm year^{-1})	TOC (mg L^{-1})	TOC export (Mg C ha^{-1} year^{-1})	CO_{2e} export (Mg C ha^{-1} year^{-1})
Reference	34.5	96.7 ± 1.8	0.33	1.20
Restored	36.5	91.1 ± 2.8	0.33	1.20
Drained	41.6	94.4 ± 2.9	0.39	1.42

Note: Dates for flow data used in the model estimates spanned from October 2010 to January 2012. Water chemistry data ($n = 22$) encompass sample results from November 2010 to January 2012. CO_{2e} represents TOC reported as CO_2 equivalents.
Abbreviation: TOC, total organic carbon.

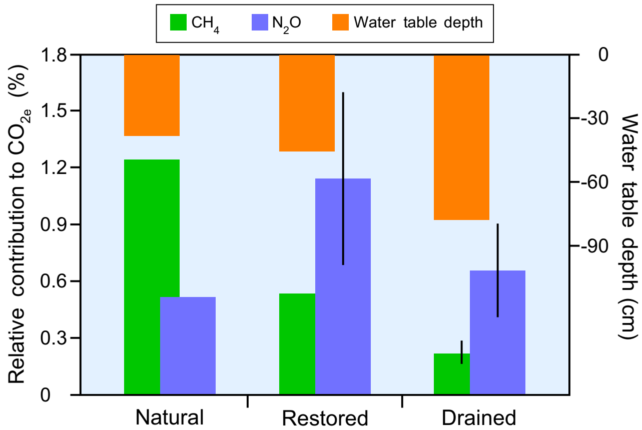

FIGURE 3 Relative contributions (mean ± SD) of CH_4 and N_2O to CO_2 equivalents compared with water levels in natural, restored, and drained areas in Pocosin peatlands in Pocosin Lakes National Wildlife Refuge (PLNWR; Phase I, see Figure 1a) from August 2011 to July 2012. CO_2 equivalents = Soil respiration $[CO_2] + 265 \times [N_2O] + 28 \times [CH_4]$ (IPCC, 2014).

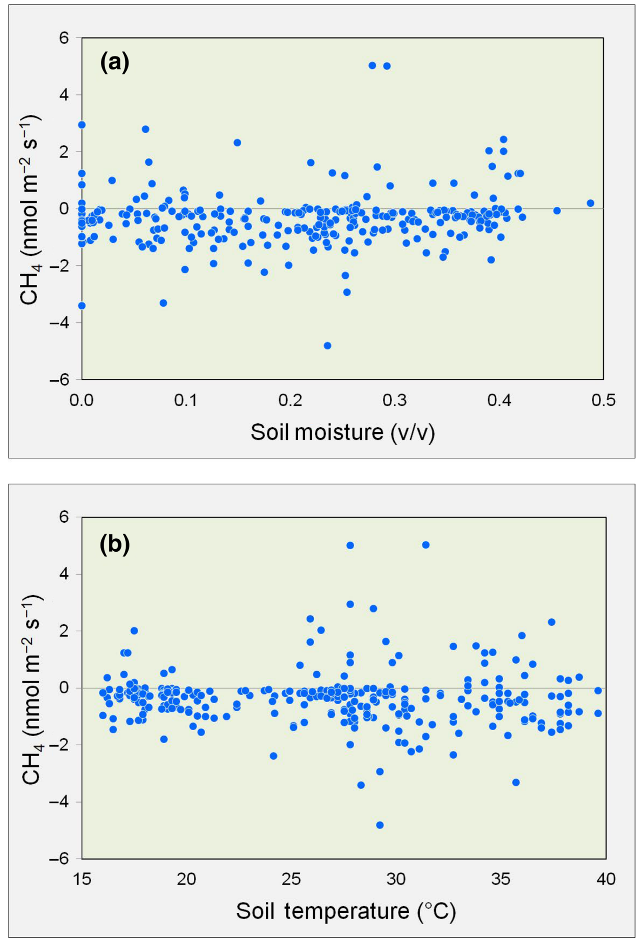

FIGURE 4 Relationship between soil methane flux (emission/uptake) and (a) soil moisture or (b) soil temperature at the drained pocosin private drained (PD) sites in Hyde County NC (Phase II see Figure 1a). Data include measurements from 16 collars that were located adjacent to two eddy flux stations and from additional 26 collars that were installed for reconnaissance purposes before eddy flux tower placement. Data collected from October 2019 through September 2021.

3.4 | Environmental and daily EC measurements

Environmental measurements supporting our EC results in Phase II for site G11 (Figure 1b) are summarized for a year beginning in March of 2020 in Figure S1a–d. Hourly measurements were compiled to give daily readings and then reported as monthly values. These data were essential to our developing and testing a proxy for predicting annual NEE values. The monthly average of total daily solar radiation (R_g) shows a typical yearly variation ranging from 8.2 MJ/m^2 in December 2020 to 25.1 MJ/m^2 in July 2021 (Figure S1a). Daily mean air and soil temperatures display an annual sinusoidal pattern with minimum February temperatures and maximum temperatures in August, with soil temperatures (5 cm below soil surface) showing less short-term variability than air temperature (Figure S1b). Figure S1c,d display annual soil water content (SWC) patterns and WTD throughout the study period. SWC ranges from 0.08 m^3 m^{-3} in dry summer periods to 0.41 m^3 m^{-3} in the winter months with a mean value of 0.25 m^3 m^{-3} (Figure S1c). WTD ranged from −25 to −97 cm and averaged −56 cm below the ground surface over the study period (Figure S1d). Both WTD and SWC responded rapidly to precipitation (Figure S1c), with a large tropical storm on November 12, 2020, depositing 140 mm of rain, causing the water table to rise 30 cm in less than 12 h.

In summary, the highest solar radiation, air, and soil temperature during the summer months have soil moisture volume and WTDs reaching their lowest values in August due to increased plant transpiration and soil evaporation. Water tables rise rapidly after large rainfall events and quickly drop during dry summertime conditions when high rates of evaporation, transpiration, and drainage occur. Our measured environmental values and patterns for solar radiation, vapor pressure deficit (VPD), temperature, and WTDs closely mimic those reported earlier for a nearby study on a coastal plain loblolly pine (*Pinus taeda* L.) forest (Noormets et al., 2010).

Diurnal patterns in gross primary production (GPP), ecosystem respiration (R_{eco}), and NEE were observed, with the largest GPP and R_{eco} values occurring during the day (Figure S2a–c). The R_{eco} values remain above 5 μmol m^{-2} s^{-1} at night, while GPP drops to 0 at nighttime. NEE values are positive at night and negative during the day, indicating the systems sequester carbon during daylight hours and lose carbon at night. VPD, an indicator of water demand by plants, shows a similar diurnal cycle, being lower at night and higher during the day (Figure S2d). Soil moisture content remains relatively constant throughout the day and night (Figure S2e). Both soil and air temperatures show a distinct diurnal pattern, with soil temperature showing a smaller fluctuation amplitude than air temperature (Figure S2f). The timing of peak temperatures of both air and soil corresponded well with peak R_{eco} (14:00–15:00) while lagging several hours behind peak GPP (12:00).

3.5 | Annual EC estimates

These data are the first EC values collected in native shrub dominated, that is, short pocosin sites, the dominant plant community type in the coastal peatlands of North Carolina, Virginia, South Carolina, and parts of Georgia. We integrated daily EC results to estimate NEE in the drained pocosin area at G11 (Figure 1a) at the PD site over an entire year. Time series plot of daily averages of NEE, GPP, and ecosystem respiration (R_{eco}) are presented in Figure 5. For the two pathways (GPP, R_{eco}), positive (+) value for R_{eco} indicates a loss of C to the atmosphere through respiration (Burba et al., 2018). A negative (−) GPP indicates an ecosystem gain of C through photosynthesis. NEE may have negative or positive values that indicate a net increase (−) or decrease (+) of C storage within the ecosystem. A negative NEE occurs when C gains in biomass plus organic matter exceed C losses due to ecosystem respiration. Figure 5 shows that NEE values were predominantly positive throughout the year indicating a net decrease in net carbon stores as losses from R_{eco} exceed C gains from GPP. Both GPP and R_{eco} were relatively low in the non-growing season months (mid-November through mid-March), with R_{eco} exceeding GPP. Both GPP and R_{eco} increased in the growing season due to longer daylight periods and higher temperatures (Figure S1a,b), with increases in R_{eco} being primarily a function of higher temperatures. Due to the drained conditions, rates of R_{eco} are very high throughout the year, but particularly during the growing season when soil temperatures are highest. As a result, there are only brief periods of the year where GPP exceeds $R_{eco,}$ resulting in negative NEE during the late-spring and early-summer. NEE remains positive for the remainder of the year, indicating R_{eco} is greater than GPP (Figure 5). Both R_{eco} and GPP decline in the winter when WTD decreases. However, in these drained conditions, winter-time C losses to R_{eco} remain high enough to more than offset C gains (sequestration) during the brief periods of negative NEE during the growing season. These results indicate the effect of continued drainage, lower water tables, and higher temperatures on C losses to the atmosphere. The accumulation of half-hourly values in NEE storage through the year quantify the cumulative gains of C stores GPP (negative values), losses of C through respiration R_{eco} (positive values), and net C accumulation (−) or loss (+) indicated by NEE through the year (Figure 6). Cumulative NEE increased (C loss) through the early spring when vegetation emerged from dormancy and declined slightly during the period of highest GPP in late spring and early summer. NEE consistently increased through the remainder of 2020 even as the plants started to senesce in late summer and remained dormant through the winter and early spring of 2021, resulting in a substantial cumulative annual loss of 21.2 Mg CO_2 ha^{-1} $year^{-1}$ due to drainage conditions.

3.6 | Development of a C storage proxy model

We modeled the relationship between NEE and multiple environmental variables collected at the EC tower at a 30-min interval. Water table data, collected at 1-h intervals were interpolated to 30 min to be consistent with other tower measurements (G11, Figure 1b) to test and develop the best proxies for predicting C storage or losses. First, a model relating NEE to all measured environmental variables was created. We then performed a backward/forward stepwise regression procedure to create a parsimonious model. The resulting model is presented in Table S1. The most robust model relates NEE to R_g (global radiation) and WTD and has an R^2 of 0.73. Global radiation R_g explains most of the variation in NEE, with WTD explaining the remaining variation. Importantly, our model is more robust than models using only WTD (Couwenberg et al., 2010) as it captures variance of NEE due to both key driving variables: (i) seasonal patterns of solar radiation and temperature and (ii) changes in both short- and long-term WTD. The high solar radiation values in the

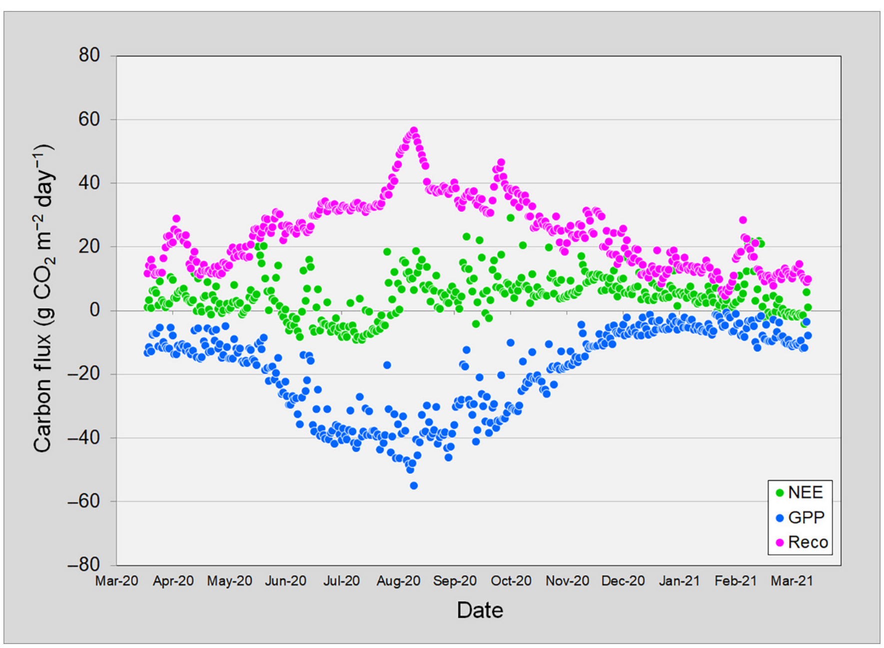

FIGURE 5 Daily eddy covariance (EC) carbon flux measured throughout 1 year in a pocosin shrub bog ecosystem at the private drained (PD) G11 site in Hyde County NC (Phase II). Net ecosystem exchange (NEE) is compared with ecosystem respiration (R_{eco}) and gross primary productivity (GPP).

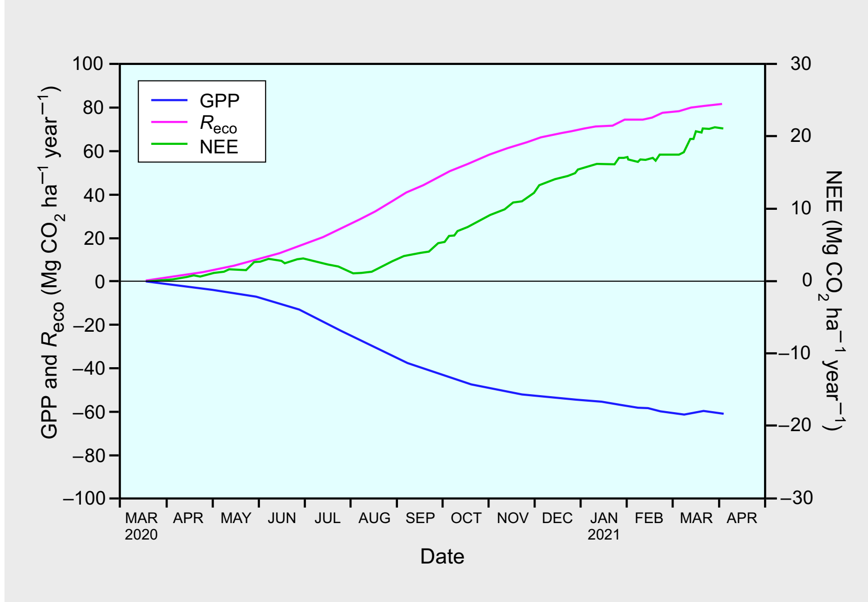

FIGURE 6 Cumulative carbon flux values for net ecosystem exchange (NEE) compared with ecosystem respiration (R_{eco}) and gross primary productivity (GPP) on G11, a drained pocosin peatland site at the private drained (PD) site in Hyde County NC (Phase II). Note the y-axis scale changes.

summer increases clearly drive GPP and C sequestration (Figure 6). The short-term changes in WTD follow storms, with mid-term changes driven by seasonal weather patterns and long-term changes in WTD driven by landscape-level factors such as catastrophic fire and human alteration of hydrology (e.g., drainage and restoration). While WTD provides a fundamental component to predicting long-term C sequestration and flux values in northern peatlands the response times to the time-scale changes noted above may reduce its accuracy in STWPs. For example, in pocosin coastal-plain peatlands, we found that our models using only WTD as a proxy does not accurately predict hourly NEE measurements within a single year, where the relationship with WTD varies seasonally. However, the addition of solar radiation to the model helps explain this high-resolution data much more effectively. At a multi-year timescale, this seasonal pattern is likely to cancel out. Additional model variables, like soil moisture and temperature, do not increase model accuracy and precision enough to justify the accompanying loss of parsimony. However, global radiation greatly increased model accuracy, as was found by Aguilos et al. (2020). Notably, the availability of solar radiation data from local weather stations along the US coast makes this variable quite realistic as solar radiation drives seasonal patterns of plant photosynthesis and ET, both key processes in the storage and loss of C within the ecosystem. When combined the two variable proxy models using solar radiation and WTD provide for a highly predictive model of C sequestration and loss rates in pocosin peatlands (Table S1).

To validate our proxy model, we tested it against an incomplete NEE dataset produced by a second Eddy-flux tower in the F11 block March 2020 and March 2021 and found that predicted cumulative annual NEE estimates from our two-parameter model were conservative and within 18% of our gap-filled tower measurements, thus providing further support for our two-parameter approach (Table S2). Our earlier findings showed that smaller WTDs (higher water tables) substantially reduced CO_2 losses to the atmosphere with almost no increase in N_2O or CH_4 losses (Wang et al., 2015; Wang, Ho, et al., 2021). This result suggests that by using the improved two-variable model, we could create scenarios for changes in CO_2 losses by decreasing observed hourly WTD measurements (i.e., raising the water table) by either 40 or 30 cm from baseline WTD (60 cm), such that the simulated annual mean WTD was 20 cm in the first scenario (REST_20) or 30 cm (REST_30) in the second scenario. We then compared simulated (restored) NEE values to those observed under the drained (unrestored) baseline conditions currently found at our sites, with the annual mean WTD being 60 cm between March 2020 and March 2021 (Figure S1d) and in a longer-term observed dataset shown in Figure 2 where WTD values often exceeded 75 cm. Global radiation values were the unaltered observed values collected from our instruments at G11. The three model scenarios include (i) a baseline representing altered conditions (unrestored, 60 cm), (ii) rewetted where annual average WTD is increased to 30 cm below the soil surface (REST_30), and (iii) a scenario, where the average annual WTD is raised to 20 cm below the soil surface (REST_20). Model outputs and variable parameters from these scenarios are presented in Figure 7 and Table 2, respectively.

Monthly mean NEE values under three scenarios, baseline, raised water tables to 30 cm and 20 cm from the peat surface, followed a sigmoid curve pattern (Figure 7). Average monthly NEE remained positive for most of the year and barely attained negative values during the summer months in the drained baseline conditions. In the REST_30 scenario, non-growing season losses of C are substantially lower, with long periods of markedly negative NEE (C storage), mainly occurring in the summer of 2020. By comparison under the REST_20 scenario, monthly NEE values are incrementally more negative than in the REST_30 scenario throughout the year, suggesting an even higher C accumulation rate when the water table is kept closer to the surface.

A summary analysis of cumulative annual NEE is presented in Table 2. Under the observed (drained baseline) scenario, annual

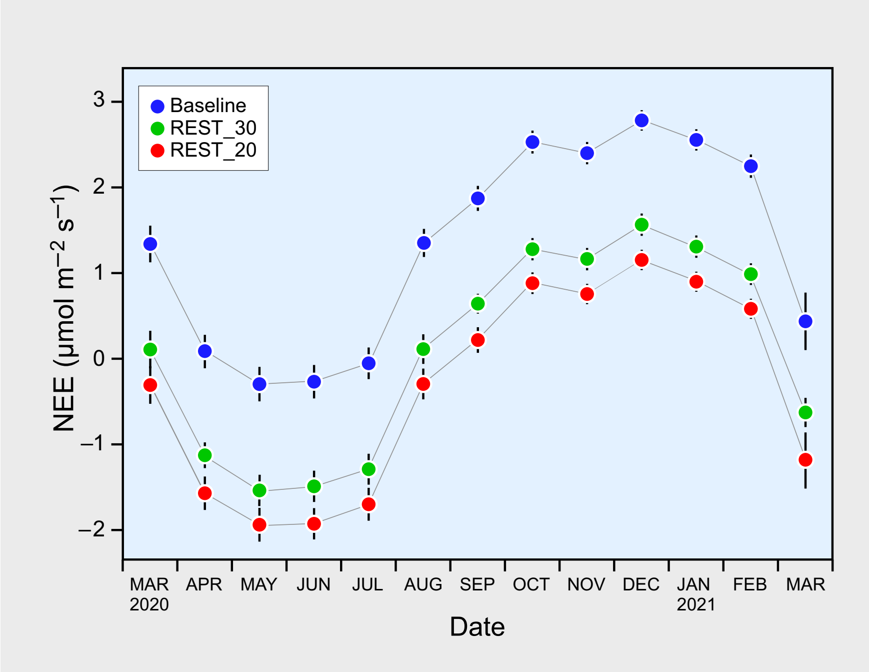

FIGURE 7 Monthly mean net ecosystem exchange (NEE) values under the three scenarios; drained (baseline), water table depth (WTD) raised to 30 cm below peat surface (REST_30), and WTD raised to 20 cm (REST_20) below peat surface on pocosin peatlands. For model inputs see Table S1.

TABLE 2 Modeling results comparing the CO_2 stored by raising water tables to within 30 or 20 cm of the surface compared with drained pocosin sites (baseline) net ecosystem exchange (NEE) value where water tables can drop to >100 cm below the surface

Scenario	Mg CO_2 ha^{-1} year^{-1}	Delta
Baseline (unrestored, WTD = 60 cm)	21.2	–
Fitted REST_30 (restored, WTD = 30 cm)	2.0	19.2
Fitted REST_20 (restored, WTD = 20 cm)	−3.3	24.5

Note: Water table depth (WTD) refer to annual mean depth below soil surface. Delta is the difference between the restored scenarios and baseline (unrestored) scenario.

C losses were 21.2 Mg CO_2 ha^{-1} year^{-1}, a value 18% lower than C losses of 25.8 Mg CO_2 ha^{-1} predicted using the Couwenberg WTD model (Couwenberg et al., 2010). By raising the mean annual WTD from 60 to 30 cm (REST_30), annual C losses are decreased to only 2.0 Mg CO_2 ha^{-1} year^{-1}. This change translates into a creditable reduction in the loss of CO_2 to the atmosphere of 19.2 Mg CO_2 ha^{-1} year^{-1} (Table 2). Our mitigation reduction estimates at 30 cm are close to the projected 15.3 Mg CO_2 ha^{-1} year^{-1} loss reported by Evans et al. (2021) when halving drainage depths on multiple croplands on organic soils. In the REST_20 scenario, the peatlands achieve a predicted annual NEE of −3.3 Mg CO_2 ha^{-1} year^{-1}, thus forecasting an increase in net CO_2 sequestration on an annual basis. The difference between the observed baseline scenario and the REST_20 scenario is 24.5 Mg CO_2 ha^{-1} year^{-1} of carbon storage when 21.2 Mg CO_2 ha^{-1} year^{-1} of baseline loss is combined with −3.3 Mg CO_2 ha^{-1} year^{-1} storage (Table 2). The amount of projected C stored via rewetting at the PD shrub-dominated sites falls below the range of C sequestration measured (−13 to −36 Mg CO_2 ha^{-1} year^{-1}) in a nearby fast-growing pine plantation forest growing on shallow organic Belhaven peats and lower soil moisture conditions (Noormets et al., 2010). However, forestry operations optimize drainage conditions and often use fertilizer to maximize aboveground production.

Our model results suggest a WTD threshold exists between annual mean WTD of 15 and 30 cm. It is difficult to precisely identify this threshold without accounting for an up-to-date estimate of carbon leakage through ditches at our current study sites, rather than the older estimates from the PLNRW. Below this threshold (drier), the peatland shows an annual C loss (positive NEE), and above the threshold (wetter), the peatland shows annual C gains (negative NEE). This result corresponds well with Evans et al. (2021) who found a similar threshold at 18 cm. Our predicted incremental effect of reducing the WTD by 10 cm (e.g., from 30 to 20 cm) provides an estimated CO_2 storage value of 5.3 Mg ha^{-1} (2.0 and −3.3 Mg ha^{-1} combined = 5.3 Mg ha^{-1}, Table 2). This increment nearly doubles the findings of Evans et al. (2021), where a 10 cm reduction in WTD results in an about 3 Mg ha^{-1} increase in CO_2 storage. The projected decrease in CO_2 losses at PD and PLNWR to the atmosphere due to a rise of the water table suggests that rewetted (rehydrated) pocosins can be a substantial C sink on the landscape. While it has been shown in other studies that rewetted peatlands can become CO_2 sinks, with large positive climatic effects coming from the avoidance of large CO_2 emissions from drained peatlands (Couwenberg et al., 2010; Joosten et al., 2016; Morse et al., 2012), problems often exist with elevated CH_4 and N_2O releases (Bossio et al., 2020), which is not the case in pocosins. Additionally, it has recently been argued using a radiative forcing model and data from the Global Peatland Database that even for peatlands that release substantial amounts of CH_4, the radiative forcing does not undermine the climate change mitigation potential of peatland rewetting (Günther et al., 2020). Consequently, with rewetting, pocosin bogs have very high-quantity C storage values on the landscape due to their low CH_4 and N_2O

emissions (Wang et al., 2015; Wang, Ho, et al., 2021). Also, under drought conditions and increased temperatures, pocosin have lower decomposition rates than northern peatlands and continue to sequester more decay-resistant peat due to the buildup of C materials high in concentration of phenolics and aromatics coupled with the recently discovered unique feature of being dominated by slow-growing fungal decomposers (Hodgkins et al., 2018; Richardson et al., 2014; Wang et al., 2015; Wang, Tian, et al., 2021). However, field observations showed differences in CH_4 emissions between boreal *Sphagnum*-dominated peatlands (Turetsky et al., 2014), and wood-derived peat in pocosin peatlands were negligible with water-table levels just 10 cm below the soil surface (Gutenberg et al., 2019; Wang, Ho, et al., 2021).

3.7 | Carbon stock response to annual oxidation, fire, and water level

Günther et al. (2020) argue that postponing rewetting increases the long-term warming effect through continued CO_2 emissions, and without rewetting, the world's drained peatlands will continue to have a direct negative effect on the magnitude and timing of global warming. Clearly, the sooner drained wetlands are rewet, the quicker climatic benefits of reducing CO_2 emissions will occur. In addition, the rewetting of peatlands dramatically reduces the risk of CO_2 emissions from catastrophic ground fires and deep peat burns (Flanagan et al., 2020).

To determine the amount of carbon already stored in the peat soil at the PD site that is at risk of being oxidized away or burned in intensive fires due to current drainage conditions (>50 km of canals), we took >100 peat depth probes to the top of the mineral substrate across the peat blocks assigned to the potential carbon farm. The average depth was 2.3 m, and the soil had an average bulk density (BD) of 0.15 g cm^{-3} with an elemental C content of 52% and an ash content of 2%. The peat mass (peat depth × area × BD) equaled 3450 Mg peat ha^{-1}, representing 6566 Mg CO_2 ha^{-1}. The carbon farm area (4046 ha) multiplied by the mass of 3450 Mg peat ha^{-1} equals 13.9 Tg of peat (1 Tg = 10^{12} g), which converts to 7.3 Tg of C or 26.6 Tg of CO_2 stored in the ground at the PD study area, which is subject to annual slow oxidization losses or rapid, intensive fire loss in its current drained state. To estimate the potential loss of this carbon at the PD site (stop/loss), we looked at both decomposition rates and losses during fire events. We determined from our current data, earlier studies, and published reports (Bridgham & Richardson, 1992; Flanagan et al., 2020) that 0.5–1 cm of peat soil is oxidized away on drained pocosin peatlands each year, which represents between 13.7 and 27.5 Mg of CO_2 loss per hectare per year indicating ~56,000 to 111,000 Mg of CO_2 loss per year on the PD area alone. In terms of fire losses, we used regional pocosin studies that measured losses of peat that ranged 1–2 cm with light fires and average regional losses of 40 cm in severe ground fires in PD peatlands (derived from Mickler et al., 2017). On the currently drained PD site, vertical peat losses of 20, 30, or 40 cm due to fire would result in CO_2 losses of 2.2, 3.3, or 4.4 Tg, respectively. A total peat burn at the PD site would result in 26 Tg of soil C being lost to the atmosphere as CO_2. Thus, rehydration (rewetting) of the peat soil is critical to stop the annual loss of C to the atmosphere as well as rapid C consumption via fire on coastal drained peatlands in the Southeast US.

Regional studies of the pocosin peatland soils in the NC Coastal Plain alone show they comprise >800,000 ha and have an estimated 325 Tg of C (1193 Tg of CO_2) stored in deposits at depths up to 5 m (Ingram & Otte, 1982; Richardson, 2012), but they are susceptible to deep peat muck fires if drained (Mickler, 2012, 2021; Mickler et al., 2017). While C emissions from surface fires in undrained pocosin wetlands are common and occur naturally in shrub-scrub and pine woodlands at fire intervals of 20–80 years (Poulter et al., 2006; Richardson, 1981), most surface fires are low severity with respect to peat combustion (Flanagan et al., 2020). Catastrophic ground fires (deep peat fires) occur on a decadal cycle, mainly on sites where long-term drainage has dropped water tables 1.5–2 m below the surface (Flanagan et al., 2020). For example, the Allen Road fire, which occurred in March 1985 on the PLNWR was ~40,000 ha in size on drained peatlands and had an estimated total C loss ranging from 1 to 3.8 Tg (or 3.5–13.2 Tg of CO_2) with a varied spatial burn pattern resulting in C emissions of 2–110 Mg C ha^{-1} (Poulter et al., 2006). In June 2008, a wildfire was ignited by lightning on the drained PD site abutting the PLNWR peatlands that had partially burned in the 1985 Allen Road fire. This fire's total emissions from combustions of organic soil carbon were estimated at 9 Tg of carbon (32 Tg of CO_2; Mickler et al., 2017). Importantly, a minimal fire effect on peat consumption was found in areas with higher water tables by Mickler et al. (2017). Surficial losses of peat during prescribed burns were also assessed in pocosin bogs located in Green Swamp, NC (Reardon et al., 2007) across a range of water table positions. During a dry-condition fire, where the WTD was 66 cm, and vertical losses of peat ranged from 12.5 to 24 cm. During a wet condition burn in 1998, the WTD was between 30.4 and 38.1 cm, and there was little or no vertical peat consumption. Our measurements of two prescribed fires in PLNWR in March 2015 also showed that when WTDs were near 30 cm, vertical peat losses ranged 1–2 cm (Flanagan et al., 2020). We also found that in a restored block at PLNWR, when the WTD was 8 cm, there was no observed loss of peat from fire. Collectively, these findings add further support for our proxy model results that suggest water tables should be maintained no further than 15–30 cm below the surface to substantially decrease CO_2 emissions and greatly reduce the likelihood of catastrophic deep peat burns. Furthermore, C lost to the ignitions of living aboveground biomass by surface fires can be recouped within a few decades, while soil C losses from deep peat fires could take millennia to recapture.

Clearly, rewetting and restoring the hydrologic patterns (hydroperiod) in pocosin are critical to sustaining ancient C stores nearly 10,000 years old (Hodgkins et al., 2018). Notably, the importance of stopping catastrophic fires in peatlands to preserve C soil stocks (stop loss) is partially incorporated in C capture protocols like the ACR and VCS by allowing C credit additions of up to 20% per year for fire prevention (ACR, 2017; VCS, 2017). However, much more attention is required to mitigate and prevent conditions favoring deep ground-fires in peatlands (water table levels and fuel loads) if we are to realize net-zero GHG emissions by 2050; a requirement under the Paris Agreement (Paris Agreement to the United Nations Framework Convention on Climate Change, 2015). Importantly, because of climate change concerns, there is a growing emphasis on negative emission strategies like reducing existing emissions from drained peatlands to offset fossil fuel emissions (Evans et al., 2021).

3.8 | Scaling carbon sequestration potential in peatlands along the southeastern USA coast

The area of restorable peatlands (Histosols) along the southeastern coast of the US was estimated using a county-by-county analysis of the USDA (http://go.usa.gov/ksU9) soil database gSSURGO (Table 3). Unlike soils with shallow organic layers (such as soils with histic epipedons), Histosols, by definition, have a surface organic layer at least 0.61 m thick (Soil Survey Staff, 2014). Therefore, areas of Histosols with land-uses consistent with an artificially drained state (agriculture, pasture, or other non-wetlands) were classified as restorable. Areas of Histosols without evidence of human alteration were not classified as restorable because they appear to be functional peatlands already. North Carolina has the largest area of peat soil, followed by Georgia, with Virginia and South Carolina having the smallest area considered potentially available for restoration. While most of the peat soils found in each state are in native vegetation, between 450 and 770 km^2 of restorable drained peatlands are used for active agricultural, forestry production, or other non-wetland land uses, making these areas of drained histic soils high potential restorable peatlands (Table 3). This amount can be considered a very conservative estimate of restorable peatlands. The non-wetland classification of landcover in these areas implies a high degree of drainage where restoration would produce substantial functional lift, with NC having over 94% of these areas with high restoration potential.

Using the previous EC measure of cumulative annual loss of 21.2 Mg CO_2 ha^{-1} year^{-1} found on drained peatlands at PD, and assuming similar conditions for these peatlands, it is estimated they are losing somewhere between 963,000 and 1,627,000 Mg of CO_2 of additional carbon to the atmosphere each year, which could be greatly reduced if a large portion of these areas were rewet. Additionally, we calculated that ground fires in unrestored, drained peatlands with a vertical loss of 40 cm, recurring every 30 years (Mickler et al., 2017; Poulter et al., 2006), would result in carbon losses of 36.7 Mg CO_2 ha^{-1} year^{-1}, so the annual losses from microbial respiration and fire are much as 57.9 Mg CO_2 ha^{-1} year^{-1} (Table S3), with total annual regional emissions of between 2.6 and 4.4 Tg CO_2 year^{-1} from restorable peatlands. Loss of the entire regional peat C store in restorable wetlands, due either to steady microbial oxidation or rapid pulsed oxidation during catastrophic fires, suggests up to 129 Tg of CO_2 are at risk of loss, assuming complete loss of the 0.6 m layer of organic matter that defines histosols, and potentially more if average peat depth is greater than 0.6 m. In addition, there are large areas of partially drained peatlands across the southeast classified by the National Wetland Inventory (USFWS, 2019) as "partially drained/ditched." For example, in North Carolina alone, approximately 125,000 ha of partially drained histosols are hydrologically altered but maintain soil moisture sufficient to support hydrophytes. These areas could potentially experience a reduction of CO_2 emissions with restoration; however, we did not include these areas in our estimates of regional C sequestration estimates due to difficulties in assessing the highly variable degree of drainage. Thus, rewetting drained and fallow peatlands along the southeastern coastal plain, which only cover less than 0.01% of the US land area, would substantially reduce GHG emissions and could potentially contribute between 1.4% and 2.4% of 0.18 Pg CO_2, the annual reduction increment required to reduce the current 5 Pg of annual US CO_2 emissions (NAS, 2021) to net-zero emissions by 2050 (28 years) and restore vital wetland habitat with more environmental services.

4 | CONCLUSIONS

Our overall results demonstrate that drained pocosin are a major annual CO_2 source to the atmosphere on the coastal North Carolina landscape, and rewetting (rehydrating) these peatlands could restore one of their main ecological functions, C sequestration. More

TABLE 3 A conservative estimate of the carbon stores and annual loss via oxidation (21.2 Mg CO_2 ha^{-1} year^{-1}) in drained peatlands in North Carolina (NC), South Carolina (SC), Georgia (GA), and Virginia (VA) along the Atlantic Coast of the USA

State	Histosols area[a] (ha)	Restorable cropland and pasture[b] (ha)	Restorable non-wetland[c] (ha)	Oxidation loss rate[b,c] (Mg CO_2 year^{-1})
NC	518,900	43,861	71,320	930,000–1,512,000
SC	46,726	1185	3104	25,000–66,000
GA	136,987	209	1159	4400–25,000
VA	55,379	186	1154	3900–24,000
Total	757,992	45,441	76,737	963,000–1,627,000

Note: Area and values were determined for each coastal county from the gSSURGO database available from USGS (Soil Survey Staff, 2019).

[a]gSSURGO: Gridded Soil Survey Geographic database.

[b]Cropland and pasture include Histosols in agricultural land uses.

[c]Non-wetland includes Histosols with all non-wetland current land uses.

than 600,000 ha of North Carolina peatlands currently in a highly drained state (primarily for crops) create a significant radiative force for GHGs on a regional basis. Our EC measurements of CO_2 fluxes under different seasons and water levels allowed us to develop a model that can be used to predict changing GHG losses in response to alterations of WTD and solar radiation. We found that raising the water table 10 cm (e.g., from 30 to 20 cm) results in ~5 Mg net CO_2 storage per hectare annually. Our analysis also shows that maintaining a WTD between 15 and 30 cm from the peat's surface would decrease net annual C losses by a range of between 19.2 and 24.5 Mg CO_2 ha^{-1}. An annual vertical loss of 1 cm of peat soil due to soil oxidation represents 27.5 Mg CO_2 ha^{-1} $year^{-1}$, and results in 85,000–110,000 Mg CO_2 annual loss in our PD study area alone. However, we estimated that losses of 20, 30, and 40 cm of peat due to fire at the drained site would result in 2.2, 3.3, and 4.4 Tg of CO_2 loss for decades, respectively, from the PD study area. A total burn in 1 year of the entire peat layer at PD would result in a total loss of 26 Tg of CO_2 currently stored in the soil as peat. The restoration of similar drained peatlands along the Atlantic seaboard could prevent an additional 1–1.6 Tg of CO_2 from entering the atmosphere each year. If only 25% of the drained peatlands along the southeastern seaboard burned in any 1 year it would release nearly 32 Tg of CO_2, 18% of the annual USA reduction goal of 0.18 Pg CO_2. Thus, rewetting not only returns pocosin peatlands as major C sinks on the coastal landscape, but also restores hydrologic conditions that prevent catastrophic ground-fire losses and restores degraded habitat for many endemic plant and animal species.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available in the Duke University Research Data Depository (https://doi.org/10.7924/r46w9hp7v).

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SUPPORTING INFORMATION

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