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RESEARCH ARTICLE

Current and future potential net greenhouse gas sinks of existing, converted, and restored marsh and mangrove forest habitats

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Marsh and mangrove forest habitats are productive at capturing and storing carbon, thus actions to protect and create coastal blue carbon sinks could help mitigate global warming. Dredged material is often used to create coastal habitats and evaluating the carbon impact of placement alternatives (PA) could help inform restoration and climate policies. Output from a Delft3D-FM morphodynamics and hydrodynamics model informed a Coastal Wetlands Carbon Model at years 2020, 2025, 2030, and 2050. Three model simulations were used and included (1) no restoration (PA1), (2) restoration dominated with mangroves (PA2), and (3) restoration dominated with marshes (PA3) at a different location. Habitats of brackish marsh, saline marsh, mangrove forest, and saline open water that surround Port Fourchon, Louisiana, U.S.A., were evaluated to estimate the net greenhouse gas (GHG) flux of the study area with and without restoration. In years 2020 and 2025, the study area was estimated to be a net GHG sink (−1.1 ± 0.2 MMT CO_2e) with or without mangrove and marsh-dominated restoration. At years 2030 and 2050, even with habitat loss due to sea-level rise, the study area for all simulations was projected to remain a net GHG sink. At year 2050, +0.1 ± 0.04 MMT CO_2e could be avoided with restoration. At the restoration project scale, mangrove-dominated restoration (PA2) had net GHG sinks (−0.07 to −0.09 MMT CO_2e) near the marsh-dominated restoration (PA3, −0.09 to −0.13 MMT CO_2e). Thus, these modeled results could help inform future restoration planning and climate policies.

Key words: blue carbon, coastal habitats, dredged material, ecosystem model, marsh creation

Implications for Practice

- Coupling morphology, hydrodynamics, and ecological models can be a useful tool to simulate carbon impacts of dredged material placement alternatives in coastal habitats at the study area and restoration project area spatial scales.
- At a restoration project area, with half the amount of dredged material, the mangrove forest habitats can support a net greenhouse gas (GHG) sink over 30 years with values near those of the restoration project dominated by marshes.
- Wetland loss can influence the reduction of the net GHG sinks at the study area and at the restoration project spatial scales.
- Understanding carbon losses following the conversion of vegetated wetlands to saline open water could help better determine the net GHG sink status of these dynamic ecosystems.

Introduction

Coastal habitats, especially those dominated by mangrove forests and saline marshes, are considered natural solutions to mitigate climate change (Bouillon et al. 2008; Bianchi et al. 2013; Kelleway et al. 2017). Marsh and mangrove forest habitats are known as blue carbon ecosystems because they are vegetated coastal ecosystems that are tidally influenced, capture carbon in their aboveground and belowground biomass, and lock the carbon away in their soils, thus producing efficient carbon sinks (Doughty et al. 2015; Kelleway et al. 2016). The soil pool tends to contribute the most carbon (∼60–80%) of the total carbon stock of these coastal ecosystems (Fourqurean et al. 2012;

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Doughty et al. 2015). Flooded soils are important reservoirs of carbon that keep carbon buried for the long-term (Baustian et al. 2021). Compared to temperate wetlands and boreal peatlands, the large area of tropical/subtropical wetlands across the globe allow them to produce high carbon retention rates (Mitsch et al. 2012). In less than a decade, mangrove encroachment into subtropical salt marshes can increase the total wetland carbon stock of an ecosystem by 22% because of the higher aboveground biomass (Doughty et al. 2015). Thus, actions to protect and create these marsh and mangrove forest habitats can be important to support carbon sequestration and to help mitigate the impacts of climate change (Chmura et al. 2012; Chmura 2013).

Restoring marsh and mangrove forest habitats by placement of dredged material is a common technique in ecosystems that have a limited supply of mineral sediment (DeLaune et al. 1990; Stagg & Mendelssohn 2011). The placement of dredged material can occur from bulk placement to thin layer application (Madrid et al. 2012; Suedel et al. 2021). Placement of uncontaminated dredged sediment provides a solution to increase the elevation capital of wetlands, helping them to adapt to rising sea levels (Baustian & Mendelssohn 2015; Davis et al. 2022). It can also help increase resilience of existing wetland habitats, thus restoring and protecting ecosystem functions (VanZomeren et al. 2018; Harris et al. 2021). For example, the ecosystem function of soil carbon storage of created mangrove forest habitats can reach equivalence of reference conditions faster than other wetland habitats (Osland et al. 2012), and drivers such as stem density can influence those carbon fluxes and pools (Abbott et al. 2019).

Various states in the United States have climate strategies where natural lands (under the Land-use Change and Forestry sector) account as an important greenhouse gas (GHG) sink (Center for Climate and Energy Solutions 2021). The Governor in the State of Louisiana, U.S.A., declared an executive order in 2020 that included the following GHG reduction goals: by year 2025 to reduce to 26–28% of the 2005 levels, by 2030 to reduce to 40–50% of the 2005 levels, and by 2050 to be net carbon neutral (Climate Initiatives Task Force 2022). State GHG inventory tools are readily available from the U.S. Environmental Protection Agency (EPA), and natural carbon sinks are listed as an important component in the GHG inventory (Dismukes 2021). However, local estimates of current and future predicted coastal habitat area conditions and their net GHG fluxes (in carbon dioxide equivalents) are lacking for these state inventories. Therefore, quantitatively assessing current and future net GHG fluxes of coastal habitats could inform restoration management and local policies to determine their potential impact in mitigating climate change (Pendleton et al. 2013). Our main research question was: What is the net GHG flux of coastal habitats that are created and protected over time from the placement of dredged material? Our goal was to use available data and modeling tools to assess coastal habitat areas and their corresponding flux of GHG to estimate current and future net GHG fluxes from the restoration action of placing dredged material and from conversion of habitats to open water due to edge erosion. Our approach included investigation at two spatial scales to allow for (1) an ecosystem approach where the larger study area allows for assessment of protected and restored marsh and mangrove forest habitats and (2) at a specific restoration project scale to estimate the influence of certain actions. Future predicted environmental drivers of relative sea-level rise (RSLR), water level, salinity, winds, waves, and tides were also considered in the modeling tools to assess net GHG fluxes at years 2025, 2030, and 2050 that align with local restoration management strategies and policies that are supporting natural climate solutions in the State of Louisiana (Climate Initiatives Task Force 2022).

Methods

Study Area—Port Fourchon, Louisiana, U.S.A.

Our study area was located near Port Fourchon, Louisiana, U.S.A. (Fig. 1). This study area (58,708 ha) is considered a subtropical salt marsh-mangrove ecotone that experiences microtides averaging 0.3 m in range (McKee & Vervaeke 2018). The RSLR is near 9 mm/year (National Oceanic and Atmospheric Administration Tides & Currents 2019), one of the fastest in the United States (Boon et al. 2018). The brackish and saline marsh habitats are dominated by *Spartina patens* (saltmeadow cordgrass), *Spartina alterniflora* (smooth cordgrass), and the mangrove forest habitat is dominated by *Avicennia germinans* (black mangrove). These marsh and mangrove forest habitats surround Port Fourchon, which is a major hub for the offshore oil and gas industry (Henry & Twilley 2013; Osland et al. 2020). Approximately, 90% of the U.S. Gulf of Mexico offshore oil and gas activities are serviced out of Port Fourchon (Lauren C. Scott and Associates, Inc. 2014). Continuous dredging of the Belle Pass channel (Fig. 1) is necessary to allow vessels access to the Port. An expansion of the Port is planned to service deep water vessels, which will require channel deepening. The total dredged material from maintenance dredging and expansion is estimated to generate >15 million cubic meters (GIS Engineering, LLC 2018; U.S. Army Corps of Engineers, New Orleans District 2021) of sediment that is suitable for marsh and mangrove forest habitat creation.

Quantifying Net GHG Flux of Coastal Habitats

To quantify the potential net GHG flux of existing, converted, and restored tidal marsh and mangrove forest habitats in years 2020, 2025, 2030, and 2050 we developed a Coastal Wetlands Carbon Model that depended on two parts. The first part was a wetland vegetation species distribution model (LAVegMod.PF) that represents the mortality and establishment of dominant species, such as *S. patens* to represent brackish marsh habitat, *S. alterniflora* to represent saline marsh habitat, and *A. germinans* to represent mangrove forest habitat. The distribution of those species depends on input from a hydrodynamics and a morphology model (see Supplement S1). The second part of the Coastal Wetlands Carbon Model was a lookup table that included major carbon (CO2e) fluxes (aboveground net primary productivity [ANPP], sediment/soil carbon accumulation, and nitrous oxide and methane emissions) to estimate the net

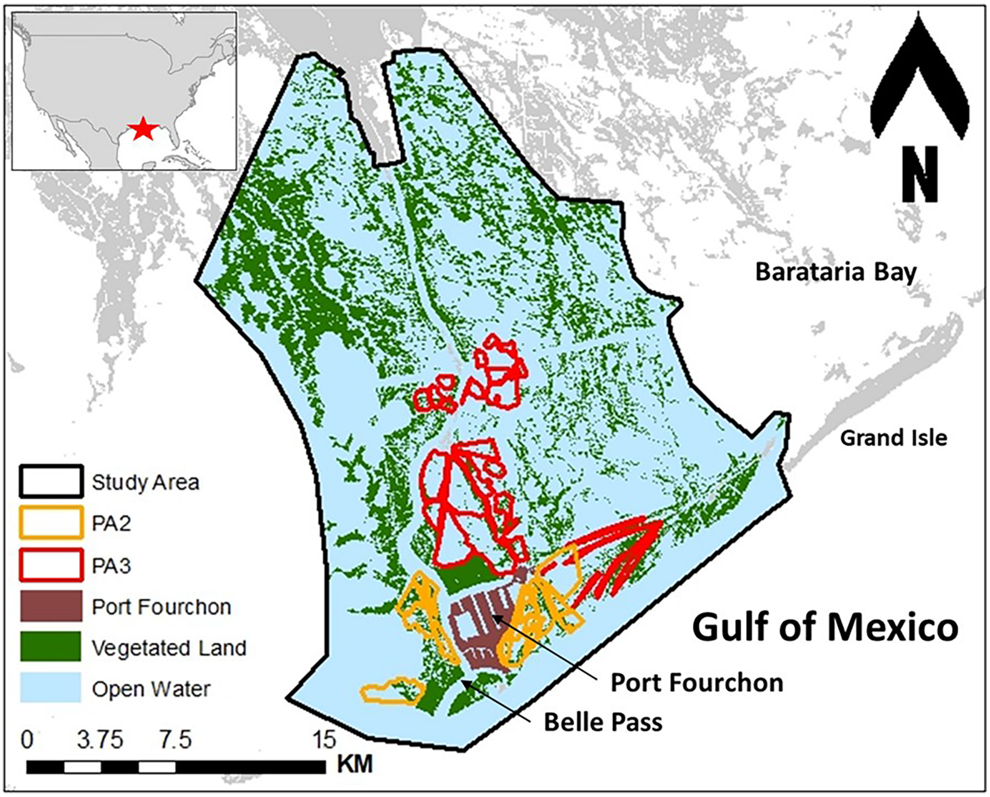

Figure 1. Map of study area including Port Fourchon, Louisiana, U.S.A. Orange and red polygons represent project areas of the modeled project alternatives (PA) from the placement of dredged material to create and protect mangrove forest (PA2) and marsh (PA3) habitats. Belle Pass is the navigation channel that tends to be the dredged material source.

ecosystem carbon balance (NECB) (Baustian et al. 2023) per coastal habitat (see Supplement S2).

Hydrodynamics and Morphology Models. The hydrodynamic model and the morphology model were developed from the Delft3D FM modeling suite (Deltares 2019) that includes flow of water (D-Flow) coupled with waves (D-Waves based on the SWAN model), and sediment transport and morphology (D-Morphology) with similar coupling as Baustian et al. (2018). The hydrodynamics model was calibrated for 2015 and 2016 based on water level, flow velocity, waves, and salinity. Water level and salinity are important variables that inform the wetland vegetation species distribution model. Water levels were calibrated by adjusting the offshore water-level boundary and changing the wind drag coefficient. Model skill was determined by comparing the modeled output to 2015 and 2016 observations of water levels in open water habitats and wetland gages for quiescent conditions, cold fronts, and tropical cyclones. Tidal amplitudes were also verified through comparison with gage observations. Overall, the 2015 and 2016 water-level statistics indicated an average bias of 0.0 and 0.0 m, root mean square error (RMSE) of 0.11 and 0.10 m, and correlation coefficient of 0.84 and 0.86, respectively. Salinity was calibrated by adjusting the horizontal eddy diffusivity coefficient and compared at U.S. Geological Survey (2022a,b,c) gages (291929089562600 Barataria Bay near Grand Terre Island, LA; 073802516 Barataria Pass at Grand Isle, LA; 07380251 Barataria Bay N of Grand Isle, LA) and Coastwide Reference and Monitoring System (CRMS, Coastal Protection and Restoration Authority of Louisiana, CPRA [2022]) open water gages (0178, 0181, 0224, 0292, 0338, 0341, 4,690) for 2015 and 2016. Overall performance for salinity in 2015 and 2016 indicated a bias of −0.8 and 0.10 ppt, RMSE of 3.9 and 3.4 ppt, and correlation coefficient of 0.68 and 0.58, respectively.

A calibrated model simulation was initiated with the morphology model that estimated changes in water bottom elevation and land elevation through interactions with waves, tides, winter meteorological front conditions, and tropical cyclones for a schematized period every 5 years from 2020 to 2050. The evolved morphology is produced every 5 years based on updates to the elevation of the tidal inlets, shoreface, shelf, bay floor, and marsh and mangrove forest habitats. The area of vegetated land (marsh and mangrove forest habitats) was adjusted to account for land loss due to edge erosion. The edge erosion was calculated by converting wave power density to edge retreat rate using the wetland edge erosion coefficients calculated for marsh habitats in Barataria Basin by Valentine and Mariotti (2019). Wave power density was obtained from D-Waves (Deltares 2013).

Total accretion of the vegetated land was set to be the same as RSLR so that the vegetated land could be maintained even under RSLR (Herbert et al. 2021). The amount of total accretion that a given cell received was adjusted based on its elevation. Cells with an elevation greater than 35 cm above local eustatic sea level at both the beginning and the end of a 5-year cycle received no total accretion during that period. They are assumed to be infrequently inundated and thus not influenced by hydrodynamic drivers. Cells with an elevation less than 30 cm above current eustatic sea level received the full component of total accretion equal to RSLR. Lastly, cells between 30 and 35 cm above eustatic sea level received a linearly prorated amount of total accretion.

The morphology model and the hydrodynamics model were coupled with the Coastal Wetlands Carbon Model to run three simulations. All three model simulations experienced the same median storm energy, eustatic SLR (i.e., 25 cm SLR by 2050) projected by the National Oceanic and Atmospheric Administration (NOAA) S07-RCP 4.5 (NOAA GFDL 2013), spatially varying shallow and deep subsidence rates, which were derived from the subsidence values from Louisiana's 2023 Coastal Master Plan (Fitzpatrick et al. 2021), and the landscape was allowed to vertically accrete to keep up with RSLR (Kirwan et al. 2016). More information about modeled future RSLR conditions can be found in Supplement S3. The first model simulation, with project alternative one (PA1), was a future without restoration action (i.e., no placement of dredged material). The second simulation included the placement of dredged material (about 18 million cubic meters) east and west of Port Fourchon (PA2) (Fig. 1). The third model run (PA3) included the placement of dredged material (about 40 million cubic meters) north of Port Fourchon.

The placement of all dredged material to restore marsh and mangrove forest habitats took place in model year 2020. It was assumed that construction of the restored habitats was also finished in year 2020. The restored habitats were built within the polygons and governed by the same model processes as the existing and converted habitats. The restored habitat areas (polygons) used to represent the project areas of PA2 and PA3 in the model were derived by local stakeholders and project scientists (Hemmerling et al. 2023). The initial elevation of the entire restored habitat area polygon (PA2 and PA3 project areas) was increased to 0.39 m NAVD88. This elevation represented the 5-year postconstruction elevation determined from settlement curves of nearby restoration project design reports (Ardaman & Associates, 2018a, 2018b, 2018c; GeoEngineers LLC 2018). The models were run over a 30-year period to provide output for the Coastal Wetlands Carbon Model at target years of 2020, 2025, 2030, and 2050.

Coastal Wetlands Carbon Model. *Wetland Vegetation Species Distribution.* Three main vegetation species were evaluated from output produced by the wetland vegetation species distribution model (the LAVegMod.PF), which was modified to consider the unstructured model grid from LAVegMod that was previously developed for tracking vegetation shifts across the entire coast of Louisiana given changing environmental conditions (Visser & Duke-Sylvester 2017). The LAVegMod.PF operates on the same grid as the hydrodynamics model (D-Flow). Within each grid cell, the percent coverage of each wetland vegetation species is tracked on an annual basis to estimate habitat area. The initial percent coverage in each cell was derived from a 2014 Land Use/Land Cover map with a 30 m resolution (Couvillion 2017). The percent coverage changes based on hydrodynamics model outputs of mean annual salinity and the standard deviation of water level. Each vegetation species has a set probability of mortality and probability of establishment for the given inputs that govern the percent coverage changes. The relationships that determine the probabilities were derived from observations of presence and absence of each vegetation species at 390 CRMS sites across coastal Louisiana (Visser et al. 2013). If the environmental conditions cannot support any of the vegetation species, the cell was set to bare ground. Additional details about the wetland vegetation species distribution model and how it was used to estimate habitat area are provided in Supplement S1.

Net Ecosystem Carbon Balance. To calculate the total net carbon benefit of the study area, we used the NECB flux approach based on Baustian et al. (2023). More information about this approach can also be found in Supplement S2. Tables with habitat areas and modeled total accretion values that were used in the NECB calculation can be found in Supplements S5 and S6. We assumed the major carbon flux that could likely change in the future is the sediment/soil carbon accumulation rate (dominated by autochthonous material) because the future conditions of RSLR will likely drive an accommodation space for the wetland vegetation plants to respond (Herbert et al. 2021) to keep up with SLR (Soper et al. 2019; Weston et al. 2023). Therefore, we used the simple linear regression (Equation 1) from Herbert et al. (2021) with an $R^2 = 0.8$ to estimate the sediment/soil carbon accumulation rate of model cells for future years based on RSLR (mm/year).

$$\text{Mean sediment/soil carbon accumulation rate}\ \left(\mathrm{g\,C\,m^{-2}\,yr^{-1}}\right) = 52.61 + 24.78 \times \text{RSLR}. \tag{1}$$

Negative values for ANPP and sediment/soil accumulation represent a potential carbon sink from atmosphere. Positive values for nitrous oxide and methane emissions indicate a potential source to the atmosphere. The mean NECB was estimated per coastal habitat and used in Equations 5–9 in Supplement S2. Uncertainty analysis was conducted for each of the carbon fluxes (ANPP, sed/soil accumulation rates and nitrous oxide and methane emissions) per coastal habitat based on standard error calculations, and those percent uncertainties were then used to estimate a combined uncertainty for lower and upper bounds of NECB estimations per coastal habitat (see appendix S7 in Baustian et al. 2023).

Existing, Restored, and Converted Coastal Habitats. Four coastal habitat types were modeled: brackish marsh, saline marsh, mangrove forest, and saline open water. Three land changes were also tracked and consist of existing, restored, and converted habitat. The changes in coastal habitat type were tracked by comparing grid cells sequentially between targeted years (2020 vs. 2025, 2025 vs. 2030, 2030 vs. 2050) to classify the habitats in the grid cell as existing, restored, or converted. Existing coastal habitats include those habitats in a grid cell that did not change between targeted years and include brackish marsh, saline marsh, mangrove forest, and saline open water. Restored coastal habitats are those habitats in a grid cell that changed because of restoration actions from the placement of dredged material to create marsh or mangrove forest habitats. Three types of restored coastal habitats were considered: saline open water to mangrove forest, saline open water to saline marsh, and saline open water to brackish marsh. It was assumed that if a coastal habitat remains the same over time (existing habitat) or becomes restored to a different coastal habitat (restored habitat), the carbon fluxes prescribed are those of the most recent coastal habitat. Converted habitats are those that change, not due to restoration or management actions, but due to changing environmental conditions such as SLR and storms. Thus, seven conversions were considered, which fall under three categories: (1) mangrove forest converted to marsh habitats, (2) marsh habitats converted to other vegetated habitats (including mangrove forest), and (3) vegetated habitats converted to saline open water. See Supplement S2 for details about the simple source and sink assumptions used when habitats convert to other habitats.

Net GHG flux. The term net GHG flux of emissions (MMT CO_2e) was used in the Louisiana's current GHG inventory (Dismukes 2021) where negative values for a specific year signify removal of GHGs and positive values indicate a source to the atmosphere for that year. The net GHG flux of the study area or restoration project area was determined by summing the flux of GHG emissions for existing, restored, and converted habitats (Baustian et al. 2023). The flux of GHG emissions was determined by the product of the NECB (tonne CO_2e ha^{-1} yr^{-1}) for each habitat and its corresponding habitat area (ha for a snapshot year) (Hopkinson 2018) produced from the wetland vegetation species distribution model (LAVegMod.PF) for all habitat changes (Table 1) and based on model years 2020, 2025, 2030, and 2050. Because the wetland vegetation species distribution model was run either every 5 or every 20 years, we were only able to resolve habitat changes on those timescales. For terms in the flux of GHG emissions calculation that account for conversion of area from one habitat to another (e.g., conversion of marsh to open water), we assumed a linear rate of change over the relevant time period in order to create a snapshot at a particular year. The net GHG flux for the total study area and at the restoration project area includes an upper and lower bound, which provides the uncertainty of estimated net GHG flux and was based on the NECB, $NECB_{Lower\ bound}$, and $NECB_{Upper\ bound}$ (Baustian et al. 2023).

Table 1. Coastal habitat conditions, types, and descriptions that were used in the calculation of net GHG flux for the total study area and at restoration project area scale.

Habitat Condition	*Number of Habitat Types*	*Habitat Type Descriptions*
Existing	4	Mangrove forest, saline marsh, brackish marsh, saline open water
Converted	7	Mangrove forested converted to saline marsh, or saline open water; saline marsh converted to mangrove forest, or saline open water; and brackish marsh converted to mangrove forest, saline marsh, or saline open water
Restored	3	Saline open water restored to mangrove forest, saline marsh, or brackish marsh

Results

NECB of Coastal Habitats

Mean values of ANPP, sediment/soil carbon accumulation, and methane and nitrous oxide emissions (see Supplement S2) of four coastal habitats were calculated to estimate the NECB per habitat (Table 2) as part of the Coastal Wetlands Carbon Model. The mean ANPP was highest for the brackish marsh and mangrove forest near -46.0 tonne CO_2e ha^{-1} yr^{-1}. All vegetated habitats were estimated to have the same varying sediment/soil carbon accumulation rates over 30 years to keep up with RSLR, and the overall mean was estimated at -9.7 ± 1.4 tonne CO2e ha^{-1} yr^{-1}. The brackish marsh habitat was estimated to have the highest methane and nitrous oxide emissions; however, methane and nitrous oxide emissions of all the habitats were relatively low compared to the ANPP and sediment/soil carbon accumulation rates (Table 2). By combining the three-carbon equivalent annual fluxes, an estimated NECB was calculated for each of the habitats. The largest NECB was for the mangrove forest habitat (-54.0 ± 26.1 tonne CO_2e ha^{-1} yr^{-1}), and the smallest was for the saline open water (-11.6 ± 1.0 tonne CO_2e ha^{-1} yr^{-1}) (Table 2).

Study Area—Coastal Habitats and Net GHG Fluxes

Most of the total study area (58,707 ha, Fig. 1) was dominated by saline marsh habitat (sum of existing, restored, and converted, 6,322–13,069 ha) from years 2020 to 2050 with some brackish marsh habitats present in the northeast corner and mangrove forest habitat in the southwest corner (Fig. 2). By adding dredged material in the polygons of PA2, additional saline marsh and mangrove forest habitat area was predicted. The placement of dredged material in the polygons of PA3 supported the creation of mainly saline marsh habitats. Vegetated habitat area of the study area decreased over time for all model runs (Fig. 3, top panels). For example, the vegetated habitat area decreased from 15,079 ha in 2020 to 7,307 ha in 2050 with no restoration (PA1) and from 16,196 to

Table 2. Lookup table that represents carbon (CO_2e) fluxes used to estimate the net ecosystem carbon balance of coastal habitats, including brackish marsh (*Spartina patens*, SPPA), saline marsh (*Spartina alterniflora*, SPAL), mangrove forest (*Avicennia germinans*, AVGE), and saline open water. The sediment/soil carbon accumulation rates estimated were variable based on RSLR but an estimated average rate was -9.7 ± 1.4 tonne CO_2e ha^{-1} yr^{-1} calculated from model output over 30 years based on Herbert et al. (2021). Details about the literature review can be found in Supplement S2. Methane (CH_4) and nitrous oxide (N_2O) emissions were converted to carbon dioxide equivalence (CO2e) by multiplying the fluxes by their specific global warming potential (over 100 years) of 25 and 298, respectively (see Supplement S2 for details).

	Carbon Flux (Mean ± 95% SE, tonne CO_2e ha^{-1} yr^{-1})				
Existing Coastal Habitat (With Identifier and Model Code)	*Aboveground Net Primary Productivity (ANPP)*	*Sediment/Soil (1 m) Carbon Accumulation*	*N_2O and CH_4 Emissions*	*Net Ecosystem Carbon Balance (tonne CO_2e ha^{-1} yr^{-1} ± Combined Uncertainty)*	*References*
Brackish marsh ($i = 1$, SPPA)	-46.5 ± 5.5	Variable, estimated using Herbert et al. (2021)	$+8.1 \pm 3.2$	-48.1 ± 21.0	Hopkinson et al. (1978), White et al. (1978), Hopkinson et al. (1980), Cramer et al. (1981), DeLaune et al. 1983, Smith et al. (1983), DeLaune et al. (1984), Delaune and Smith (1984), Sasser & Gosselink 1984, Feijtel et al. (1985), White and Simmons (1988), Pezeshki and DeLaune (1991), Nyman et al. (1995), Flynn et al. (1999), Cardoch et al. (2002), Day et al. (2013), Holm et al. (2016), Krauss et al. (2016), Lane et al. (2016), Stagg et al. (2016), Sasser et al. (2018), Herbert et al. (2021)
Saline marsh ($i = 2$, SPAL)	-29.4 ± 2.6	Variable, estimated using Herbert et al. (2021)	$+1.6 \pm 0.7$	-37.5 ± 17.6	Kirby and Gosselink (1976), Hopkinson et al. (1978), White et al. (1978), DeLaune et al. (1983), Smith et al. (1983), Sasser and Gosselink (1984), Feijtel et al. (1985), Kaswadji et al. (1990), Pezeshki and DeLaune (1991), Edwards and Mills (2005), Darby and Turner (2008), Stagg and Mendelssohn (2011), Day et al. (2013), Pham (2014), Lane et al. (2016), Stagg et al. (2016), Sasser et al. (2018), Herbert et al. (2021), Hopkinson et al. (1980)
Mangrove forest ($i = 3$, AVGE)	-45.9 ± 6.7	Variable, estimated using Herbert et al. (2021)	$+1.6 \pm 0.7$	-54.0 ± 26.1	Lugo and Snedaker (1974), DeLaune et al. (1983), Smith et al. (1983), Pham (2014), Lane et al. (2016), Weaver and Armitage (2020), Herbert et al. (2021)
Saline Open Water ($i = 4$)	-3.7	Variable, estimated using Herbert et al. (2021)	$+0.03$	-11.6 ± 1.0	Day (1973), DeLaune et al. (1983), Smith et al. (1983)

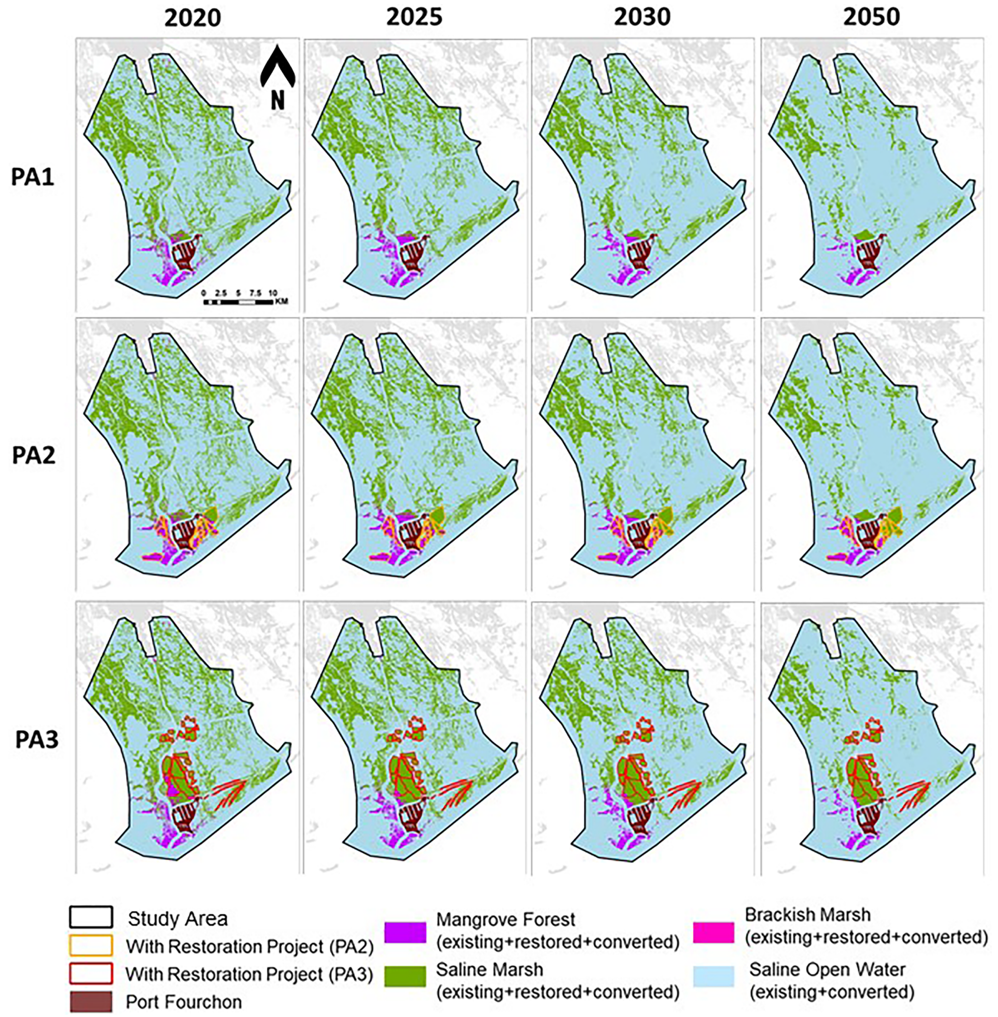

Figure 2. Study area timeseries of model-projected coastal habitat areas (brackish marsh, saline marsh, and mangrove forest, saline open water) that were categorized as existing, restored, or converted at years 2020, 2025, 2030, and 2050 with no action of restoration (PA1) and with restoration projects (PA2 and PA3) via the placement of dredged material.

8,607 ha (PA2) and 17,367 to 9,726 ha (PA3) with restoration for the same time periods. With the loss of vegetated habitats, the saline open water habitat area increased over time for all model runs.

The study area surrounding Port Fourchon, Louisiana was estimated to be a net GHG sink (from -1.1 ± 0.2 MMT CO_2e) at years 2020 and 2025 with or without restoration (bottom panels in Fig. 3). The study area for all model runs remains a net GHG sink in years 2030 and 2050, but due to the conversion of brackish and saline marsh to saline open water (red bars in Fig. 3) the amount of GHG stored each year was reduced by a factor of two. With restoration, a smaller area of brackish and saline marsh was converted to saline open water, as were the losses of GHG to the atmosphere. Thus, with restoration (PA2 and PA3), about $+0.1 \pm 0.004$ MMT CO_2e was estimated to be avoided as a source from the study area.

Restoration Projects—Coastal Habitats and Net GHG Fluxes

The total area of coastal habitats that received dredged material was near 1,500 ha for PA2 (dominated by mangrove forests) polygons and over 2,500 ha for PA3 (dominated by marsh) polygons (top panels of Fig. 4). Coastal habitats of mangrove forest, brackish marsh, and saline marsh were observed in both model simulations that included restoration projects (PA2 and PA3). Mangrove forest habitat area over time was greater in PA2 than PA3. The conversion to saline open water habitat was greatest at year 2050 in PA3 (dominated by marsh). However, over time both PA2 and PA3 polygon area experienced an increase in saline open water habitat (Fig. 4A & 4B). The restoration project areas were estimated to be a net GHG sink (near -0.1 ± 0.04 MMT CO_2e) at all years from 2020 to 2050 (Fig. 4C & 4D). Placement of dredged material in PA2 polygons created a high proportion of mangrove forest habitat that

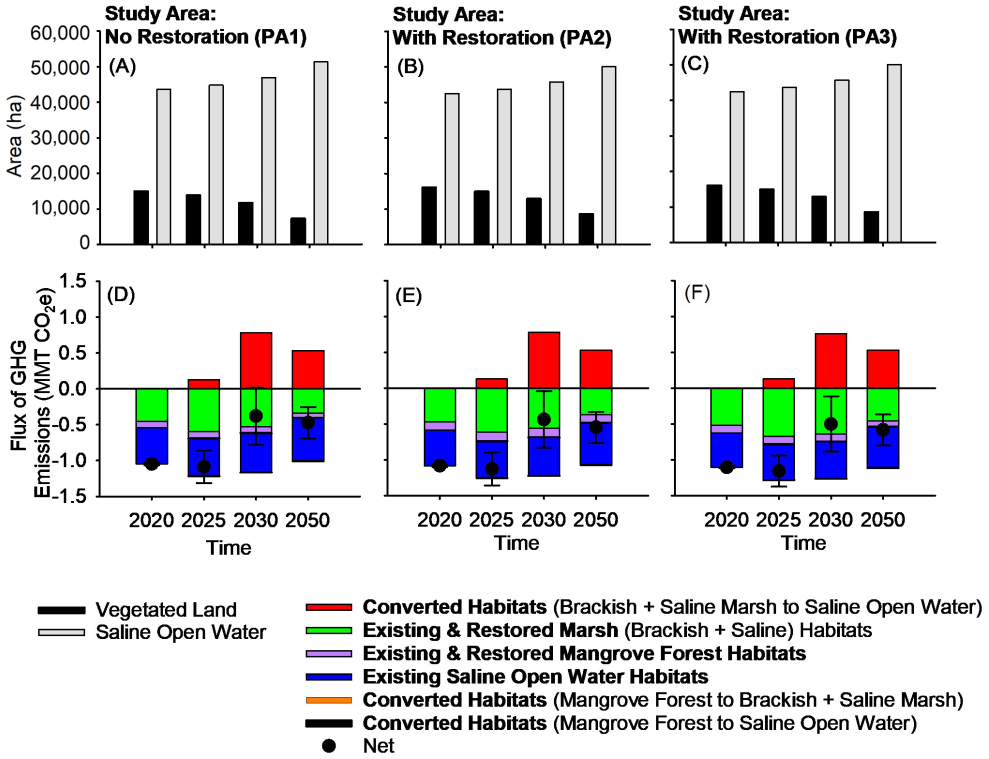

Figure 3. Study area of model-projected vegetated habitats (brackish marsh, saline marsh, and mangrove forest) that were existing, restored, or converted as well as saline open water habitat that was existing or converted from vegetated habitats (top panels of A–C) and flux of GHG emissions (MMT CO_2e) at years 2020, 2025, 2030, and 2050 with and without restoration by placement of dredged material (PA2, PA3; bottom panels of D–F). Stacked bars are in the order listed in the legend. The orange and black bars in panels D–F are present but small. Positive flux values indicate a source and negative values indicate a sink. Error bars of net flux indicate the upper and lower bound.

supported a consistent negative flux of GHG emissions (near -0.05 ± 0.04 MMT CO_2e, Fig. 4C, mangrove forest only) compared to PA3 (Fig. 4C & 4D). In addition, from 2020 to 2050, the area of PA2 was projected to remain a net GHG sink (Fig. 4C). The habitats of PA3 were dominated by brackish and saline marshes that were restored from saline open water habitats (green bars, Fig. 4B) and overtime had a high negative flux of GHG emissions (green bars, Fig. 4D), near -0.10 MMT CO_2e. The positive flux of GHG emissions from the conversion of brackish and saline marsh habitats to saline open water habitats within the PA2 and PA3 polygons was evident at years 2025, 2030, and 2050 (red bars in Fig. 4C & 4D). The sink terms (from restoration activities) were able to compensate for the loss, and the restoration project areas remained net GHG sinks (see Supplement S4).

Discussion

Projecting Sinks and Sources

Creating coastal marsh and mangrove forest habitats via the placement of dredged material could help mitigate global warming by restoring and protecting ecosystem functions of carbon fixation by aboveground biomass and carbon accumulation in sediment/soils, although see the following section *Reducing Uncertainties*. Including these human actions of protecting and restoring wetlands could also support better estimates of current and future coastal carbon budgets (Hopkinson 2018). In this modeling study, we estimated changes in coastal habitat area and corresponding net GHG fluxes over a 30-year period with and without restoration via the placement of dredged material. By modeling the coastal habitat area changes within the total study area, we found that between 2020 and 2050 the ecosystem could lose half of its capacity to be a net GHG sink with or without restoration (PA2 or PA3) due to the conversion of marsh habitats to saline open water habitats. However, at year 2050 with restoration projects (PA2 or PA3), about $+0.1 \pm$ MMT CO_2e could be avoided as a source to the atmosphere. Other ecosystem studies suggest coastal wetlands can be a net source of GHG emissions, such as Gulf Coast and U.S. mid-Atlantic wetlands because of soil losses (Holmquist et al. 2018) and SLR (Warnell et al. 2022). It is also important to consider alternative scenarios of wetland loss with coastal restoration activities because combined with the potential mitigating effect of restoration activities, the net GHG sink could be smaller.

Location of Restoration Projects

Not only are restoration projects predicted to decrease the source of GHG emissions from the total study area at year 2050, the

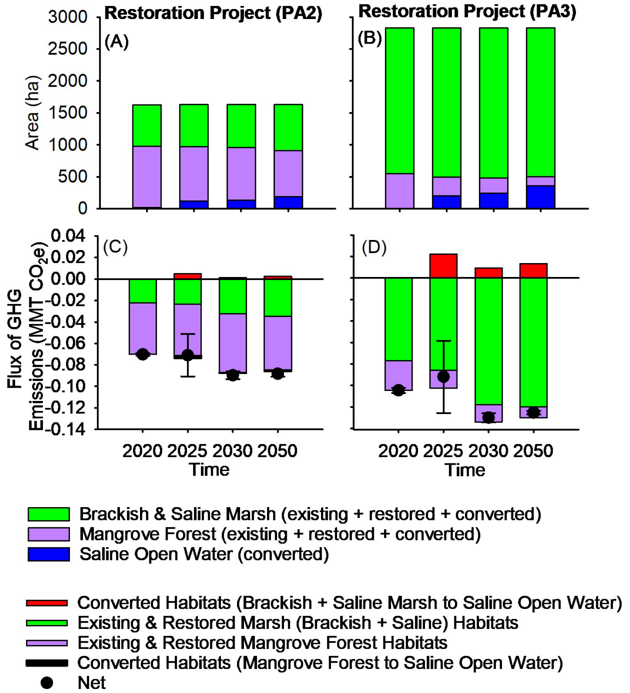

Figure 4. Model-projected restoration project areas (PA2 and PA3) from the placement of dredged material including the types of coastal habitats of saline open water, mangrove forest, and brackish and saline marsh (A, B) and the corresponding flux of GHG emissions (C, D). Stacked bars are in the order listed in the legend. Positive flux values indicate a source and negative values indicate a sink. Error bars of net flux indicate the upper and lower bound.

location and type of habitat restored can influence how much carbon is captured and remains buried. Two restoration project scenarios (PA2 with mangrove-dominated restoration and PA3 with marsh-dominated restoration) were modeled over time, and it was observed at the restoration project scale that even though PA2 mangrove restoration had about half the total habitat area (that required about half the total dredged material) of PA3 marsh restoration, the net GHG flux over time (−0.07 to −0.09 MMT CO_2e) were near levels of PA3 (−0.09 to −0.13 MMT CO_2e). The restoration project area of PA2 had relatively higher salinity than PA3 and allowed black mangroves to dominate and influence the net GHG flux estimates. Thus, expansion of mangrove forest areas in restored habitats could help increase carbon storage in above- and belowground pools (Simpson et al. 2019; Soper et al. 2019; Rogers 2021). Additional field observations in this region could improve estimates of carbon fluxes by assessing the black mangrove growth rate over time (Patterson et al. 1993) and utilizing remote sensing (Woltz et al. 2023) to assess potential changes in ANPP as young mangroves colonize and grow, but also how carbon fluxes might change when areas convert to open water or during freeze events that drive black mangrove mortality in this region (Osland et al. 2020).

Reducing Uncertainties

Estimating the future climate mitigation role of coastal habitats that are converting to saline open water remains challenging because of the unknown future responses of the carbon fluxes within the aquatic pool (Holmquist et al. 2018; Baustian et al. 2023; Webb et al. 2019). The total study area surrounding Port Fourchon, Louisiana has high salinity waters (>15 ppt), and the vegetated habitats are dominated by brackish and saline marshes as well as mangrove forests. These coastal habitats tend to produce low methane and nitrous oxide emissions (DeLaune et al. 1983; Smith et al. 1983; Poffenbarger et al. 2011). Thus, the GHGs emitted to the atmosphere from these coastal habitats during this modeling study were driven by simple assumptions that stored carbon in aboveground biomass and soils are lost immediately when wetland habitats convert to saline open water habitats (RAE 2017; U.S. EPA 2021; Sapkota & White 2021). Some of the carbon in the aboveground biomass and soils could be transported or deposited in channels, ponds, bays, and on the continental shelf (Troxler et al. 2013) or retained in open water ponds (Schoolmaster et al. 2022), thus remaining preserved and stored in the ecosystem, and not necessarily released to the atmosphere. Assumptions about carbon loss when habitats convert to open water could potentially impact the classification of an ecosystem as a net GHG source or sink (Pendleton et al. 2012). In addition, the GHG emissions produced from burning fossil fuels to dredge sediment and from the use of heavy construction equipment (Bates et al. 2015; Andreo-Martínez et al. 2021) may also be considered in future calculations of restoration activities.

Tracking coastal habitat areas and changes in GHG fluxes via annual field investigations combined with finer temporal resolution in ecosystem models may help improve net GHG flux estimates for future years. Annual model time steps could reduce net GHG flux uncertainty because a finer temporal resolution (e.g., <5–10 years) could allow for better representation of the restoration and conversion processes occurring. For example, it was simply assumed that the carbon fluxes were immediately restored at values of natural marshes in areas with placement of dredged material but soil carbon accumulation rates in created marshes can be lower (Kelsall et al. 2023) or higher than natural wetlands (Krauss et al. 2017). Vegetated habitats converting to open water is also likely a slow process and not immediate and the loss of carbon from the soil stock can take hundreds of years to stabilize (Schoolmaster et al. 2022). Between 2030 and 2050, the change in RSLR is expected to be about 0.18 m and thus wetland loss dynamics and corresponding GHG source emissions are projected to be high compared to the change in RSLR between the other targeted years (e.g., 2020 vs. 2025 or 2025 vs. 2030). Thus, incorporating these ecosystem processes at a finer temporal resolution in ecosystem models could improve net GHG flux estimates at future target years. The contribution of autochthonous and allochthonous carbon to the soil carbon accumulation rates (Krause et al. 2022) of natural and created wetlands could also help better understand if using the total soil carbon accumulation rates are overestimating the role of these habitats in sequestering carbon.

Informing GHG Inventories

Modeling future coastal habitat changes and corresponding net GHG fluxes are important as various states in the United States, including Louisiana, have climate strategies where natural lands account as an important GHG sink (Climate Initiatives Task Force 2022). These results provide a case study of what might be expected to change at years that align with GHG reduction target years. For example, at year 2025 when the State of Louisiana hopes to reduce to 26–28% of the 2005 GHG levels, these coastal habitats in study area are projected to remove carbon from the atmosphere and could help reach that goal. At year 2050, when the target is to be net zero, marsh and mangrove forest habitats in the study area are projected to experience high edge erosion due to RSLR that drives the conversion of marsh and mangrove forest habitats to saline open water habitats producing a smaller net GHG sink. These results may not only inform restoration managers, local policy analysts, and other stakeholders (Hemmerling et al. 2023) about where and when to create marsh and mangrove forest habitats to help reach GHG reduction targets, but also, how projected emissions from coastal habitat conversion to saline open water could be reducing the role of coastal habitats in mitigating climate change at year 2050 (Pendleton et al. 2013).

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Supporting Information

The following information may be found in the online version of this article:

Supplementary S1. Coastal wetlands carbon model development.
Supplementary S2. Net ecosystem carbon balance, flux of GHG emissions, and net GHG flux.
Supplementary S3. Modeling future environmental conditions.
Supplementary S4. Future research opportunities to reduce uncertainties about climate change mitigation potential.
Supplementary S5. Modeled habitat areas of the total study area.
Supplementary S6. Modeled spatially averaged total accretion.

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