



Impact of different restoration methods on coastal wetland loss in Louisiana: Bayesian analysis

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Abstract Extensive efforts have been undertaken in the northern Gulf of Mexico to restore coastal wetlands that have been lost rapidly. The evaluation of these restorations mostly focused on individual-project scales. A modeling framework that can coherently synthesize multi-scale monitoring data and account for various uncertainties would improve quantitative evaluations at broader spatial scales needed for regional decision-making. We aim to develop such a framework to investigate the impact of different restoration methods (hydrological alteration, breakwater infrastructure, vegetative planting, or marsh creation using dredged materials) on wetland loss on the outermost mainland coastlines in Louisiana. We did this by implementing multi-level Bayesian models to predict areal wetland loss (1996–2005 before Hurricane Katrina) as a function of local geophysical variables (relative sea-level rise, wave height, tidal range) and a dummy variable indicating presence/absence of restoration. We assumed the effects of these variables varied by broader watershed scales. The restoration's effect also depended on temporal scales of implementation. The results indicate the sites with hydrological alteration, when implemented for longer than 7 years, had significantly smaller areal wetland loss, compared to the reference sites controlled for the local geophysical variables, in the Chenier Plain

watershed, but not in the lower Mississippi River watershed. The effects of the other restoration methods on wetland loss were not significant based on limited numbers of sites. The Bayesian modeling framework we developed can integrate monitoring data/key drivers across projects with uncertainties accounted for, it is adaptable, and presents a useful tool in restoration evaluations spatially and temporally.

Keywords Coastal wetland restoration · Relative sea-level rise · Hydrological alteration · Beneficial use · Breakwater structure · Vegetative planting

Introduction

Coastal wetlands provide important ecosystem services such as habitat for wildlife, food production, water quality improvement, flood protection, storm surge reduction, etc. (Costanza et al. 2008; Engle 2011; Yoskowitz et al. 2015). However, the coastal wetland ecosystems in the northern Gulf of Mexico (NGOM) have been disappearing quickly within the past few decades due to a variety of river-borne and marine-borne stressors, including river engineering, accelerated sea-level rise, and locally extreme subsidence rates of up to 6 mm/year (Karegar et al. 2015; Turner 1997). Subsidence in the NGOM is driven by geological faults (O'Leary and Gottardi 2020), fossil fuel extraction activities (Mallman and Zolback 2007), sediment compaction of poorly packed Holocene deposits (Coleman et al. 1998), and reduced sediment input (Morton et al. 2006;

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Yuill et al. 2009). It is positively related to land loss (Kolker et al. 2011). High subsidence rate leads to high relative sea-level rise (RSLR) and extended inundation (Couvillion and Beck 2013). Historically, allocthonous sediment contributions from the Mississippi River and other rivers in the NGOM had formerly played an important role in offsetting wetland loss due to subsidence (Coleman et al. 1998). However, the allocthonous sediment contributions have decreased since the 1950s due to damming on upstream rivers and diversion of freshwater inputs for oyster farming (Meade and Moody 2010). Other factors contribute to wetland loss, such as wave action (Fagherazzi et al. 2013) and oil spills (McClenachan et al. 2013; Turner et al. 2016). Additionally, tidal range could lower coastal wetland loss as larger tidal range could deposit more ocean-borne sediment onto coastal wetlands (Kirwan and Guntenspergen 2010). In general, there exists large spatial variability in wetland loss due to a variety of biological and geomorphic variables (Gutierrez et al. 2011a; Hardy et al. 2020 in review; Spencer et al. 2016) in addition to the influence of sea-level rise (Linhoss et al. 2015; Schile et al. 2014; Spencer et al. 2016; Warren and Niering 1993; Wu et al. 2015, 2020). These factors act at multiple spatial scales, from local to watershed (Braswell and Heffernan 2019; Hardy et al. 2020 in review). The coastal wetlands that are located in the same watersheds receive more similar riverine inputs (such as freshwater and sediments) and may respond similarly to local drivers compared to those in different watersheds (Hardy et al. 2020 in review).

Massive restoration efforts have been conducted to restore coastal wetlands in the NGOM and many other locations in the world. The most notable restoration methods involve passive and active methods. Passive methods include alteration of sediment supplies such as by freshwater diversion, and reduction of wave action by breakwater structures. Active procedures include direct construction of wetlands by vegetative planting or dumping dredged sediments.

Sediment dynamic restoration methods create or maintain wetland by supplying sufficient sediment to counteract sediment loss and sea-level rise directly or indirectly (Allison and Meselhe 2010). One important method is the beneficial use (BU) of dredged materials—that is deposition of sediments dredged from open water or riverine habitats to create or restore wetlands (Howard et al. 2020; Mchergui et al. 2014). This increases sediment availability immediately on

the restored sites. BU's goal is to increase the soil surface elevation to appropriate ranges to facilitate vegetation colonization and growth (Howard et al. 2020). Site properties (e.g., soil properties, morphology, hydrodynamics) and availability of seeds and propagules influence the pace of establishment and growth of restored marsh plant communities.

The diversion of silt-laden fresh waters is another common way to increase sediment supplies to coastal wetlands (Allison and Meselhe 2010; Couvillion et al. 2013; Kearney et al. 2011; Kenney et al. 2013). This approach is slow compared to enhancing the sediment supply from dredging but can be continuous over a long time. Freshwater diversion has had various results in wetland restoration. Some studies, including the Louisiana Coastal Master Plan (Coastal Protection and Restoration Authority of Louisiana 2012), have noted significant wetland gain that is proportional to the amount of water and sediment diverted (Day et al. 2016; Kenney et al. 2013). Other studies observed deleterious effects of long-term freshwater diversion onto the marshes (Kearney et al. 2011) or lack of wetland gains (Turner et al. 2019). Diversion could make coastal wetlands vulnerable to storm damage by reducing below-ground biomass due to nutrient input instead of promoting wetland gain. However, a recent review shows that the negative impact of increased nutrients could be offset by the increased sediment input (Elsey-Quirk et al. 2019).

Restoration is also done by planting habitat-forming species, especially in areas where natural recolonization is rare or limited by poor seed dispersal (Silliman et al. 2015; Zedler and Kercher 2005). The vegetation serves multiple functions in the maintenance of coastal wetlands. Vegetation reduces wave height and velocity by disrupting water flow (Tsihrintzis and Madiedo 2000) and reducing mechanical stress on the wetland sediments. Reduced wave action is positively related with vegetation height and density both in salt marshes (Möller 2006) and in mangroves (Parvathy and Bhaskaran 2017). Wave energy reduction due to vegetation also promotes sedimentation (Reed et al. 1999), allows more sediment particles in water columns to settle on the wetland surface, and reduces net erosion (Allen and Duffy 1998; Morris et al. 2002; Wu et al. 2017, 2020). Furthermore, the rate of settling is positively related with vegetation height and density. Vegetation directly contributes to sediment accretion via organic matter supplementation by root production

(Zedler 2000; Zedler and Kercher 2005). Belowground biomass also promotes soil cohesion via mycorrhizae and bulk biomass (Feagin et al. 2015). Soil cohesion and root stabilization thus prevent wave action from eroding wetland beds. Combined, the multiple functions of wetland vegetation help stabilize coastal wetlands (Nyman et al. 2006).

Breakwaters are a common restoration feature along coastlines, with a goal of reducing mechanical wave erosion on shorelines (Boumans et al. 1997), promoting sedimentation (Birben et al. 2007), and protecting shorelines from flash floods or overwash events (Vona et al. 2020). Offshore breakwaters are structures or collections of debris placed offshore that dissipate wave energy before it reaches the shore (Losada et al. 2008; Ryu et al. 2016), with up to 90% wave energy reduction (Armbruster 1999). This restoration is similar to natural wave attenuation exhibited by marsh vegetation, mangroves, or reefs (Parvathy and Bhaskaran 2017). Shear stress and the volume of sediment deposited into the marsh behind the breakwater increases between 20 and 40% on average, in proportion to the slope and distance of the breakwater from the shoreline (Vona et al. 2020). If carefully designed, breakwaters can also function as artificial reefs (Coen et al. 2007) for oysters or coral communities (Burt et al. 2010). Breakwater restoration projects in the NGOM have ranged from extremely successful (Holly Beach, LA; Edwards and Namikas 2011) to mixed (Raccoon Island, LA; Armbruster 1999).

Studies show that the efficiency of these restoration efforts is strongly site dependent (Kenney et al. 2013). Thus, design and evaluation of the restoration outcomes should account for site-specific dynamics and spatial variability. Different metrics have been developed to evaluate the success of restoration projects on tidal wetlands, including hydrology, soil and sediments, vegetation, nekton, and bird (Neckles et al. 2002; Zhao et al. 2016). Vegetation is most commonly used. With the development of GIS and remote sensing, dynamics of wetland area have been investigated to evaluate restoration at broader spatial scales though the research at the broad scale is limited (Rozas et al. 2007).

While evaluating the performance of these restoration practices at the individual-project scales is important to guide more effective restoration practices in the future (Couvillion et al. 2013), the information obtained from the evaluation has limitation to apply to other sites with a different geomorphic and physical setting. There is a need to synthesize the data from different projects/

sites in order to conduct evaluation at a broad spatial scale required for regional decision making. Coordinating monitoring activities among the sites is necessary to facilitate the quantitative synthesis. In addition, a modeling framework is required that can integrate data from different sources and scales and accommodate variances. Hierarchical Bayesian (HB) models show a useful tool in developing such a modeling framework to evaluate the effects of restoration on coastal wetland loss. HB models have advantages of decomposing a complex question into manageable components and can account for uncertainties from data, models, and parameters (Clark 2005). The assimilation and estimation of uncertainties are important in effective resource management considering large variability involved, but these are largely lacking or not comprehensively evaluated in the previous studies. Bayesian inference facilitates quantification of uncertainties using credible intervals from the resulting posteriors. In this study, we aim to develop hierarchical Bayesian models to evaluate the effect of different restoration methods on coastal wetland loss with uncertainties accounted for.

Methods

We implemented multi-level Bayesian models to predict wetland loss (1996–2005 before Hurricane Katrina) at 322 sites as a function of local geophysical variables, watersheds where they are located, and a dummy variable to indicate presence or absence of a particular restoration method. The outcome of the restoration projects was evaluated on the reduction of wetland loss at the respective projects' locations, derived from wetland maps. Reduction of wetland loss is an important benefit, particularly in the areas that have experienced large wetland loss. Through the models, particularly the posteriors for the dummy variables indicating presence/absence of restorations, we evaluated the impact of the restorations on coastal wetland loss compared to the reference sites where there were no restoration practices in the same watershed controlled for the local geophysical variables.

Local-scale variables included relative sea-level rise, wave height, and tidal range. We modeled the effect of these variables on coastal wetland loss to vary by watershed, a broader spatial scale. Similarly, we varied the effect of restoration by watershed, and whether the restoration had been implemented for short (≤ 10 years)

or long term (> 10 years) by 2005. We used 10 years as a cutoff following the design of a study on lake restoration (Søndergaard et al. 2007). We focused on restoration sites that had only one restoration method as it was difficult to separate the effect of individual restoration methods if multiple existed. We did not intend to evaluate the interactive effects of multiple restoration methods because there were limited sites where multiple methods were applied. We intentionally selected before Hurricane Katrina in 2005 because major hurricane events could complicate wetland loss patterns and evaluation of restoration projects. Between 1996 and 2005 before Katrina, Louisiana only experienced three category 1 hurricanes (https://en.wikipedia.org/wiki/List_of_United_States_hurricanes#Louisiana).

Study area and data

Our study area is the Louisiana Gulf Coast in the USA (Fig. 1). We focused on wetlands on the outermost mainland coastlines—the areas particularly vulnerable to relative sea-level rise. Marine-related variables were considered, including relative sea-level rise, wave height, and tidal range, because of the marine influence on these sites.

The spatial data for the restoration projects came from the Louisiana Coastal Protection and Restoration Authority (CPRA; <https://cims.coastal.louisiana.gov/Viewer/>). The data list included name of restoration project, construction status, year of construction, and project type. We limited restoration projects to those with construction completed prior to 2005. Information of construction years for some projects was missing, but we were able to derive it from the 2017 Louisiana Coastal Master Plan Attachment A1 (McMann et al. 2017). This data was retrieved via searching projects on the CPRA Website, replacing <projId> with the corresponding project ID in the following URL: <https://cims.coastal.louisiana.gov/outreach/ProjectView.aspx?projID=<projId>>. Project type in the original dataset was defined by the goal of the restoration project, and not necessarily the method. The four restoration method types we examined were “hydrological alteration” (HA), “marsh creation” (MC), “breakwaters” (BW), and “vegetative planting” (VP). We recoded vague and/or goal-oriented project types (“Barrier Island/Headland Restoration” and “Shoreline Protection”) to the appropriate restoration methods based on the project descriptions from the CPRA Website. Project

types “hydrological restoration” and “sediment diversion” were aggregated as the restoration method of “hydrological alteration”. “Infrastructure” restoration types were recoded as either breakwaters for off-shore constructions, or hydrological alteration for on-land constructions (e.g., plugs or levees). Wetland restoration projects implementing beneficial use of dredged materials were recoded as “marsh creation”, while the projects that involved vegetation planting were coded as “vegetative planting”. Though the cost-benefits of these CPRA projects were evaluated before approval (Merino et al. 2011), it is necessary to monitor these projects empirically to assess whether the predicted benefits were realized. We hypothesized that the effect of the restoration projects on coastal wetland loss varied by watershed and how long they had been implemented, or temporal lags since implementation (Braswell and Heffernan 2019; Hardy et al. 2020 in review; Huang et al. 2016).

To represent the watershed boundaries, we applied watershed boundaries from Hydrological Unit Code-4 (HUC-4) of US Geological Survey (<https://www.usgs.gov/core-science-systems/ngp/national-hydrography/watershed-boundary-dataset>). There are two HUC-4 watersheds in Louisiana: lower Mississippi River watershed on the east, and Chenier Plain watershed on the west. Based on how long the restoration projects had been implemented or temporal lags since implementation by 2005, we divided the projects into short term (≤ 10 years) and long term (> 10 years) (Søndergaard et al. 2007).

The data on coastal wetland loss came from the NOAA coastal change analysis program (C-CAP) (spatial resolution of 30 m, <https://coast.noaa.gov/digitalcoast/data/ccapregional.html>) The physical and geomorphic variables came from the USGS’ THK99 dataset (Hardy et al. 2020 in review) (<https://pubs.usgs.gov/dds/dds68/htmldocs/data.htm>). The THK99 (Gutierrez et al. 2011b; Thieler and Hammar-Klose 1999) is a spatially referenced dataset consisting of long-term averages of RSLR rate, tidal range, wave height, and coastal slope at regular ~ 5 km intervals along the outmost shorelines of the contiguous USA and Alaska. The purpose of the data is to evaluate coastal vulnerability to relative RSLR (<https://pubs.usgs.gov/dds/dds68/data/gulf/gulf.htm>). We created a circular buffer with a radius of 2.5 km around the centroid for each polyline segment on the Louisiana’s coastlines from the THK99 dataset and used the buffer as our study site (Hardy et al. 2020 in review).

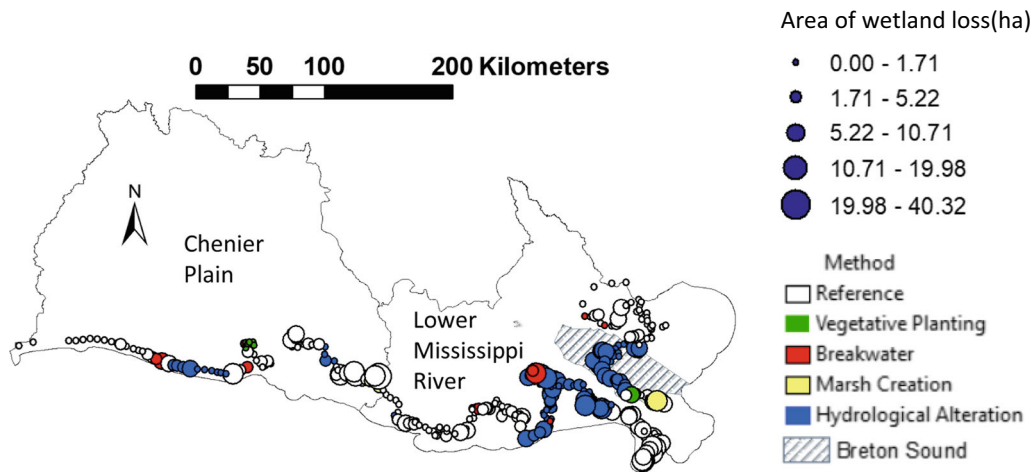


Fig. 1 The study sites on the Louisiana Gulf Coast with two HUC-4 watersheds where the sites are located: lower Mississippi River watershed on the east and Chenier Plain watershed on the

west. The map also shows restoration sites, the reference sites without restoration practices, and the areal wetland loss at each site (represented by size of circles)

We chose 2.5 km to maximize spatial coverage while minimizing overlaps of buffered areas. We extracted the geophysical properties at the centroid of each buffer area to represent the properties for that buffer. We removed the buffers that cover two watersheds. We expect the covariates’ impact on wetland loss to vary by watershed.

Model

To evaluate the effect of restoration projects on wetland loss, we started with an earlier model of areal wetland loss for the entire northern Gulf of Mexico that did not include the restoration sites (Hardy et al. 2020 in review). The areal loss model was a multi-level Bayesian model that included the covariates of relative sea-level rise, tidal range, and wave height, the effects of which on wetland loss varied by watershed. We added a predictive dummy variable that indicated the presence or absence of restoration, and we assumed its effect on wetland loss varied by watershed, how long or how long ago the restoration had been/was implemented. We developed models for each of the different restoration methods, and evaluated the effect of each method on coastal wetland loss separately (Fig. 2).

To represent the function of areal wetland loss (W) at the site j in the watershed i (W_{ij}), let $W_{ij} \cdot \mu$ represent the mean of W_{ij} and σ_S^2 represent the variance of wetland loss among the sites. W_{ij} was modeled by assuming it was distributed as (\sim) a normal distribution (N) (Eq. 1):

$$W_{ij} \sim N(W_{ij} \cdot \mu, \sigma_S^2) \tag{1}$$

We modeled the mean of wetland loss ($W_{ij} \cdot \mu$) as a function of the geophysical covariates and a dummy variable that indicated the presence or absence of a particular restoration method. The geophysical variables included relative sea-level rise (RSLR), wave height (WH), and tidal range (TR), the effects of which on coastal wetland loss varied by watershed. The intercept also varied by watershed (Hardy et al. 2020 in review). We assumed the effect of the restoration practice on coastal wetland loss varied by watershed, how long the restoration had been implemented, or the temporal lags since implementation by 2005. The linear function was described using Eq. (2) (for hydrological alteration HA method as an example):

$$W_{ij} \cdot \mu = f(\beta_{i,0}, \beta_{i,1}, \beta_{i,2}, \beta_{i,3}, \beta_{ki}) = \beta_{i,0} + \beta_{i,1}RSLR_{ij} + \beta_{i,2}WH_{ij} + \beta_{i,3}TR_{ij} + \beta_{ki}HA \tag{2}$$

where β_s represent the intercept and coefficients for the covariates in the multilevel model. $\beta_{i,0} - \beta_{i,3}$ represent the coefficients for RSLR, WH, and TR, respectively, the effect of which on wetland loss varied by watersheds (i). β_{ki} represents the coefficient for the dummy variable HA that indicates the presence (1) or absence (0) of restoration practices (here hydrological alteration shown in Fig. 2), the effect of which on wetland loss varied by watershed (i) and how long the restoration had been implemented or the temporal lags since implementation by 2005 (k) (k of 0 indicating no restoration, 1 indicating short-term restoration projects, and 2 indicating long-term restoration projects). Therefore, wetland loss (W) for the two watersheds (i) and S_i sites at each watershed i with no, short- or long-

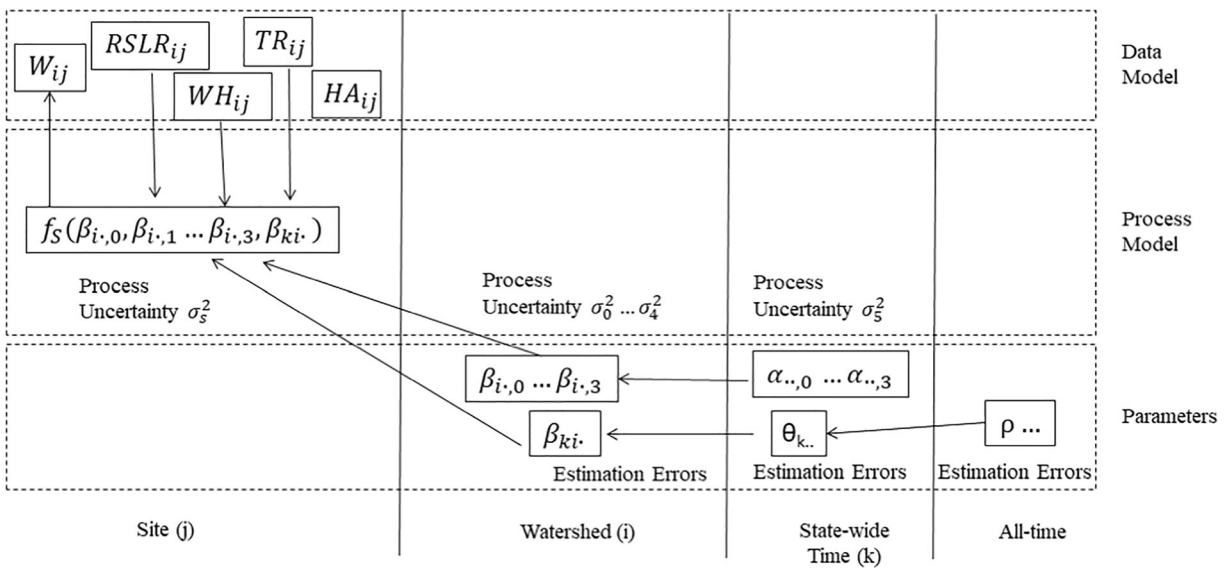


Fig. 2 Structure of the multi-level hierarchical Bayesian model used to identify the efficiency of different restoration methods on wetland loss. β_s represent the parameters associated with intercept and normalized covariates including relative sea-level rise (RSLR), wave height (WH), and tidal range (TR), as well as the dummy variable HA to indicate the presence or absence of restoration activities, here hydrological alteration (HA). α_s and θ

represent the statewide-scale hyper-parameters from which watershed-scale β_s were sampled. ρ represents the all-time and statewide-scale hyper-parameter from which θ was sampled. W represents logarithm of wetland loss, k represents temporal scales ($0 \leq 10$ or > 10 years), i represents watersheds, and j represents sites

term implementation of restoration projects, was modeled as in Eq. (3):

$$p(W|\beta_{i,0}, \beta_{i,1}, \beta_{i,2}, \beta_{i,3}, \beta_{ki}) \propto \prod_{k=0}^2 \prod_{j=1}^S N(W_{ij} | f(\beta_{i,0}, \beta_{i,1}, \beta_{i,2}, \beta_{i,3}, \beta_{ki}), \sigma_s^2) \quad (3)$$

The coefficients for the covariates that varied by watershed and temporal length of the projects ($\beta_{i,0}, \dots, \beta_{i,3}, \beta_{ki}$) were sampled from the parameters at the coarser spatial and temporal scales (Fig. 2) using normal distributions (Eq. 4):

$$\begin{aligned} \beta_{i,0} &\sim N(\alpha_{.,1}, \sigma_0^2) \\ &\dots \\ \beta_{i,3} &\sim N(\alpha_{.,3}, \sigma_3^2) \\ \beta_{ki} &\sim N(\theta_{k..}, \sigma_4^2) \\ \theta_{k..} &\sim N(\rho_{...}, \sigma_5^2) \end{aligned} \quad (4)$$

where α_s and $\theta_{k..}$ represent the means of intercept and coefficients for the local geomorphological variables and restoration variable at the state-wide scale, and $\sigma_0^2 - \sigma_4^2$ represent the variances between watersheds for these parameters. $\rho_{...}$ and σ_5^2 represent the mean and variance of $\theta_{k..}$ at the temporal scale that includes 0 year, short, and long terms.

To complete the Bayesian model, we defined prior distributions for unknown parameters (α_s , ρ , and σ_s^2). We used conjugate priors for computation efficiency (Calder et al. 2003) therefore the priors and posteriors had the same probability distribution forms. The priors for α_s and ρ were normally distributed, and the prior for σ_s^2 s followed the inverse gamma distribution (IG). The prior distributions were flat and only weakly influenced the posteriors, which reflected the lack of knowledge on these parameters (Lambert et al. 2005).

By combining the parameter (priors), process, and data models, we derived the joint distribution in Eq. (5) (Clark 2005).

$$p \left(\begin{matrix} \beta_{i,0}, \beta_{i,1}, \dots, \beta_{i,3}, \\ \beta_{ki}, \\ \alpha_{.,0}, \dots, \alpha_{.,3}, \\ \theta_{k..}, \rho_{...}, \\ \sigma_s^2, \sigma_0^2, \dots, \sigma_5^2 \end{matrix} \middle| WL, RSLR, WH, TR, HA \right) \propto N \left(WL \middle| \begin{matrix} \beta_{i,0}, \beta_{i,1}, \dots, \beta_{i,3}, \\ \beta_{ki}, \\ \sigma_s^2 \end{matrix} \right) \times N(\beta_{i,0} | \alpha_{.,0}, \sigma_0^2) \times N(\beta_{i,1} | \alpha_{.,1}, \sigma_1^2) \dots N(\beta_{i,3} | \alpha_{.,3}, \sigma_3^2) \times N(\beta_{ki} | \theta_{k..}, \sigma_4^2) \times N(\theta_{k..} | \rho_{...}, \sigma_5^2) \times N(\alpha_{.,0}) \times N(\alpha_{.,1}) \dots N(\alpha_{.,3}) \times N(\rho_{...}) \times IG(\sigma_s^2) \times IG(\sigma_0^2) \times IG(\sigma_1^2) \dots \times IG(\sigma_5^2) \quad (5)$$

The model in Eq. (5) was implemented using Markov Chain Monte Carlo (MCMC) simulations (Gelfand and

Smith 1990) in JAGS (Plummer 2016) through the CRAN R (R Core Team 2015) package “rjags.” The local geomorphological variables were normalized (Zar 2010), which allowed comparison of each of the covariates’ impact on wetland loss based on the magnitudes of their coefficients because they were transformed to be of the same range (Hooten and Hobbs 2015). The response variable was log transformed, to fit a normal distribution and account for nonlinearity and non-additive effect of the geomorphological variables and the dummy variable for restoration. In order to evaluate model convergence, we simulated three MCMC chains that used three different sets of initial values for the parameters. We examined the three chains to determine the burn-in (number of iterations of MCMC before they converged to posteriors) (50,000). We then discarded the pre-convergence burn-in iterations and ran the chains 300,000 more iterations, thinning every 10 steps to reduce within-chain autocorrelation. The simulated posteriors helped identify the magnitude of effects of restoration projects, along with the geomorphological variables, on wetland loss.

When we described the results, we used two terms commonly used in Bayesian analysis: medians and credible intervals. The medians of the posteriors describe the central tendency of the corresponding parameters, while credible intervals described intervals within which the parameters fall with a particular probability.

Results

In the lower Mississippi River watershed, coastal wetland loss increased significantly with relative sea-level rise and wave height, while it significantly decreased with tidal range. In the Chenier Plain watershed, the same variables had no significant effect on coastal wetland loss (Fig. 3).

The effect of restoration on coastal wetland loss varied by method, watershed, and how long it had been implemented or the temporal lags since implementation by 2005. In general, the restorations showed more effective in reducing coastal wetland loss in the Chenier Plain watershed than in the lower Mississippi River watershed.

The majority of the restoration projects used hydrological alteration (92 sites). Though the areal wetland loss was not significantly lower in the short-term restoration sites using hydrological alteration compared to the reference sites, the long-term restoration sites showed significantly lower wetland loss in the Chenier

Plain watershed (Fig. 3a). This result showed that hydrological alteration was effective in reducing coastal wetland loss when implemented more than 10 years in this watershed. This is a substantial finding because these restoration sites likely experienced more wetland loss than the reference sites in the past, justifying the restoration efforts. By comparison, no significant reduction in wetland loss was found in the lower Mississippi River watershed.

Due to the small number of sites that implemented breakwaters, vegetative planting, and marsh creation, we need to interpret the results related to these three restoration methods with caution. At the eleven breakwater restoration sites, wetland loss was not significantly lower than the reference sites in both watersheds, whether they were short or long term (Fig. 3b). However, the medians of the restoration effects were negative and of a similar magnitude at the long-term breakwater sites, showing less wetland loss when the method was implemented for more than 10 years in both watersheds on average based on the limited data. In addition, long-term breakwaters seemed to be more effective in reducing coastal wetland loss than hydrological alteration in the lower Mississippi River watershed as the credible interval of the parameter for long-term restoration shifted to left in breakwater scenarios compared to hydrological alteration (Fig. 3a, b).

At the sites with vegetative planting restoration (only 5 sites), coastal wetland loss did not show significant reduction compared to the reference sites (Fig. 3c). However, the medians of the restoration effects on coastal wetland loss were negative at both short- and long-term sites in the Chenier Plain watershed. Meanwhile, the median of the short-term restoration effect was positive in the lower Mississippi River watershed. This suggests that vegetative planting, when implemented in the short term, was more effective in the Chenier Plain watershed than in the lower Mississippi River watershed on average based on limited data. It is worth noting that we did not have long-term restoration sites using vegetative planting in the lower Mississippi River watershed.

Marsh restoration using dredged sediments did not lead to significant decrease of wetland loss beyond 10 years compared to the reference sites based on the limited data (only 8 sites, all longer than 10 years) (Fig. 3d). The median of the long-term restoration effect was negative in the Chenier Plain watershed and positive in the lower Mississippi River watershed, indicating this method was more effective in the Chenier Plain watershed on average based on limited data.

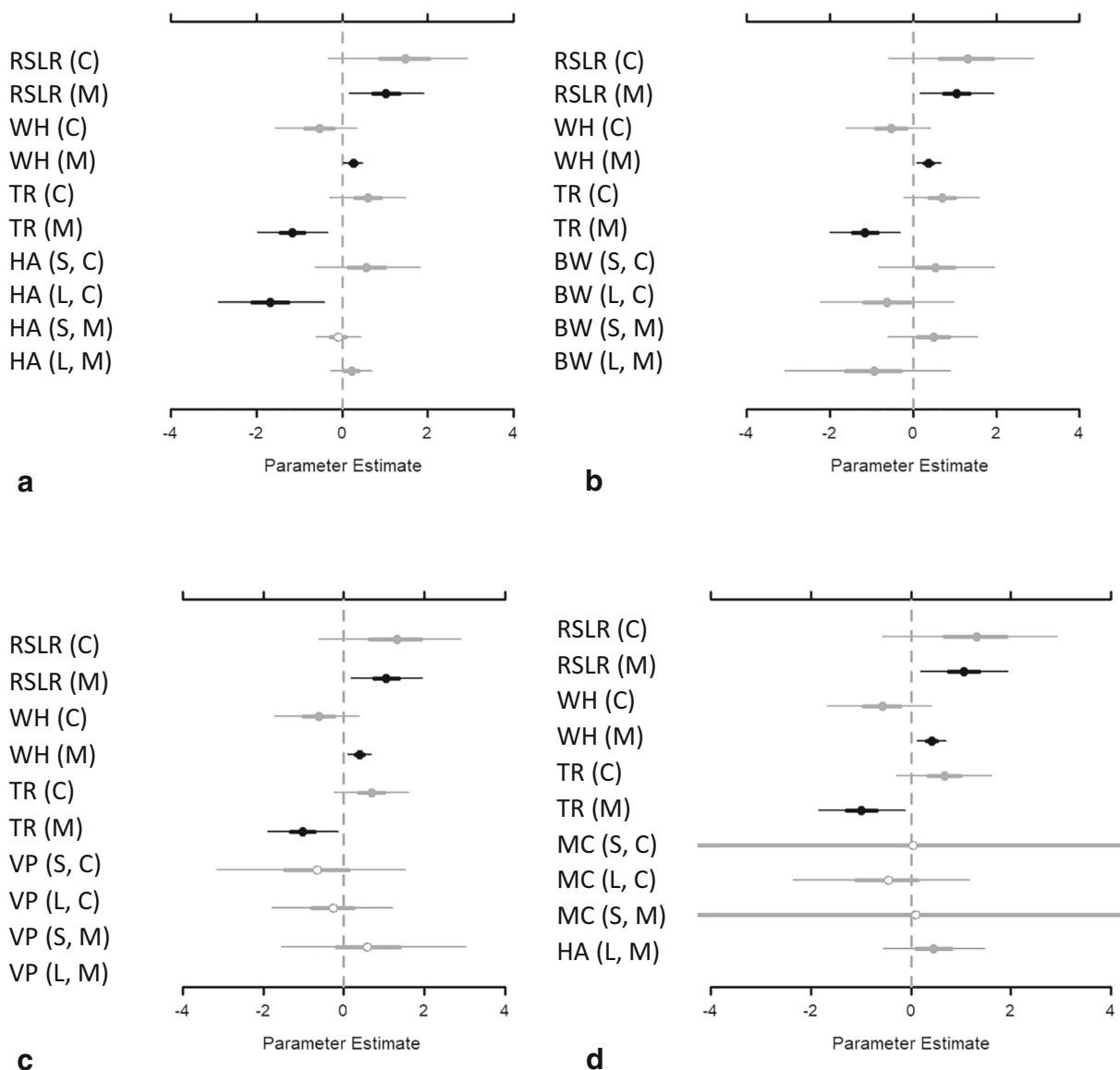


Fig. 3 Credible intervals for the parameters in the areal wetland loss model accounting for restoration activities using hydrological alteration (HA, **a**), breakwater infrastructure (BW, **b**), vegetation planting (VP, **c**), and marsh creation using dredged sediments (MC, **d**). “M” and “C” in the parentheses correspond to lower Mississippi river watershed and Chenier Plain watershed respectively, while “S” and “L” denote whether the restoration had been

implemented in the short term (less than or equal to 10 years) or long term (longer than 10 years). The thin line shows the 95% credible interval, and the thick line shows the 50% credible interval. The dot is the median value and is filled in black when the 95% credible interval does not contain zero (significant effect), filled in grey when the 50% credible interval does not contain zero. Plot generated using MCMCvis package in R (Youngflesh 2018)

Discussion

We developed a modeling framework in Bayesian inference to investigate the effect of various restoration methods on coastal wetland loss in Louisiana. This modeling approach accounted for multiple spatial scales (e.g., watershed and site in this study) akin to the

dynamic coastal systems that are continuously affected by factors at different spatial and temporal scales. It integrated the spatial heterogeneity of SLR coupled with other geomorphological variables. More importantly, it assimilated uncertainties that are important but largely lacking in many of the quantitative restoration evaluations.

Restoration methods

Of the four examined restoration methods—hydrological alteration, breakwaters, marsh creation, and vegetative planting—only hydrological alteration, when implemented over the long term, reduced wetland loss in Chenier Plain watershed significantly. This could be due to the limited number of sites that implemented the other three methods based on the data we have. Although an increased number of sites will provide a more robust evaluation, we think it is important to show application of the modeling framework for these restoration methods based on the available data.

In order to study the effect of site numbers on wetland loss, we selected 11 sites randomly out of the 92 sites that implemented hydrological alteration to match the number of breakwater structure sites. We found that the reduction of coastal wetland loss at the long-term sites was generally larger than at the short-term sites in the Chenier Plain watershed on average, same as what we found based on all 92 sites. However, the significant effect only showed up twice in 20 trials. This shows that it is very likely that we missed effects of other restoration methods due to their limited sites available.

We expected that vegetative planting would reduce wetland loss given literature demonstrating the enhanced shoreline stabilization due to shoreline planting (Ford et al. 2016). However, we did not find vegetative planting as effective as hydrological alteration. The number of the restoration sites using vegetative planting was very small. In addition, wetland loss in Louisiana is largely driven by compaction of poorly-consolidated soil, and sediment starvation (Fagherazzi et al. 2013; Feagin et al. 2015; Hardy et al. 2020 in review; Reed 1989). Whether vegetative planting is useful would likely be resolved at a finer spatial scale because vegetation stabilizes soil and would be more heterogeneous than sediment supply that acts over the broader spatial scales. A wider set of restoration data (multiple states and/or regions) with finer-resolution watershed such as HUC-8 watersheds or the ten hydrological basins used in Louisiana (Couvillion et al. 2013) would allow for finer-scale examination of this type of restoration efficiency. Resolution aside, it requires time for roots to develop dense structures needed for shoreline stabilization (Broome et al. 1986). However, the vegetation may be hard to maintain if not planted at an appropriate elevation. This partly explains why vegetative planting was slightly less effective in reducing wetland loss when implemented in the long term than in the

short term in the Chenier Plain watershed. It is also possible that the temporal coverage we focused on is not fine enough for examining the vegetative planting method given the intense mechanical wave action (Georgiou et al. 2005) and/or high rates of subsidence in Louisiana (Hatton et al. 1983; Yuill et al. 2009). Vegetative planting has seen tremendous success in other regions such as Southern Florida where *Spartina* and mangrove planting successfully restored wetlands, as well as studies outside the Gulf Coast (Curado et al. 2014).

In contrast to vegetative planting, both breakwater restoration and hydrological alteration increase sediment availability continuously. Breakwater restoration encourages littoral sediment trapping through reduction of wave energy, thus allowing the formation of tombolos, lagoons, and eventually wetland area (Birben et al. 2007). Hydrological alteration is the modification of riverine hydrology and thus modification of allocthonous sediment transport, supplying sediment to areas where wetland formation is desired. Both methods are continual restoration methods, i.e., they continue to provide restoration and/or protection once they are constructed. As what we found out, long-term breakwaters seemed to be more effective in reducing coastal wetland loss than hydrological alteration in the lower Mississippi River watershed where relative sea-level rise was lower than the Chenier Plain watershed (Fig. 3a, b). Breakwaters do not only supply continued restoration through sediment trapping but also provide protection from waves and mechanical erosion (Edwards and Namikas 2011).

Hydrological alteration did not significantly reduce wetland loss in the lower Mississippi River watershed, despite extensive hydrological alterations in Louisiana to resupply the Mississippi Delta with river-borne sediment (Allison and Meselhe 2010; Campbell et al. 2005). This result agreed with the previous two landscape studies on the evaluation of Caernarvon freshwater diversion project, built in 1991 to divert water and sediments into Breton Sound (Fig. 1, part of the lower Mississippi River watershed) (Metzger 2007; Turner et al. 2019). Both studies did not detect reduction of wetland loss or wetland gains in Breton Sound, likely due to the positive influences of new sediments being counter-balanced by other factors such as elevated nutrients. Freshwater diversions bring not only sediments but also increase inundation, reduce salinity, increase seasonal fluctuations of salinity, and increase nutrient

loading (Elsey-Quirk et al. 2019). Sediment deposition within an optimal elevation range will likely offset the effects of prolonged inundation limiting plant growth and therefore increase overall wetland productivity. A reduction in salinity may increase plant productivity; however, the effect of its seasonal fluctuations is uncertain. The elevated nutrients will lead to greater above-ground productivity that helps trap more sediments but may lower below-ground productivity. However, nutrients can increase belowground productivity into the newly deposited sediments and positively contribute to soil organic matter accumulation, accretion, and elevation change. Thus, an array of environmental variables need to be considered before restoration as they can interact to affect the outcomes of restoration projects.

In marsh creation using dredged materials, if the soil elevation is appropriate, natural colonization by wetland plants will occur, but the time frame will depend on availability of seeds and propagules and site characteristics (Edwards and Proffitt 2003; Howard et al. 2020; Schrift et al. 2008). *S. alterniflora* colonization and succession have been studied by Edwards and Proffitt (2003) in Sabine National Wildlife Refuge located in the Chenier Plain watershed. However, the created marshes which had comparatively higher elevation were colonized by high marsh species *Spartina patens* and *Distichlis spicata*, and shrubs (Edwards and Proffitt 2003; Howard et al. 2020). Beneficial use of dredged materials has shown some success in the Round Island restoration project in Mississippi as the plants started to colonize the site one year after restoration (personal communication with George Ramseur at Mississippi Department of Marine Resources).

Multiple restoration methods may be applied simultaneously, for example, vegetative planting can be combined with beneficial use of sediments and breakwater structure to increase the pace of restoration. The combination of breakwater construction and vegetative planting is an emerging restoration method known as “living shorelines.” Vegetation has a greater chance of survival and growth with breakwater-induced sediment promotion in the NGOM (Campbell et al. 2005). Marsh sills are created as means of vegetative planting behind breakwaters, and were found to increase both wetland area and nursery production (Gittman et al. 2016). Living shorelines provide restoration beyond the initial restoration project, in comparison to the single-time restoration methods such as marsh creation and vegetative planting.

Though we did not study the effect of multiple restoration methods due to the small numbers of sites, the combinations may further increase performance of restoration practices and should be considered in the future restoration practice. One study in a brackish marsh along Bayou Dupont in the Barataria Basin in the lower Mississippi River watershed (Howard et al. 2020) demonstrates no difference in richness was shown among the planting, non-planted BU sites, and reference sites, or difference of vegetation cover between non-planting sites and reference sites. However, the BU sites with planting had higher vegetation cover and higher elevations than the non-planted BU sites.

Short and long terms

We used different temporal lengths up to 10 years as the cutoffs to represent short- and long-term effect (Søndergaard et al. 2007). We found that 7 years separating short and long terms of hydrological alteration yielded similar results as the application of 10 years. This indicates reduction of coastal wetland loss started to show up when freshwater diversion had been implemented for at least 7 years in the Chenier Plain watershed. When we applied an exponential function to study the relation between coastal wetland loss and the temporal lags in this watershed, we found the median for the coefficient of temporal lags was negative. This indicated that wetland loss was reduced, on average, as the restoration projects aged over time. Due to the limited number of sites, we did not study the different short- and long-term cutoffs in the other restoration methods.

Data used

It is challenging to conduct evaluation at a scale as broad as the entire Louisiana coastline due to the lack of consistent monitoring data across the sites at broad spatial scales (including data in pre-restoration era and reference sites) between 1996 and 2005, especially for extended time (longer than 5 or 10 years) (Allen and Hoekstra 1992; Edwards and Proffitt 2003; Kusler and Kentula 1989; Zedler 2000). Publically available GIS data, some of which were derived from remote sensing images, provide a cost-effective way to evaluate coastal wetlands (Metzger 2007), as shown in our study.

The large variances in the covariates' effects were likely a result of the attempt to describe heterogeneous and complex wetland dynamics with limited

geomorphic covariates. Further evaluation of the restoration methods requires other key drivers missing in the current model such as sediment availability, salinity, nutrients (Elsley-Quirk et al. 2019), planting density (Silliman et al. 2015), vegetation condition, disturbance regime, topography, and seed and propagule supply (Howard et al. 2020). In more detail, the success of freshwater diversion will depend on interaction of sediment availability, salinity, inundation, and nutrient loading. The deposition of sediment could offset the negative impacts of prolonged inundation and nutrient loading on vegetation productivity including below-ground productivity (Elsley-Quirk et al. 2019). However, the sediment data is not available for us to include in our current models.

Success of vegetative planting in restoring coastal wetlands depend on elevation, vegetation, and how far apart the vegetation is planted. While we considered geomorphic drivers, we did not account for planting density. A previous common practice was planting vegetation at a distance to avoid negative competition. Recent restoration experiments in both Western (Netherlands) and Eastern Atlantic salt marshes (Florida in the USA) showed that placing propagules next to each other rather than at a distance may enhance positive species interaction and increase vegetation yields under physically stressed environment as neighboring plants can ameliorate physical stress for each other, including anoxic stress and wave-induced erosion stress (Silliman et al. 2015).

In addition, the exact areas where the restorations took place or impacted were only coarsely represented using site scales. We focused on the locations on the outermost mainland coastlines, while the areas of hydrological alterations affected ranged from upstream to coastlines, so the study did not try to capture the effect of individual restoration projects but only the impact of these restoration projects on wetlands located in the areas that were particularly vulnerable to RSLR. Though we did not detect the significant impact of beneficial use on reduction of wetland loss, the project-scale evaluation showed elevation increased and therefore promoted wetland restoration as it boosted vegetation growth in Barataria Basin, part of the lower Mississippi River watershed (Howard et al. 2020).

Though limited data are available to evaluate the restorations between 1996 and 2005, Louisiana's coastwide reference monitoring system (CRMS) (<https://www.usgs.gov/special->

[topic/gom/science/coastwide-reference-monitoring-system-crms](https://www.usgs.gov/special-topic/gom/science/coastwide-reference-monitoring-system-crms)) provides a valuable dataset for restoration evaluations after 2005, including evaluation metrics and potential drivers. About 390 stations are part of the CRMS that started to collect data including hydrology, accretion, herbaceous marsh vegetation, forested swamp vegetation, soil properties, surface elevation, and land/water composition since 2005 in southern Louisiana. Future studies should consider application of the dataset as done in Turner et al. (2019) that evaluated two diversions of the Mississippi River in the lower Mississippi River watershed: Davis Pond and Caernarvon initiated in 1991 and 2002, respectively.

Modeling framework

The modeling framework developed here shows a useful tool for data-model assimilation. It uses Bayesian framework, so it is readily updatable to refine the results by integrating the new monitoring data, including the new metrics for evaluation, and controlling factors, when they become available, with previous evaluation results as priors. It does not only largely facilitate adaptive resource management but also provides estimates of uncertainties in evaluating performance of restoration activities, which are important for decision makers but largely lacking in the current literatures. The model also suggests the importance of coordinated monitoring network like CRMS for a broad-scale restoration evaluation.

The missing drivers in the model may limit the model's utility in explaining why a particular restoration method was or was not effective mechanistically. However, the statistical model we used is suitable for application at a broad spatial scale (Wu et al. 2015) and provides basis for further research that focuses on mechanisms at the hotspots (successful and failing sites). Therefore, the modeling framework provides a cost-effective way to provide the initial evaluation, and the derived information can be used to guide mechanistic evaluation next that will require intensive on-the-ground monitoring, such as collection of sediment concentration and planting density, etc. The two-stage evaluation can help efficient allocation of limited resources and efforts and lead to more effective conservation and restoration plans.

Conclusion

We developed a coherent modeling framework in Bayesian inference that can integrate multi-scale monitoring data and account for uncertainties to help evaluate the outcomes of coastal wetland restorations. Based on the models, we found that hydrological alteration, when implemented for longer than 7 years, led to significant reduction of coastal wetland loss in the Chenier Plain watershed but not in the lower Mississippi River watershed. The effects of breakwaters, vegetative planting, and beneficial use of dredged sediments on wetland loss were not significant based on the limited data available. However, breakwaters showed higher effectiveness than hydrological alteration in reducing coastal wetland loss, on average, in the lower Mississippi River watershed when implemented for more than 10 years. The interaction of multiple restoration methods were not considered due to limited sites available. The Bayesian modeling framework developed here shows a useful tool for data-model assimilation in restoration evaluation. It is adaptable and accounts for uncertainties, and therefore can guide plans for adaptive management and restoration projects in the future to achieve effectiveness.

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