

Using ecosystem modeling to evaluate trade-offs in coastal management: Effects of large-scale river diversions on fish and fisheries



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ABSTRACT

A coupled ecosystem modeling approach was used to evaluate how select combinations of large-scale river diversions in the lower Mississippi River Deltaic Plain may affect the distribution, biomass, and landings of fish and shellfish over decades relative to a future without action. These river diversions are controlled openings in the riverbank of the Mississippi River designed to reintroduce sediment, water, and nutrients into hydrologically isolated coastal wetlands in order to mitigate wetland loss. We developed a spatial ecosystem model using Ecopath with Ecosim (EwE) software, and prepared it to receive output from a Delft3D hydrodynamic model coupled to primary production models. The Delft3D model provided environmental drivers including salinity, temperature, Chl *a*, total suspended solids, and change in wetland cover as a result of simulated river diversions over decadal model runs. Driver output was averaged either daily, monthly, or annually depending on the parameter. A novel oyster-specific subroutine is introduced in this paper to incorporate information at daily intervals in Ecospace, while Ecospace runs on a monthly time step. The ecosystem model simulates biomass and distribution of fish and shellfish species, and landings of targeted fisheries species, as a result of environmental changes projected for a preliminary set of management scenarios designed to evaluate and screen select combinations of river diversions. Abundant local field samples and landings data allowed for model calibration and validation. The results of simulations indicate that inflow of Mississippi River water in estuaries may cause local shifts in species assemblages. These changes were in some cases direct effects of decreased salinity, such as locally reduced Spotted Seatrout biomass. Changes in some other species in the affected areas resulted from indirect effects; for example, reduced Chl *a* (as a result of increased TSS) resulted in near-field reductions of Gulf Menhaden. The simulations also showed that local biomass reductions were mostly the result of redistribution, since the scenario with the proposed diversions open had minimal impact on the total biomass or landings of species simulated in the Mississippi River Delta as compared to a future without action. The model and its output were used as a decision support tool to help evaluate and compare alternative management actions. The results of this study played a role in the decision by the Coastal Protection and Restoration Authority to prioritize moving forward to conduct more detailed analyses through engineering and design of the two middle diversions but not the two lower diversions that were tested in this study.

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Abbreviations: CFS, cubic feet per second; Chl *a*, chlorophyll *a*; CPRA, Coastal Protection and Restoration Authority; CPUE, catch per unit effort; EwE, Ecopath with Ecosim software; FIMP, Fisheries Independent Monitoring Program; FWOA, future without action; LDWF, Louisiana Department of Wildlife and Fisheries; MRDM, Mississippi River Delta Management; NMFS, National Marine Fisheries Service; OECL, oyster environmental capacity layer.

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1. Introduction

Louisiana is losing land at an alarming rate as a result of the subsidence and erosion of coastal wetlands (Boesch et al., 1994). Because the Mississippi River once built this Deltaic plain (Coleman et al., 1998), large-scale river diversions are a promising restoration measure (Kemp et al., 2014), since they mimic the natural processes that form deltas. River diversions are controlled openings in the levees of the Mississippi River that are used to divert

freshwater containing nutrients and sediment into subsiding estuaries. The main objective of such an approach is to stimulate land building by delivering inorganic and organic sediments, as well as nutrients that stimulate wetland plant growth, resulting in accumulation of organic matter and land stabilization (Wang et al., 2014). Some of the concerns of this approach include the potential negative effects of eutrophication on wetland vegetation (Turner et al., 2002), and the potential negative effects of freshening the estuaries on fish and shellfish (Rose et al., 2014). Simulation models can serve as tools to evaluate the potential of success and trade-offs of planned restoration projects such as river diversions (Yang et al., 2010). Here we present a coupled modeling approach to evaluate effects of coastal restoration on fish and fisheries, and demonstrate the results from a preliminary set of simulations designed to evaluate select combinations of proposed river diversions. The models were developed as decision support tools under the framework of science-based restoration efforts of the Coastal Protection and Restoration Authority (CPRA) in partnership with the US Army Corps of Engineers (USACE).

Output from a physical-biological model, namely the environmental variables that were expected to change under a diversion scenario and affect fish and shellfish, was used as environmental driver input of an ecosystem model. Using salinity, Chl *a*, percent wetland cover per km², total suspended solids, and temperature as environmental drivers, effects on fish and shellfish of proposed operation plans of four river diversions (Lower Breton, Lower Barataria, Mid-Breton, and Mid-Barataria) were compared to a future without action in 50-year simulations.

We describe the model developed for this purpose, and the results of one 50-year scenario compared to a future without action. The scenario simulated four large-scale sediment diversions opened for approximately 5 months each year based on a representative annual hydrograph and an on/off trigger of 600,000 CFS (cubic feet per second) in the Mississippi River. While we included 70 groups (species, life stages and functional groups) in the model to represent the biological community (Table 1), we focus on 6 species in this paper that are of ecological and economic importance: Blue Crab (*Callinectes sapidus*), Brown Shrimp (*Farfantepenaeus aztecus*), Eastern Oyster (*Crassostrea virginica*), Gulf Menhaden (*Brevoortia patronus*), Red Drum (*Sciaenops ocellatus*), and Spotted Seatrout (*Cynoscion nebulosus*). The potential effects on fish and shellfish described here are only one component in the decision-making process to determine how to prioritize further analyses of diversion operations and design, and it provides information needed to evaluate various trade-offs.

2. Methods

2.1. Study area

The study area covers the Lower Mississippi River Delta (Fig. 1). The proposed river diversion locations are indicated in the image. We have focused our research on Barataria Bay and Breton Sound, which are the two basins that directly receive freshwater and sediment flow from the proposed river diversions.

2.2. Modeling approach and simulation scenarios

A coupled modeling approach was used to first simulate the environmental response to a preliminary set of scenarios including select combinations of diversions, and then simulated effects of the changing environment on fish and fisheries (Fig. 2). Delft3D (Roelvink and Van Banning, 1995) was used for the physical-biological modeling, while the fish and shellfish modeling was performed in Ecopath with Ecosim software (EwE; www.ecopath.org).

EwE was used to develop and run an Ecospace model, which allows for temporal and spatial simulations of fish biomass and fisheries yield response to environmental changes, predator-prey interactions, and fishing. The Delft model used was Version 1 of the Mississippi River Delta Management (MRDM) Delft3D, described in detail in Meselhe et al. (2015a).

The MRDM Delft3D model (Version 1) provided outputs used as environmental drivers in the Ecospace model, which included salinity, Chl *a*, percent wetland, total suspended solids, and temperature. The MRDM Delft3D model was calibrated to observed data for all parameters and details specific to the calibration can be found in Chapter 4 of Meselhe et al. (2015a). An additional habitat feature added in Ecospace was percent cultch (the area of stones, old shells and gravel that forms the basis of an oyster bed) based on information gathered by the CPRA. The simulation scenario examined here involved opening four proposed diversions from March to June each year based on a representative annual hydrograph for the river in order to capture effects of maximizing sediment capture from high Mississippi River flow during that time (Fig. 3). This simulation scenario is compared to one that represents a future without action (FWOA), which only has the existing Davis Pond and Caernarvon diversions open with the operational plan as shown in Fig. 3. These two existing diversions are operational during both the 'action' scenario and the FWOA scenario.

2.3. Ecopath model development

To simulate fish and shellfish response to this operational plan, first an Ecopath model was developed using the EwE open source software. An Ecopath model is a mass-balanced representation of the food web of the ecosystem of interest, in this case the Lower Mississippi River Delta. For each group in the model (which includes species, life stages, and functional groups; see Table 1), the initial biomass ($g\ m^{-2}$), the production to biomass ratio ($\frac{P}{B}$), and the consumption to biomass ratio ($\frac{Q}{B}$) were included as initial conditions. With these input parameters, the ecotrophic efficiency (*EE*) was calculated by EwE using the first Ecopath master equation (Christensen and Pauly, 1992). The relative proportion of prey items in the diet of each group was entered based on stomach content data collected in a Louisiana estuary when available (De Mutsert, 2010), published literature, and FishBase (www.fishbase.org), and adjusted based on prey availability under the assumption of opportunistic feeding. Fisheries were included with effort and yield of select groups (Table 2). The first master equation describes the production of each functional group as a set of *n* linear equations for *n* groups:

$$\left(\frac{P_i}{B_i}\right) \cdot B_i \cdot EE_i - \sum_{j=1}^n B_j \cdot \left(\frac{Q_j}{B_j}\right) \cdot DC_{ji} - Y_i - E_i - BA_i = 0 \quad (1)$$

where $\left(\frac{P_i}{B_i}\right)$ is the production to biomass ratio for group *i*, *EE_i* is the ecotrophic efficiency (the proportion of production used in the system), *B_i* and *B_j* are the biomasses of the prey and predators respectively, $\left(\frac{Q_j}{B_j}\right)$ is the consumption to biomass ratio, *DC_{ji}* is the fraction of prey *i* in predator *j*'s diet, *Y_i* is catch rate for the fishery for group *i*, *E_i* is the net migration rate, and *BA_i* is the biomass accumulation for group *i*.

The Ecopath model assumes conservation of mass over a year. Energy balance within each group is ensured with the second master equation:

$$\text{Consumption} = \text{production} + \text{respiration} + \text{unassimilated energy} \quad (2)$$

Table 1
Mass-balanced Ecopath parameters.

| Group # | Group Name | Parameters based on Species | Biomass (g/m ²) | $\frac{P}{B}$ | $\frac{Q}{B}$ | Stanza Break (mo.) | VBGF K | EE |
|---------|----------------------------|--|-----------------------------|-----------------------|---------------------|--------------------|--------------------|---------------------|
| 1 | juvenile shark | <i>Carcharhinus leucas</i> | 0.084 ¹ | 2.0 ¹ | 17.96 ¹ | | | 0.002 ⁴² |
| 2 | adult shark | | 6.87E-07 ⁴² | 0.51 ¹ | 3.9 ¹ | 114 ²⁵ | 0.14 ⁴⁰ | 0.285 ⁴² |
| 3 | dolphins | <i>Tursiops truncatus</i> | 0.002 ² | 0.10 ¹⁵ | 30 ²⁰ | | | 0.001 ⁴² |
| 4 | Kemp's Ridley Sea Turtle | <i>Lepidochelys kempii</i> | 0.007 ³ | 0.19 ³ | 3.5 ³ | | | 0.226 ⁴² |
| 5 | sea birds | <i>Pelecanus occidentalis</i> | 0.008 ¹ | 1.0 ^{15,17} | 17.74 ¹ | | | 0.203 ⁴² |
| 6 | juvenile Red Drum | <i>Sciaenops ocellatus</i> | 0.2 ⁴ | 2.2 ¹⁸ | | | | 0.587 ⁴² |
| 7 | adult Red Drum | | 1.48E-03 ⁴² | 0.62 ¹⁸ | 1.86 ¹⁸ | 54 ²⁶ | 0.46 ²² | 0.373 ⁴² |
| 8 | juvenile Black Drum | <i>Pogonias cromis</i> | 0.109 ⁴ | 2.0 ¹⁹ | | | | 0.856 ⁴² |
| 9 | adult Black Drum | | 1.17E-03 ⁴² | 0.5 ¹⁹ | 6.36 ²¹ | 66 ²⁷ | 0.17 ²² | 0.274 ⁴² |
| 10 | juvenile Sand Seatrout | <i>Cynoscion arenarius</i> | 1.01E-04 ⁴² | 3.7 ¹⁸ | 38.61 ¹⁸ | | | 0.059 ⁴² |
| 11 | adult Sand Seatrout | | 0.023 ⁴ | 0.7 ¹⁸ | 5.4 ¹⁸ | 8 ¹⁸ | 0.13 ²² | 0.172 ⁴² |
| 12 | juvenile Spotted Seatrout | <i>Cynoscion nebulosus</i> | 1.00E-04 ⁴ | 3.7 ¹⁸ | | | | 0.031 ⁴² |
| 13 | adult Spotted Seatrout | | 2.16E-03 ⁴² | 0.7 ¹⁸ | 5.4 ²² | 12 ²⁸ | 0.13 ²² | 0.587 ⁴² |
| 14 | juvenile Largemouth Bass | <i>Micropterus salmoides</i> | 0.003 ⁴ | 2.0 ¹⁹ | | | | 0.46 ⁴² |
| 15 | adult Largemouth Bass | | 0.093 ⁴² | 0.6 ¹⁹ | 2.81 ²¹ | 12 ²⁹ | 0.37 ²⁹ | 0.454 ⁴² |
| 16 | juvenile Gulf Sturgeon | <i>Acipenser oxyrinchus desotoi</i> | 0.002 ⁵ | 2.0 ¹⁹ | | | | 0 ⁴² |
| 17 | adult Gulf Sturgeon | | 1.77E-04 ⁴² | 0.15 ⁵ | 2.1 ²² | 72 ³⁰ | 0.06 ²² | 0 ⁴² |
| 18 | juvenile Blue Catfish | <i>Ictalurus furcatus</i> | 0.037 ⁶ | 2.0 ¹⁹ | | | | 0.72 ⁴² |
| 19 | adult Blue Catfish | | 1.35E-03 ⁴² | 0.8 ²⁰ | 3.3 ²² | 54 ³¹ | 0.15 ²² | 0.608 ⁴² |
| 20 | juvenile Sea Catfish | <i>Ariopsis felis</i> ; <i>Bagre marinus</i> | 0.018 ⁴² | 2.0 ¹⁹ | | | | 0.27 ⁴² |
| 21 | adult Sea Catfish | | 0.156 ⁶ | 0.8 ¹⁸ | 3.3 ²² | 18 ³² | 0.15 ²² | 0.293 ⁴² |
| 22 | juvenile Grey Snapper | <i>Lutjanus griseus</i> | 0.024 ⁷ | 3.0 ¹⁸ | | | | 0.135 ⁴² |
| 23 | adult Grey Snapper | | 2.06E-04 ⁴² | 0.6 ¹⁶ | 2.0 ¹⁶ | 48 ²² | 0.18 ⁴¹ | 0.865 ⁴² |
| 24 | juvenile Atlantic Croaker | <i>Micropogonias undulatus</i> | 0.19 ⁶ | 2.0 ¹⁹ | | | | 0.6 ⁴² |
| 25 | adult Atlantic Croaker | | 1.15 ⁴² | 1.5 ¹⁸ | 8.02 ²² | 12 ³³ | 0.35 ²² | 0.12 ⁴² |
| 26 | juvenile Sheepshead | <i>Archosargus probatocephalus</i> | 0.076 ⁴² | 2.0 ¹⁹ | | | | 0.819 ⁴² |
| 27 | adult Sheepshead | | 0.04 ⁶ | 0.42 ²¹ | 5.9 ²² | 36 ³⁴ | 0.36 ²² | 0.911 ⁴² |
| 28 | juvenile Southern Flounder | <i>Paralichthys lethostigma</i> | 0.007 ⁶ | 2.0 ¹⁹ | | | | 0.658 ⁴² |
| 29 | adult Southern Flounder | | 0.006 ⁴² | 0.42 ²¹ | 4.5 ²² | 36 ³⁵ | 0.23 ²² | 0.737 ⁴² |
| 30 | juvenile Spot | <i>Leiostomus xanthurus</i> | 0.12 ⁴² | 2.0 ¹⁹ | | | | 0.146 ⁴² |
| 31 | adult Spot | | 0.09 ⁶ | 1.1 ¹⁸ | 7.2 ²² | 24 ³⁶ | 0.43 ²² | 0.761 ⁴² |
| 32 | juvenile Pinfish | <i>Lagodon rhomboides</i> | 9.44E-04 ⁴² | 1.02 ¹⁸ | 19.19 ⁴² | | | 0.041 ⁴² |
| 33 | adult Pinfish | | 0.009 ⁶ | 0.3 ²² | 8.0 ¹⁸ | 28 ²² | 0.3 ²² | 0.755 ⁴² |
| 34 | juvenile Silver Perch | <i>Bairdiella chrysoura</i> | 0.022 ^{4,8} | 3.7 ²⁰ | 17.31 ⁴² | | | 0.27 ⁴² |
| 35 | adult Silver Perch | | 0.024 ⁴² | 1.4 ¹⁸ | 9.0 ¹⁸ | 12 ³⁷ | 1.26 ²² | 0.6 ⁴² |
| 36 | juvenile Gulf Menhaden | <i>Brevoortia patronus</i> | 0.138 ⁴ | 2.3 ¹⁸ | 19.38 ⁴² | | | 0.055 ⁴² |
| 37 | adult Gulf Menhaden | | 0.524 ⁴² | 1.9 ¹⁸ | 8.1 ²² | 12 ³⁶ | 0.32 ²² | 0.86 ⁴² |
| 38 | juvenile Striped Mullet | <i>Mugil cephalus</i> | 0.38 ⁴² | 2.4 ¹⁸ | 32.97 ⁴² | | | 0.218 ⁴² |
| 39 | adult Striped Mullet | | 1.44 ⁴ | 0.8 ¹⁸ | 12.28 ²² | 18 ³⁸ | 0.34 ²² | 0.124 ⁴² |
| 40 | juvenile Threadfin Shad | <i>Dorosoma petenense</i> | 0.05 ⁴² | 3.0 ²⁰ | 17.79 ⁴² | | | 0.065 ⁴² |
| 41 | adult Threadfin Shad | | 0.018 ^{6,8} | 2.53 ¹⁸ | 8.2 ²² | 19 ²² | 0.39 ²² | 0.155 ⁴² |
| 42 | juvenile Gizzard Shad | <i>Dorosoma cepedianum</i> | 1.15E-03 ⁴² | 3.0 ²⁰ | 17.79 ⁴² | | | 0.038 ⁴² |
| 43 | adult Gizzard Shad | | 3.77E-04 ⁸ | 2.53 ¹⁸ | 8.2 ²² | 19 ²² | 0.39 ²² | 0.137 ⁴² |
| 44 | juvenile Sunfishes | <i>Lepomis microlophus</i> ; <i>Lepomis macrochirus</i> | 1.96E-04 ⁴² | 2.0 ¹⁹ | 16.46 ⁴² | | | 0.722 ⁴² |
| 45 | adult Sunfishes | | 4.66E-04 ⁴ | 0.8 ¹⁹ | 4.97 ²¹ | 12 ³³ | 0.3 ²² | 0.759 ⁴² |
| 46 | juvenile Bay Anchovy | <i>Anchoa mitchilli</i> | 0.132 ⁴² | 3.0 ²⁰ | 39.78 ⁴² | | | 0.239 ⁴² |
| 47 | adult Bay Anchovy | | 0.172 ⁶ | 2.53 ¹⁸ | 19.42 ²² | 12 ³³ | 0.6 ²² | 0.397 ⁴² |
| 48 | Killifishes | <i>Fundulus</i> spp. | 0.108 ^{4,8} | 2.53 ¹⁸ | 19.4 ²⁰ | | | 0.492 ⁴² |
| 49 | Silversides | <i>Menidia beryllina</i> ; <i>Membras martinica</i> | 0.648 ^{4,8} | 2.3 ¹⁸ | 19.4 ²⁰ | | | 0.852 ⁴² |
| 50 | juvenile Blue Crab | <i>Callinectes sapidus/similis</i> | 0.44 ⁶ | 3.0 ¹⁹ | 17.04 ¹⁸ | | 0.7 ¹⁹ | 0.179 ⁴² |
| 51 | adult Blue Crab | | 0.563 ⁴² | 2.4 ¹⁸ | 8.5 ⁴² | 12.0 ³⁹ | | 0.554 ⁴² |
| 52 | juvenile Brown Shrimp | <i>Farfantepenaeus aztecus</i> | 0.14 ⁶ | 3.0 ¹⁹ | 66.65 ⁴² | | 1.5 ¹⁹ | 0.149 ⁴² |
| 53 | adult Brown Shrimp | | 10.73 ⁴² | 2.4 ¹⁸ | 19.2 ¹⁸ | 3.0 ¹⁹ | | 0.035 ⁴² |
| 54 | juvenile White Shrimp | <i>Litopenaeus setiferus</i> | 0.06 ⁶ | 3.0 ¹⁹ | 66.65 ⁴² | | 1.5 ¹⁹ | 0.335 ⁴² |
| 55 | adult White Shrimp | | 4.78 ⁴² | 2.4 ¹⁸ | 19.2 ¹⁸ | 3.0 ¹⁹ | | 0.046 ⁴² |
| 56 | other shrimp | <i>Farfantepenaeus duorarum</i> ; <i>Xiphopenaeus kroyen</i> | 0.01 ⁹ | 7.76 ²³ | 26.7 ²³ | | | 0.976 ⁴² |
| 57 | oyster spat | | 1.55E-03 ⁴² | 2.0 ¹⁹ | 40.01 ²⁴ | | | 0.735 ⁴² |
| 58 | seed oyster | <i>Crassostrea virginica</i> | 1.05 ⁴² | 1.8 ¹⁹ | 14.65 ²⁴ | 4.0 ¹⁹ | | 0.713 ⁴² |
| 59 | sack oyster | | 0.69 ^{10,11} | 2.4 ¹⁹ | 10.0 ¹⁹ | | 1.2 ¹⁹ | 0.839 ⁴² |
| 60 | Oyster Drill | <i>Thais haemastoma</i> | 1.5 ¹² | 4.5 ¹⁸ | 18.0 ¹⁸ | | | 0.238 ⁴² |
| 61 | Grass Shrimp | <i>Palaemonetes</i> spp. | 0.45 ^{4,8} | 4.5 ¹⁸ | 18.0 ¹⁸ | | | 0.848 ⁴² |
| 62 | Mud Crab | <i>Rhithropanopeus harrissii</i> | 1.0 ⁹ | 4.5 ¹⁹ | 18.0 ¹⁸ | | | 0.993 ⁴² |
| 63 | zoobenthos | annelids | 3.96 ¹³ | 4.5 ¹⁸ | 22.0 ¹⁸ | | | 0.717 ⁴² |
| 64 | benthic crustaceans | Amphipods, isopods | 4.39 ¹³ | 4.5 ¹⁸ | 22.0 ¹⁸ | | | 0.707 ⁴² |
| 65 | mollusks | clams | 4.03 ¹³ | 4.5 ¹⁸ | 22.0 ¹⁸ | | | 0.439 ⁴² |
| 66 | zooplankton | | 4.12 ^{18,21} | 28.77 ²¹ | 84.87 ²¹ | | | 0.496 ⁴² |
| 67 | phytoplankton | | 12.84 ¹⁴ | 101.7 ^{1,21} | | | | 0.327 ⁴² |
| 68 | benthic algae | | 29.78 ⁷ | 3.91 | | | | 0.842 ⁴² |
| 69 | SAV | | 9.78 ¹⁶ | 9.01 | | | | 0.764 ⁴² |
| 70 | detritus | | 100.0 | | | | | 0.321 ⁴² |

Sources: ¹Geers (2013); ²Waring et al. (2011); ³Okey and Pugliese (2001); ⁴LDWF FIMP Seine; ⁵Kirk (2008); ⁶LDWF FIMP Trawl; ⁷SEAMAP (2014); ⁸De Mutsert (2010); ⁹Reed et al. (2007); ¹⁰LDWF FIMP Oyster Square Meter; ¹¹C. Villaruba unpublished data (Davis Pond Monitoring); ¹²M. La Peyre unpublished data; ¹³Rozas and Minello (2011); ¹⁴R. Lane unpublished data; ¹⁵Okey et al. (2004); ¹⁶De Mutsert et al. (2014); ¹⁷Acosta and Dunnire Venier (1998); ¹⁸Walters et al. (2008); ¹⁹C. Walters, personal communication, University of British Columbia; ²⁰Expert Opinion (EO); ²¹Althausen (2003); ²²www.fishbase.org; ²³Arreguin-Sanchez et al. (2008); ²⁴von Bertalanffy (1938); ²⁵Curtis et al. (2013); ²⁶Murphy and Taylor (1990); ²⁷Murphy and Taylor (1989); ²⁸Nieland et al. (2002); ²⁹Boudreaux (2013); ³⁰Huff (1975); ³¹Graham (1999); ³²WRI (2010); ³³De Mutsert et al. (2012); ³⁴Beckman et al. (1991); ³⁵Reagan and Wingo (1985); ³⁶Benson (1982); ³⁷Grammer et al. (2009); ³⁸Collins 1985; ³⁹Guillory et al. (2001); ⁴⁰Cruz-Martinez et al. (2005); ⁴¹Wilson and Nieland (2001); ⁴²Estimated by Ecopath.

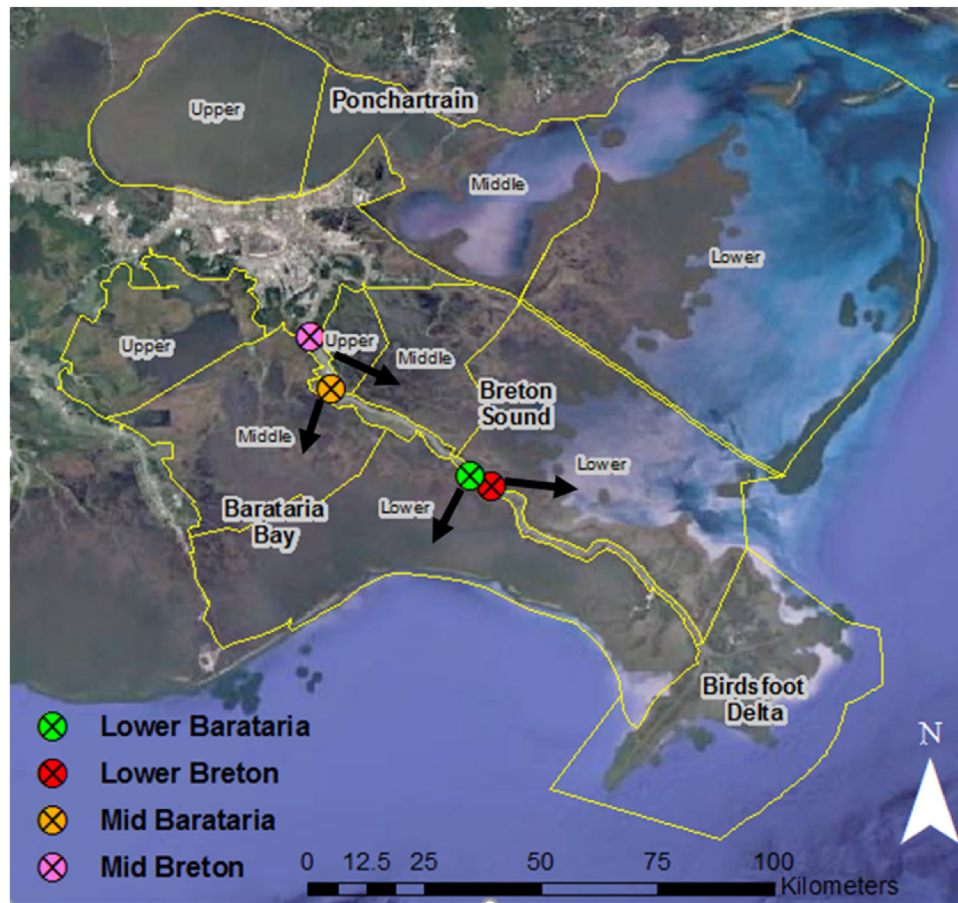


Fig. 1. Model area with location of four proposed large-scale diversions. The model area covers the entire area visible in this image, while the yellow lines indicate the boundaries of sub-regions within the model area. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

where production can be described as:

$$\text{Production} = \text{predation mortality} + \text{catches} + \text{net migration} + \text{biomass accumulation} + \text{other mortality} \quad (3)$$

The list of species or groups in the model were chosen under advisement of the MRDM team, especially based on feedback from Louisiana-based NMFS managers, and include all species that were indicated to be representative of the local environment and for which responses to large-scale diversions are of interest to managers. Many species were split into age classes to better reflect ontogenetic changes in a species life history. This approach resulted in 70 groups varying from a life stage of a single species (e.g. juvenile Red Drum), to guilds (e.g. zooplankton; Table 1).

A vast majority of the data used to create the Delta Management Ecosystem model was derived from the LDWF Fisheries Independent Monitoring Program (LDWF FIMP). The three primary datasets (and gear types) from which these data were obtained were the LDWF Finfish (seine), Shrimp (trawl), and Oyster (square meter). Specific methods for collecting the fishery independent data can be found in the Marine Fisheries Division Field Procedures Sample Design and Data Collection Manual (LDWF, 2002). Initial biomasses were calculated from LDWF FIMP data for the years 1995–2000, and LDWF FIMP data for the years 1995–2013 were used to create the time series of biomass observations that are used in calibration. Abundance (number of organisms collected) and length data are the most consistently reported variables in the LDWF fisheries independent monitoring database. Since EwE requires biomass values in g m^{-2} for initial biomass and calibration data, LDWF abundance

Table 2

Fleets in the model, the species targetted with the fleets, and the bycatch of each fleet.

| Fleets | Target Species | Bycatch |
|--------------|---|--|
| Brown Shrimp | Brown Shrimp | Sharks Sea turtles Sand Seatrout Sea Catfish Atlantic Croaker Southern Flounder |
| Blue crab | Blue Crab | ~ |
| Oyster | Eastern Oyster | Oyster Drill |
| Black drum | Black Drum | ~ |
| White Shrimp | White Shrimp | sharks sea turtles Sand Seatrout Sea Catfish Atlantic Croaker Southern Flounder |
| Recreational | Blue Crab Red Drum Spotted Seatrout Largemouth Bass Blue Catfish Grey Snapper Sheepshead Southern Flounder | ~ |

data per unit effort (CPUE) were converted to g m^{-2} for each species in the model by using length-weight relationships and estimating the area sampled with each gear. LDWF measures 30–50 organisms per sample in the trawl and finfish datasets, so we determined the proportion of species at length for each sample. We found the

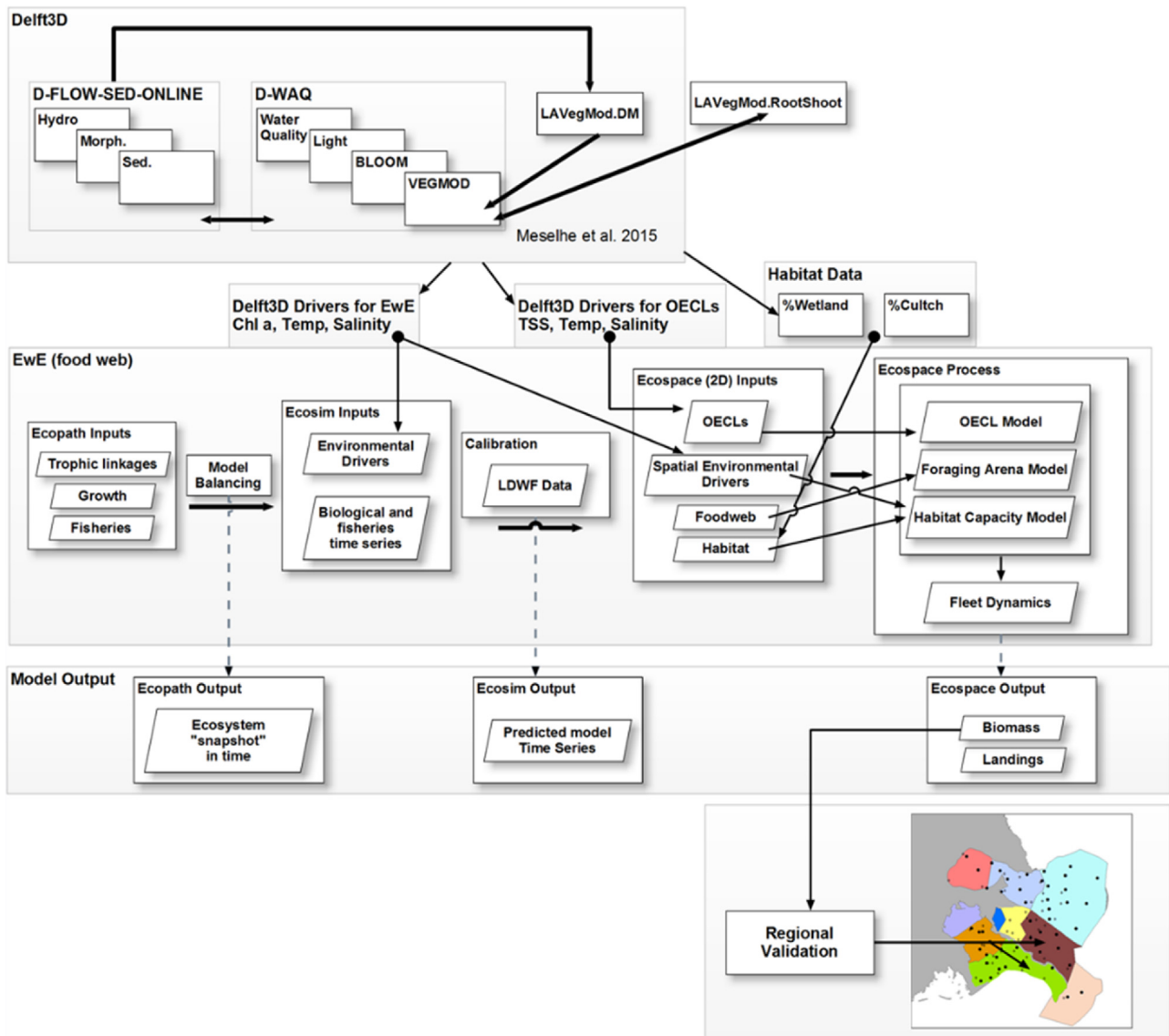


Fig. 2. Conceptual diagram of coupled modeling approach.

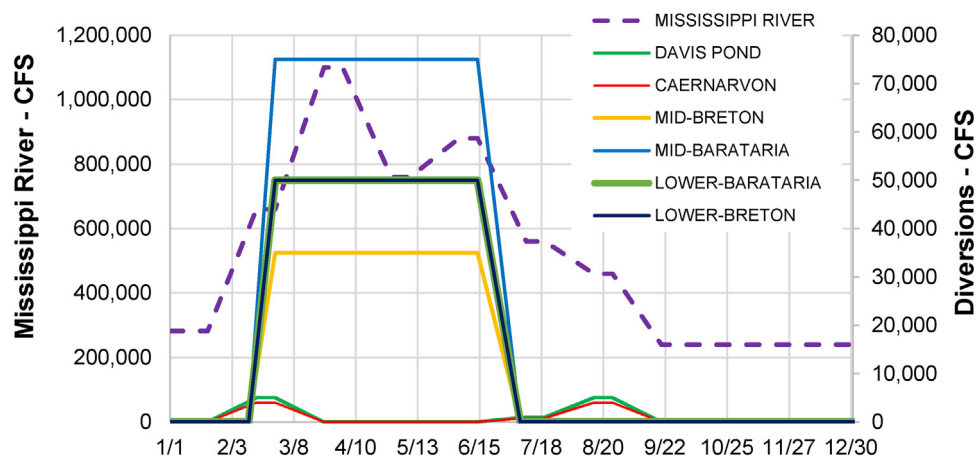


Fig. 3. Representative annual hydrograph for the Mississippi River (left axis) and resulting operations for four simulated large-scale sediment diversions in cubic feet per second (CFS) using an on/off trigger for the sediment diversions of 600,000 cfs in the river: Mid-Breton, Mid-Barataria, Lower-Barataria, and Lower-Breton (right axis). Two existing diversions, Davis Pond and Caernarvon, are included in the figure for reference, and are also part of the simulations (right axis). Adapted from [Meselhe et al. \(2015a\)](#).

Table 3

Goodness of fit metrics of the biomass and catch calibration of the six focal species. No data were available to calibrate the catch of red drum and spotted seatrout, but a recreational fishery on these species is included in the model. EwE SS = the sum of squares calculated in EwE, and used during calibration (best fit model had the lowest total sum of squares), %RMSE = percent root mean square error (calculated post-hoc), PBIAS = percent bias (calculated post-hoc), Corr. Coeff. = Correlation coefficient (calculated post-hoc).

| Group | EwE SS | %RMSE | PBIAS | Corr. Coeff. |
|--------------------------|--------|--------|--------|--------------|
| Brown Shrimp biomass | 2.88 | 19.00 | 0.50 | 0.44 |
| Brown Shrimp catch | 3.37 | 44.57 | 31.10 | 0.66 |
| Blue Crab biomass | 7.16 | 52.00 | −11.00 | 0.51 |
| Blue Crab catch | 0.65 | 119.20 | −44.50 | 0.45 |
| Eastern Oyster biomass | 41.68 | 0.70 | 199.90 | 0.09 |
| Eastern Oyster catch | 1.97 | 28.39 | 32.60 | 0.88 |
| Gulf Menhaden biomass | 29.28 | 297.00 | −69.00 | 0.13 |
| Gulf Menhaden catch | 1.06 | 29.79 | 32.80 | 0.06 |
| Red Drum biomass | 3.90 | 34.00 | 32.30 | 0.19 |
| Spotted Seatrout biomass | 7.41 | 40.00 | 17.20 | 0.11 |

product of the total catch (abundance) per sample (when above the 30–50 organism threshold) and the proportional catch-at-size per sample to determine an estimated catch-at-size for the entire sample. Biomass per sample was determined by using abundance and length data in a length-weight (L-W) regression, $W_i = aTL_i^b$, where a and b are species specific parameters, and TL is total length of species i . All species' biomasses were then divided by the sampling area of each gear type to determine the $g_i m^{-2}$. The length distributions showed that the FIMP survey data we used, which was collected with seines and trawls, mostly targets juvenile nekton species, while the oyster surveys were used to get an estimate of sack oysters (adult oysters greater than 7.62 cm). Therefore the observed initial biomass was entered for the juvenile life stages for nekton, and sack oysters for the Eastern Oyster. The adult biomass (or spat and seed oyster biomass in the case of the Eastern Oyster), was determined using the von Bertalanffy growth function in EwE. The von Bertalanffy K values and the age break for each group are listed in Table 1. Commercial fishing fleets and recreational fishing are included in the model based on trip ticket data provided by LDWF (Table 2). The trip ticket data were used to define which fisheries ("fleets") are operational in the modeled area, which species are part of the catch of each fleet, and what the average biomass landed in metric tonnes $km^{-2} year^{-1}$ was during 1995–2000 to set up the initial conditions (the Ecopath model). Fishing in the model mostly targets the adult life stages.

2.4. Model calibration and validation

The model was mass-balanced in Ecopath using Eq. (1), which ensures no more biomass of any group is consumed or removed by fishing than present in the model over the period of a year. In addition, the time-dynamic module of EwE, namely Ecosim, was used to calibrate the model over a 19-year period, by fitting the species' biomass and catch simulations to annual time series previously described (observations). The fitting to time series routine in Ecosim searches for the lowest total Sum of Squares (SS) fit of the model to observations by adjusting the vulnerability of groups to predation and fishing (Walters and Juanes, 1993; Walters et al., 1997; Christensen and Walters, 2004; Ma et al., 2010; Fig. 4). Since the observed biomass from the FIMP survey represents the juvenile life stages of nekton, while the fisheries represent landings of the adults, biomass was fitted to the juvenile biomass predictions in the model, and catch was fitted to the adult catches in the model. We used fishing effort, based on LDWF trip ticket data, to drive catch. For Eastern Oyster, both biomass and catch calibration was performed on sack oysters. We calculated additional goodness of fit metrics (PBIAS, %RMSE, correlation coefficient) for each of the six focal species and commercial catch of focal species when commercial catch data were available (Table 3). We did not have recreational catch data available to calibrate the recreational catch

in the model. The calibrated model with the lowest total SS was used for the diversion simulations. Calibration was performed on annual average survey and landings data.

We performed regional validation of Ecospace model simulations. While the model calibration described above was performed in Ecosim, the same 19 years (1995–2013) were simulated in Ecospace with all spatial features included as described in 'Ecospace model description' below. Local output was subsequently compared to local observations, averaged by the sub-regions indicated in Fig. 1. We purposely used no fitting procedure during the spatial validation, in order to test model performance.

The same 19-year calibration/validation period preceded each simulation and simultaneously served as a start-up period. 'Year zero' as reported in results is the last year of the start-up run before the 50-year FWOA or large-scale diversion simulations commenced.

2.5. Ecospace model description

The Delta Management Ecospace model domain represents the Barataria, Breton Sound, and Lake Pontchartrain basins (Fig. 1). The model has a grid-cell resolution of 1 km^2 and consists of 21,363 connected active cells. Active cells are cells with water that can be occupied by groups in the model and receive environmental driver input. Nekton and fleets move between cells, while habitat and environmental drivers determine the suitability in each cell using the habitat capacity model as described in Christensen et al. (2014). The habitat drivers included are bathymetry, percent cultch to determine the suitability for oyster settlement, and percent wetland, which is updated each simulation year based on output from the coupled MRDM Delft3D model. Only oysters respond to percent cultch, with grid cells reaching optimal suitability for oyster settlement at 1% cultch or higher. Juveniles of all nekton species respond to percent wetland using a response curve based on a relationship developed by Minello and Rozas (2002). Temperature and salinity for each model cell was updated every month of each year in both the FWOA and the four-diversion scenario simulations based on output from the coupled MRDM Delft3D model. Species-specific salinity and temperature response curves (and sources) of the six focal species are shown in Fig. 5. Chl a output from the coupled MRDM Delft3D model was normalized and included as a primary productivity driver on phytoplankton biomass, which was also updated every month of both simulations. Daily MRDM Delft3D model output of salinity, temperature, and TSS for each cell determined the monthly suitability for oysters by grid-cell based on the combined response of oysters to these variables represented with the curves in Fig. 6. With these monthly suitability values per cell, "oyster environmental capacity layers" (OECs) were devel-

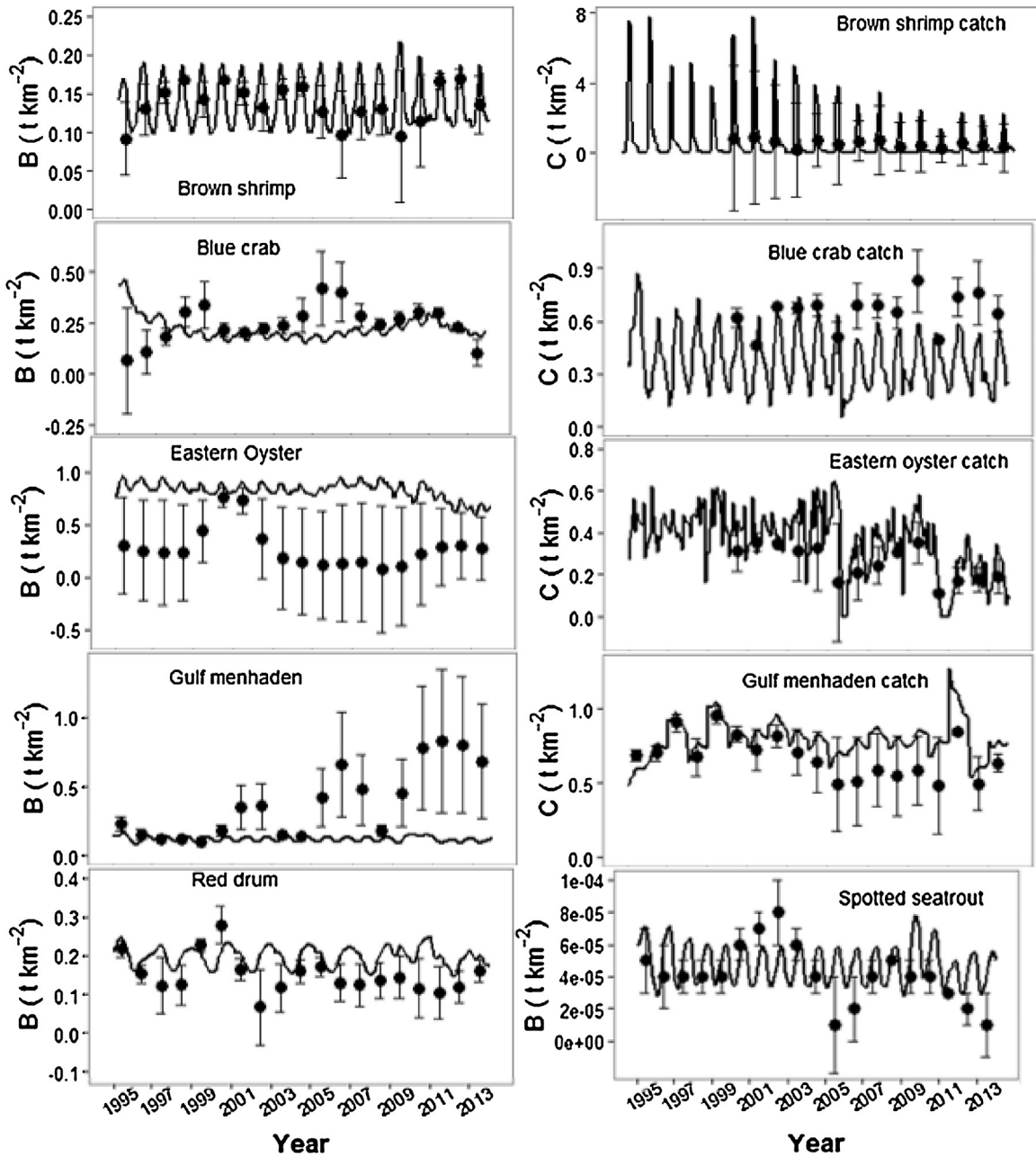


Fig. 4. Model prediction (lines) of biomass (B) and commercial catch (C) of the six focal species. Annual observations (dots) used for calibration are indicated with standard deviation.

oped per oyster life stage (spat, seed oysters, and sack oysters) with the following equation:

$$OECL_{\text{Combined}} = \left(\prod_{i=1}^n OECL_i \right)^{1/n} \quad (4)$$

where $OECL_i$ is the oyster environmental capacity layer of environmental parameter i .

The $OECL_{\text{Combined}}$ was subsequently read back into the Ecospace model at a monthly time step with a linear relationship with oyster habitat capacity in the Ecospace model.

3. Results

The simulation runs were preceded by a 20-year start-up run; regional validation of these 20 years of model output against observations resulted in the goodness of fit metrics listed in Table 4. Two separate model simulations were completed; the same start-up run preceded each scenario. First, we simulated the effects of a 50-year FWOA scenario, and then we simulated the effects of environmental changes due to opening four large-scale sediment diversion for 50 years following the operation regime indicated in Fig. 3. The scenario with four diversions opened from March to June each year freshened the receiving basins, and lowered Chl a near the outfall due to increased TSS, which increased light attenuation (Meselhe et al., 2015a). These changes affected the distribution patterns of the fish and shellfish groups (Fig. 7). Lowered biomass near the diversion outfalls (see Fig. 1 for diversion locations) in year 50 of

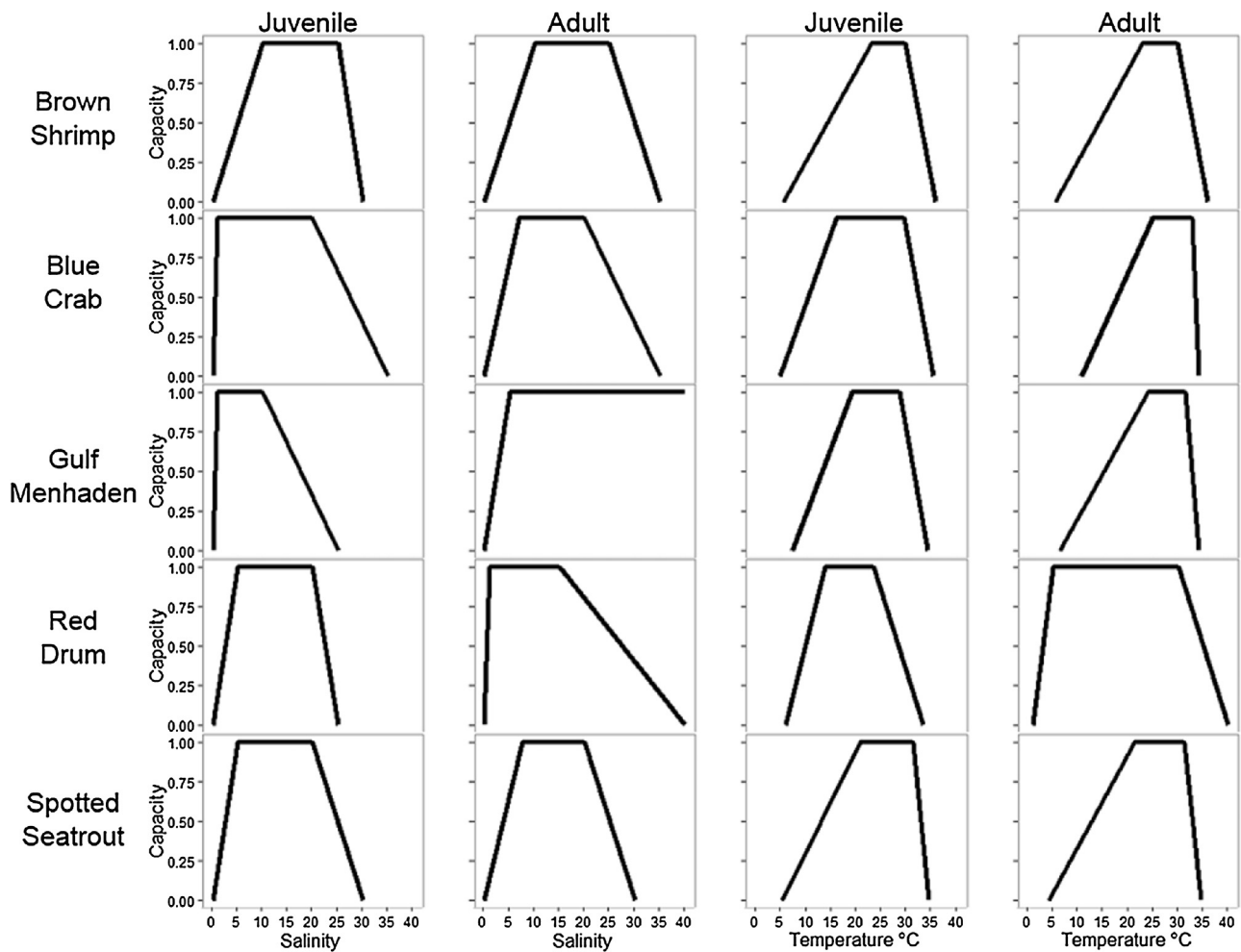


Fig. 5. Salinity and temperature response curves of the juvenile and adult life stages of five focal nekton species in the model.

Table 4

Results of model validation. Goodness of fit metrics of model biomass output of juveniles compared to observations of the five of the six focal species in three regions of a 20-year simulation are shown; we did not have sufficient information to validate oysters. See Fig. 1 for the location of regions.

| Species | Middle Barataria Bay | | | Lower Barataria Bay | | | Lower Breton Sound | | |
|------------------|----------------------|--------|-------------|---------------------|---------|-------------|--------------------|--------|-------------|
| | %RMSE | %BIAS | Corr. Coef. | %RMSE | %BIAS | Corr. Coef. | %RMSE | %BIAS | Corr. Coef. |
| Brown Shrimp | 67.56 | 155.90 | 0.07 | 54.16 | 84.50 | 0.02 | 84.34 | 252.70 | 0.34 |
| Blue Crab | 77.81 | 299.70 | 0.20 | 87.41 | 517.50 | 0.29 | 90.19 | 310.80 | 0.58 |
| Gulf Menhaden | 185.60 | 39.30 | 0.14 | 76.10 | 212.60 | 0.39 | 126.04 | 1.40 | 0.25 |
| Red Drum | 253.40 | −2.30 | 0.16 | 94.26 | 687.00 | 0.15 | 63.17 | 101.80 | 0.20 |
| Spotted Seatrout | 90.43 | 755.00 | 0.01 | 95.82 | 1422.30 | 0.03 | 69.78 | 133.80 | 0.18 |

the diversion scenario as compared to year 50 of the FWOA can be seen in all species that prefer higher salinities, which of our focal species are Brown Shrimp, Spotted Seatrout, and Eastern Oyster. Gulf Menhaden also showed a distinct redistribution away from the diversion outfalls (Fig. 7). This was not entirely a salinity response, considering the juvenile life stage of Gulf Menhaden is quite tolerant of low salinities (Fig. 5), but a bottom-up food web response due to the lowered phytoplankton biomass with increased TSS. There were species with a positive response to the opening of the diversions as well. Of our focal species, Blue Crab did not experience reduced biomass or move away from the diversion outfall location, and biomass increased in areas with emergent marsh. Red Drum exhibited a mixed response, with redistribution away from the Mid Breton Sound diversion, but little response near the outfall of the other diversions, and slight increases in marsh areas when the diversions are open.

These biomass redistributions influenced total biomass in the lower Mississippi River Delta and in the separate sub-regions in the basins (Fig. 8). Fig. 8 shows juvenile biomass response in correspondence with Fig. 7; adults are not shown but had similar results. The sub-regions chosen in Fig. 8 are areas where most changes were expected as a result of the discharge locations of the four diversions in the middle and lower regions of Barataria Bay and Breton Sound (Fig. 1). The model-wide effect (averaged over complete model area) is smaller since other areas less affected by the management action are included in the average. An additional factor in the dampening of effects on a model-wide scale is the redistribution of biomass to more suitable areas within the model area.

Model-wide Brown Shrimp biomass decreased, as well as in all sub-regions. Blue Crab biomass increased in Barataria Bay, but decreased in Breton Sound. Model-wide, Blue Crab biomass increased. Eastern Oyster decreased in Barataria Bay, middle Bre-

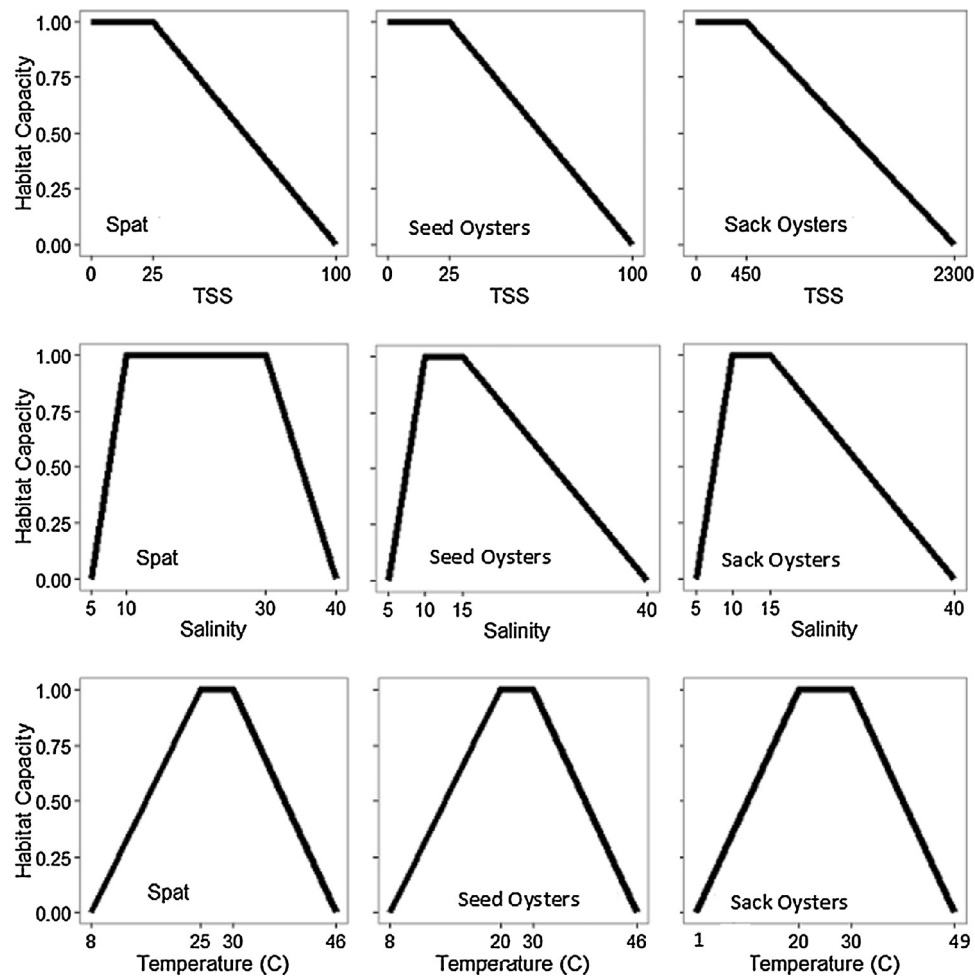


Fig. 6. Total suspended solids (TSS), salinity, and temperature response curves of the three life stages of Eastern Oyster in the model.

ton Sound, and model-wide, and showed no change in lower Breton Sound. Biomass of Gulf Menhaden and Spotted Seatrout decreased in all regions and model-wide. Red Drum biomass decreased in Barataria Bay and middle Breton Sound, and showed no change in lower Breton Sound.

We additionally considered fisheries landings impacts of the scenario runs (Fig. 9). Since the fisheries mostly target the adult life stages, changes in fisheries landings of the adult life stages are shown. All changes are relative to a future without action. Brown Shrimp and Eastern Oyster landings declined in the Barataria and Breton Sound basins, but slightly increased model-wide. Blue crab landings slightly increased in Barataria Bay, but decreased in Breton Sound. Model-wide, there was a slight decrease in Blue Crab landings. Gulf Menhaden and Spotted Seatrout landings decreased in all four sub-regions, and model-wide. Red Drum landings decreased in Breton Sound and Lower Barataria Bay, but increased in Upper Barataria Bay and model-wide.

4. Discussion

A distinct pattern of changes in biomass and catches is that species that prefer higher salinities decrease, and species that prefer lower salinities increase (e.g., Largemouth Bass, not shown). Salinity effects on nekton community composition and distribution in response to diverting river water into estuaries have been shown in model simulations before (De Mutsert et al., 2012; Rose et al., 2015), and have been observed in the field (Piazza and La Peyre, 2011; De Mutsert and Cowan, 2012). Interestingly, the

reduced Chl *a* concentration near the diversion outfalls caused an additional effect. While an influx of nutrient-rich river water could be expected to fuel the food web from the bottom up, the decreased light availability associated with high sediment loads reduced phytoplankton biomass near the diversion outfall, and subsequently the biomass of filter feeders such as Gulf Menhaden. Negative effects of shading on secondary production through bottom-up foodweb linkages have been observed in previous studies (Okey et al., 2004; Henley et al., 2000), as well as general negative effects of suspended sediment on secondary production (Newcombe and MacDonald, 1991 and referenced papers therein; Wilber and Clarke, 2001). Further away from the outfall, but still within Barataria and Breton Sound basins, phytoplankton biomass did increase in response to the increased nutrients, an effect that was partially responsible for the increase in fish biomass away from the outfall when the diversions were open as compared to a future without action. The results emphasize the importance of taking suspended sediments into consideration when evaluating effects of river diversions on fish and fisheries. It also demonstrates the positive effect nutrient loading can have on secondary production through bottom-up processes if the nutrients are used towards primary production (Nixon and Buckley, 2002). It should be noted that nutrient over-enrichment of coastal ecosystems can have several negative impacts as well, such as loss of seagrasses and depletion of bottom oxygen (Boesch, 2002); consequences that are not represented with our current model.

The sediment load in diverted river water plays an important role in the restoration of subsiding wetlands, and is the main reason

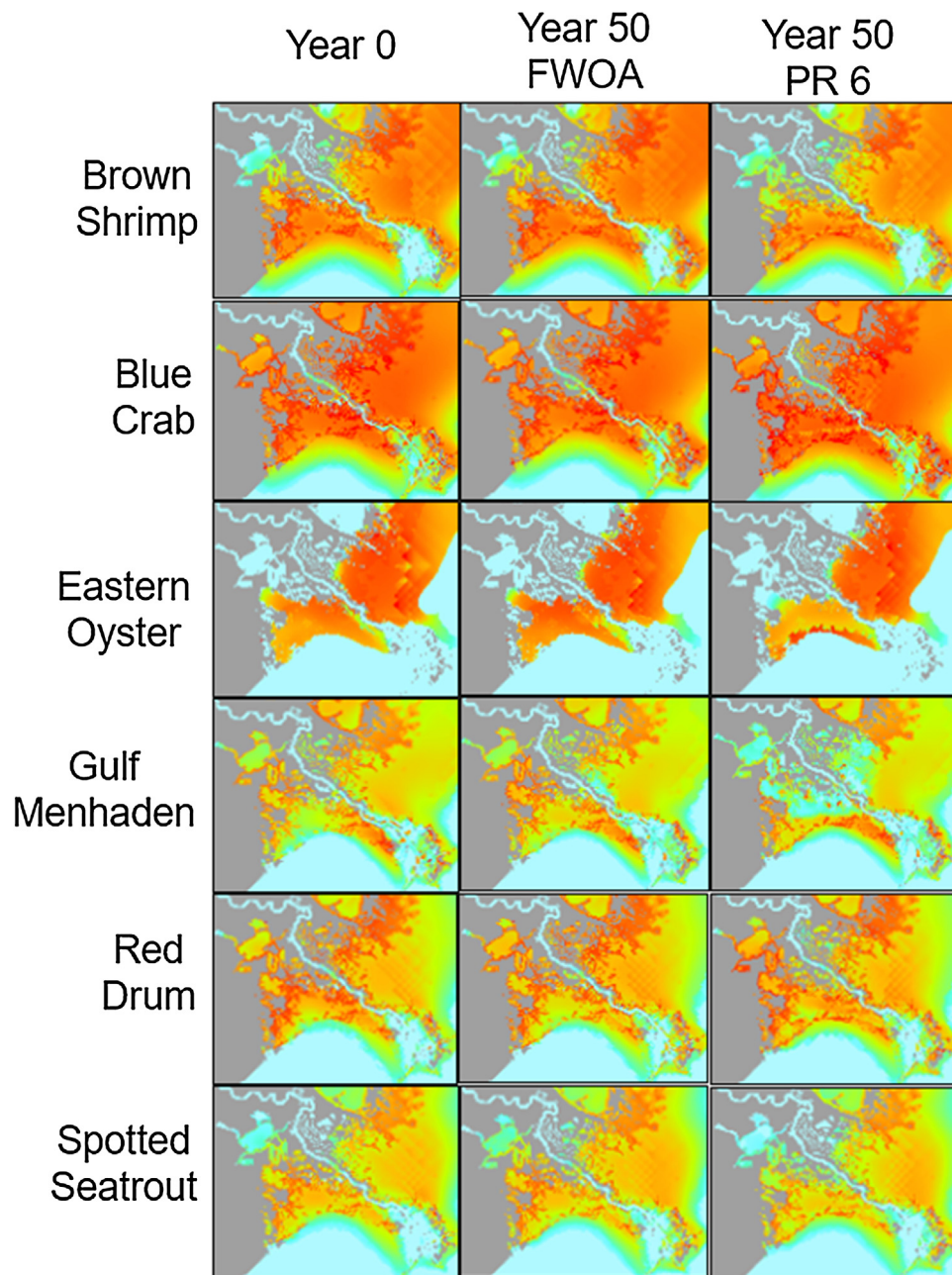


Fig. 7. Biomass distribution maps of the six focal species in June of year 0, year 50 of a future without action (FWOA), and year 50 of a future with action (FWA), which represents all four large-scale diversions open. The distribution patterns from the nekton species are from the juvenile life stage, while eastern oyster is represented by sack oysters. The colors represent relative biomass, with warmer colors indicating higher biomass and cooler colors indicating lower biomass relative to the Ecopath baseline.

the construction of these diversions is being considered. Positive effects on nekton biomass was observed as a result of this process; euryhaline species that are not dependent on the phytoplankton-based food web, such as Blue Crab and Red Drum, increased in the receiving basins as compared to a future without action. The reason for this increase may be partially due to an increase in marsh cover, which serves as a refuge for juveniles and estuarine residents. In current simulations, positive effects of increased marsh cover on fish biomass do not seem to outweigh the negative effects of salinity changes and increased suspended sediments for most species. Previous studies have pointed out that despite the well-known nursery function of marshes, direct links between marsh cover and fisheries productivity have been difficult to find (Chesney et al., 2000; Lewis et al., 2016). However, the net result of this trade-off is likely a factor of the timescale, as the difference between the amount of

marsh cover of a future with action (diversions open) and a future without action increases over time. As marsh erosion continues in a future without action, with rates of loss exacerbated by sea level rise, there will be a point in the future where marshes are eroded to such an extent they can no longer fulfill their nursery function (Chesney et al., 2000; Cowan et al., 2008).

An important general trend is that the model-wide effects of the management action on species biomass are smaller than the local effects. The dual presentation of the results can inform managers where local efforts should be focused, while also putting local changes into context; total stocks of fish and shellfish may not be severely reduced when a local area becomes (temporarily) unsuitable due to low salinity and/or high amounts of suspended sediment, because stocks will move to more suitable areas. While the magnitude of change is different per region, care should be

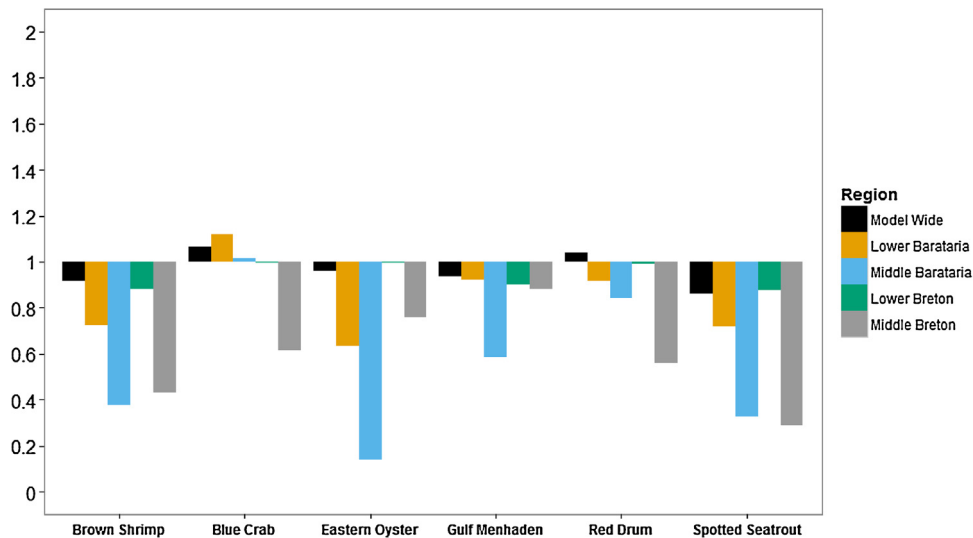


Fig. 8. Juvenile biomass of the six species of interest in year 50 of the FWA simulation (four large-scale diversions open) as compared to a FWOA in four sub-regions of the model and the complete model area. The y-axis represents change in biomass relative to a future without action, where 1 indicates no difference.

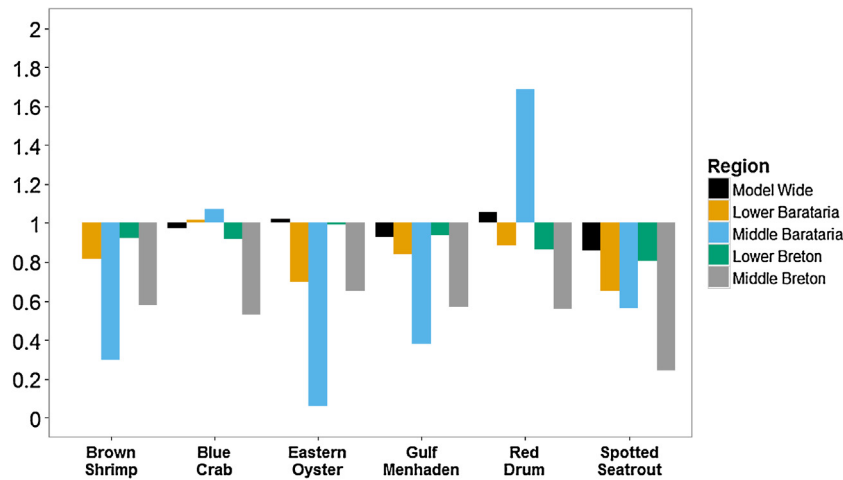


Fig. 9. Catch of the six species of interest in year 50 of the FWA simulation (four large-scale diversions open) as compared to a FWOA in four sub-regions of the model and the complete model area.

taken when interpreting the between-region differences in magnitude. For example, while the decrease in juvenile Blue Crab biomass relative to a future without action in the Middle Breton Sound region is larger than the increase in the lower Barataria Bay region, the model-wide effect shows an increase. In this case the amount of biomass in lower Barataria Bay is higher than that of middle Breton Sound, so a small relative increase in lower Barataria Bay has a bigger effect on the model-wide difference than a larger relative decrease in middle Breton Sound.

One of the limitations of our calibration method is that there is no spatial component to Ecosim, therefore habitat characteristics or movement that could affect biomass is not well represented in the calibration process. This factor can result in a poor fit in biomass calibrations in cases where habitat features play a large role in species biomass and distribution. For example, the SS of the Eastern Oyster biomass fit is 41.68 (poor fit), while the SS of Eastern Oyster catch fit is 1.97 (good fit). Note however, that these spatial attributes are included in the eventual Ecospace model we have used to generate the simulation results, they were just not part of the calibration process. Spatial calibration in Ecospace with EwE software is currently under development.

Of note as well is that we use fishery independent survey data as observations and that we calibrate time-series data of modeled species biomass as well as landings. Calibrating to fishery-independent data is inherently less likely to provide good fits because of natural variability, which has a large effect on the generally small catches of surveys. This issue is partially addressed here by creating LOESS curves from the field observations, and using the resulting points as observations in the calibration. However, in addition to that, the model is restricted to tweaking the biological processes that could result in increases or decreases of biomass (e.g., vulnerability to predation) to match the observations. For landings calibrations, the amount fished and landed can more easily be adjusted to match recorded landings. These notions must be considered when goodness of fit metrics are developed; 'benchmark' values should be different for biomass calibrations than for landings calibrations. When the goodness of fit of two models is compared, the data used to calibrate the model, and what model output is calibrated, needs to be considered. The establishment of such benchmarks and performance metrics is in the early stages, and some documentation relevant to EwE is under development (see e.g. [Meselhe et al., 2015b](#)).

The presented Ecospace model is the first with an added subroutine to allow for the incorporation of effects of fluctuation in environmental variables at a daily time step. We specifically included this routine to simulate environmental effects on Eastern Oyster, and termed the resulting drivers Oyster Environmental Capacity Layers, which determine the suitability for three separate life stages of oysters based on daily values of salinity, temperature, and total suspended solids. The assertion that environmental effects of these parameters on oysters are not represented well with the monthly mean of those parameters (E. Melancon, pers. comm.; La Peyre et al., 2016), made this subroutine necessary. This now existing ability to incorporate daily effects overcomes the well-known limitation of EwE of operating on a monthly time-step, and can be applied in other ecosystems and species where such an adjustment is deemed necessary. This model is also the first case where the spatial-temporal framework in Ecospace, introduced in Christensen et al. (2014), is used in an ecosystem-based management approach.

Results of these and other simulations have been taken under consideration by CPRA in the decision-making process regarding prioritizing further analysis of proposed sediment diversions. The decision has been made to not move forward with more detailed study of the two lowermost large-scale diversions discharging into the lower Barataria Bay and Breton Sound basins at this time, because the gains do not seem to outweigh the costs of construction at those locations. Additional calculations and modeling efforts were also instrumental in this decision, such as presented in Meselhe et al. (2015a). Moving forward, new simulations are planned involving just the two Mid diversions, discharging into Upper Breton Sound and Middle Barataria Bay, before final decisions on diversion operation and construction design are made. Future simulations should investigate at what point in time marsh building yields a net positive effect for fish and fisheries, which is only seen for select species during a 50-year simulation. Additional recommendations for improvement include spatial model calibration, and fine-tuning the relationship between percent cultch and oyster settlement, as including that relationship into model simulations would provide a more conservative estimate of oyster growth.

This work demonstrates that modeled changes in the environment in response to coastal restoration projects can elicit estimates of changes in fisheries biomass, yield, and distribution by employing model coupling and an ecosystem-based approach. Current simulations, comparing the opening of four large sediment diversions compared to a future without action, shows fish and shellfish response to reduced salinity and Chl *a* near the diversion outfalls. While some net biomass changes (mostly reductions) do occur, most regional changes are a result of biomass redistribution, as is evident from the relatively small biomass changes over the complete Mississippi River Delta as compared to biomass changes seen in smaller sub-regions. Outputs were used in a science-based evaluation and screening approach of CPRA. Their subsequent decision was to not prioritize detailed analyses of the two lowermost diversions, and to concentrate science and simulations on the two middle diversions to support decisions on operation and construction design.

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