# The Effect of the Deepwater Horizon Oil Spill on Two Ecosystem Services in the Northern Gulf of Mexico

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Highlights:

- The yield of five functional groups were used to calculate changes in fishery catch.
- Detritus biomass was used as a proxy for carbon buried offshore to calculate POC sequestration.
- The DWH simulation led to an increase in fisheries overall and decrease in POC sequestration ecosystem services in 2010.
- The model predicted an estimated loss of $15-16 million per year in stone crab fisheries but estimated gains of up to $20 million per year in the other groups from 2010-2012.
- Model simulations estimated a loss of 1,200 in the ability of the Northern Gulf of Mexico offshore environment to sequester POC in 2010

**Abstract**

The Deepwater Horizon (DWH) oil spill likely affected ecosystem services in the Gulf of Mexico. To test this hypothesis, we configured a "Ecopath with Ecosim" model and quantified the effects of commercial fisheries and particulate organic carbon (POC) sequestration from 2004 – 2014, encompassing DWH. The yield of five functional groups were used to calculate changes in fishery catch and detritus biomass as a proxy for carbon buried offshore to calculate POC sequestration. The model predicted an estimated loss of $15-16 million per year (-13%) in stone crab fisheries but estimated gains of up to $20 million per year (11%) in the other four groups from 2010-2012. Model simulations estimated a loss of $1,200 (-0.15%) in the ability of the Northern Gulf of Mexico offshore environment to sequester POC in 2010. The DWH simulation led to an increase in fisheries overall and decrease in POC sequestration ecosystem services in 2010.

Keywords: Ecopath; Gulf of Mexico; Ecosystem Services; Oil Spill; Model; Deepwater Horizon

## 1. Introduction

The Gulf of Mexico (GoM) is affected by multiple stressors such as habitat loss, degraded water quality, overfishing, hypoxia, and harmful algal blooms (NRC, 2013). The explosion of the Deepwater Horizon (DWH) drilling platform on 20 April 2010, followed by an 87-day uncontrolled oil spill added another stressor, affecting approximately a 11,200 $km^2$ of the surface offshore environment (MacDonald et al. 2015), and 8,400 $km^2$ of the bottom (Chanton et al. 2014). This large area of contamination likely impacted offshore ecosystem services. The provisioning services in the offshore environment include the acquisition of fish, shellfish, oil, gas, minerals, and chemical compounds for manufacturing (Armstrong et al., 2012). The regulating services include the regulation of gas and climate through the biological pump, waste regulation and detoxification through bioturbation, and biodiversity (Armstrong et al., 2012). The offshore supporting services include habitat, nutrient cycling, water cycling, chemosynthetic primary production, and resilience (Armstrong et al., 2012). It is possible that any of the aforementioned services could have been affected by the DWH. However, work to date on offshore ecosystem services has focused on market-based services such as tourism and commercial and recreational fisheries (Worm et al. 2006; White et al. 2012; Cavanagh et al. 2016; Martin et al. 2016), identification of goods and services that exist in the offshore environment (Armstrong et al. 2010; Werner et al. 2014; Barbier, 2017), and assessing the value that stakeholders place on specific offshore services (Yoskowitz et al. 2016; Lau et al. 2018). Yet these studies have not addressed the potential loss in carbon sequestration due to this offshore uncontrolled oil spill from the deep ocean.

Ecosystem services are the direct and indirect contributions from ecosystems that support, sustain, and enrich human life (Peterson and Lubchenco, 1997; Holmlund and Hammer, 1999; Carollo et al. 2013; Yoskowitz et al. 2016). There are four different ecosystem service categories: provisioning, regulating, cultural, and supporting services (MEA, 2005). Provisioning services are the goods produced by ecosystems and directly consumed by humans. Regulating services are the processes that maintain the conditions favorable to life. Cultural services are the non-material benefits such as aesthetic values. Supporting services drive the other three services. Therefore, valuation of ecosystem services focuses on provisioning, regulating, and cultural services to people (NRC, 2013). Considering value provided by the ecosystem can change the way decisions makers manage ecosystems but requires more data on interactions within the ecosystems and connections to specific human benefits (NRC, 2013).

In this study we investigate whether and how offshore ecosystem services were affected by the DWH blowout. Waste regulation ecosystem services did change following the DWH event (Washburn et al. 2018), but what about the other services? To estimate ecosystem services changes resulting from the oil spill, effects on the ecosystem must be quantified, changes in goods and services must be quantified, and change in cost to society must be quantified (NRC, 2013). The effects on two ecosystem services, commercial fisheries and particulate organic carbon (POC) sequestration were estimated. This was accomplished by building a model of multiple species to account for changes at the level of the fishing sector, which each catch multiple species. The model also captured detrital production from several sources such as dead fish, benthos, and plankton as a proxy for POC sequestration. Ecopath with Ecosim (EWE) was used because it takes into account the aforementioned processes, has relaxed data requirements, is commonly used, is user friendly, and is free. Therefore, the aim of this study was to build an EWE model to test whether there were losses in commercial fisheries and POC sequestration as a result of the DWH oil spill.

## 2. Methods

To test whether there were losses in ecosystem services as a result of the DWH oil spill, an EWE model (version: 6.5.14034.0) of the Northern Gulf of Mexico was built, and the changes to ecosystem services were calculated from the model outputs. EWE utilizes a trophic flows model based on the mass-balance fluxes of biomass (Christensen et al. 2005). The foundation of the Ecopath model is formed by two equations (Christensen et al. 2005):

(1) the production equation (Christensen et al. 2005)

$$Bi * \left(\frac{P}{B}\right) i = Yi + \sum_{j=1}^{n} \sum Bj * \left(\frac{Q}{B}\right) j * DCji + Ei + BAi + Bi \left(\frac{P}{B}\right) i * (1 - EEi) \text{ eq. 1}$$

or, more simply for species *i*,

Production = Catches + Predation Mortality + Net Migration + Biomass Accumulation + Other Mortality.

(2) the consumption equation (Christensen et al. 2005)

$$B * \left(\frac{Q}{B}\right) = B * \left(\frac{P}{B}\right) + (1 - GS) * Q - (1 - TM) * P + B * \left(\frac{Q}{B}\right) * GS \text{ eq. 2}$$

or, more simply,

Consumption = Production + Respiration + Unassimilated Food

In equation one (eq. 1), *i* refers to the prey and j refers to the predator. For the remaining representations in both equations above, *B* is biomass, *P* is production rate, *Y* is fishery catch, *Q* is consumption, *DC* is the fraction of prey (*i*) in the average predator (*j*) diet, *E* is emigration, *BA* is biomass accumulation, *EE* is ecotrophic efficiency, *GS* is autotrophy, and *TM* is the unassimilated fraction. A trophic flow approach enables consideration of the whole ecosystem from phytoplankton, to detritus, to benthos, to fish (Christensen et al. 2005). Within EWE, two main linked routines were used, Ecopath, and Ecosim. Ecopath is a static mass-balance picture of the ecosystem, and Ecosim allows for the representation of temporal dynamics (Christensen et al. 2005).

The model is described in full in the supplementary materials. In brief, the model was generated by expanding upon an existing Northern Gulf of Mexico model by Suprenand et al. (2015). First, the original infauna functional group was divided into meiofauna and macrofauna size classes because these two groups responded differently to the DWH oil spill. Second, oil forcing functions were added to simulate the effect of the DWH blowout (Figure 1). Finally, ecosystem services were linked to the relevant functional groups and monetary evaluation methods were applied.

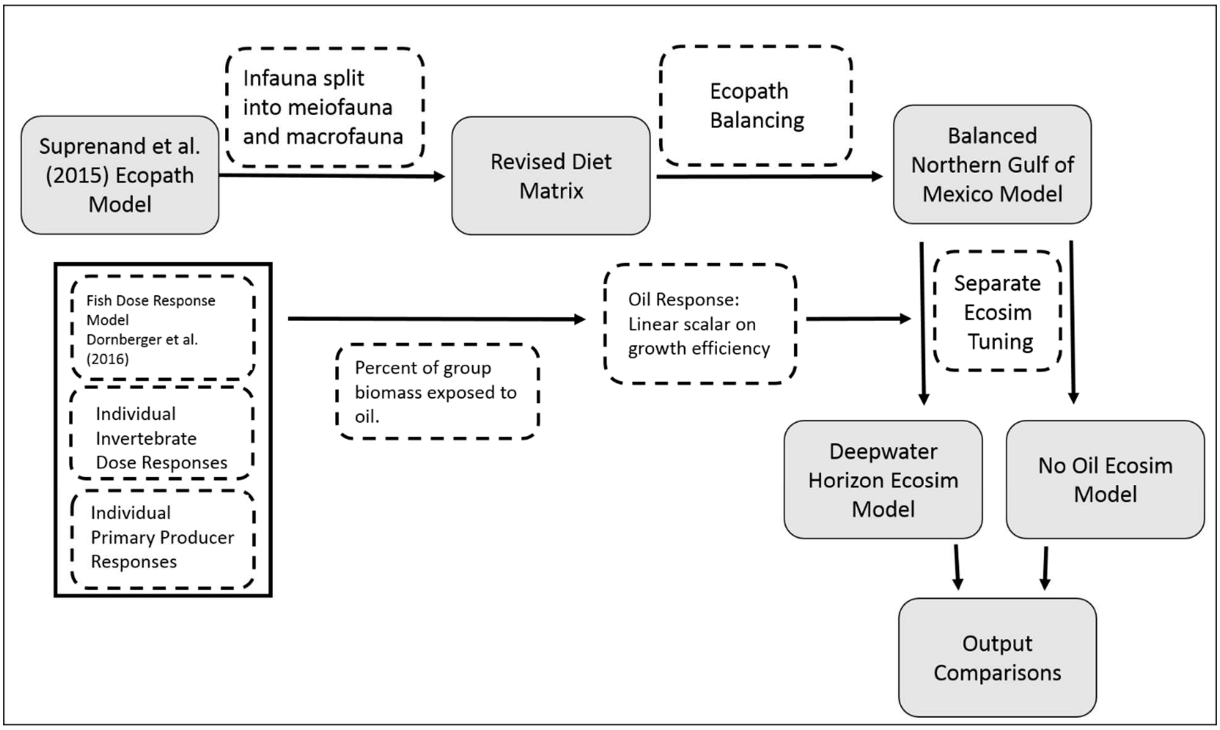

Figure 1. Conceptual layout of model creation.

## 2.1 EWE Model Simulations

The domain of the EWE model ranges from 24 – 31 °N latitude to 80 – 98 °W longitude with depths ranging from 0 – 2000 m, including both nearshore and offshore zones (Figure 2). The original model (Suprenand et al. 2015) contained 48 functional groups (Tables 1 and 2). We added meiofauna and macrofauna functional groups, oil forcing functions, and removed red tide as a fishery from the Suprenand et al. (2015) model. The meiofauna functional group includes nematodes, copepods, ostracods, and kinorhynchs. The macrofauna functional group includes polychaetes, isopods, and amphipods. Two simulations starting with initial conditions in 2004 were run and predicting forward to 2014: (1) no oil and (2) oil. To improve the predictive power of the model, vulnerabilities were optimized by fitting to a time series and outputs were compared to observational data when possible following Heymans et al. (2016)'s best practices. Statistical analysis of model fit was checked against the 2004-2014 observational data by calculating correlation coefficient (r), root means squared error (RMSE), reliability index (RI), average error (AE), average absolute error (AAE), modeling efficiency (MEF), Pearson correlation, Spearman correlation, and Kendell Correlation in Excel 2016 for catch and relative biomass of model outputs and observational data (Stow et al. 2009, Olsen et al. 2016). Correlations greater than 0.5 are highly correlated (Olsen et al. 2016). RMSE, AE, and AAE are the measure of the discrepancy size and indicate a good fit when the values are close to one. RI is the average factor by which predictions differ from observations, values close to one indicate good predictions (Stow et al. 2009). MEF is the objective model performance, values above zero indicate above average performance (Olsen et al. 2016).

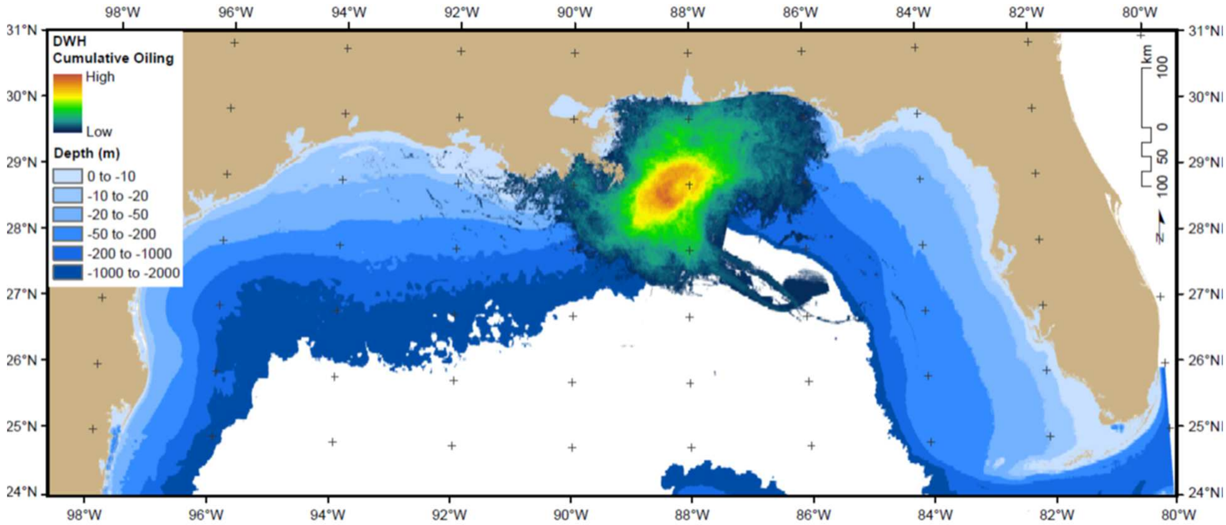


Figure 2. Map of the area modeled within Ecopath with Ecosim and the area of the surface oil slick.

Table 1. A list of functional groups with different age classes included in the model and the associated species within each group. This list is largely based off the original model of Suprenand et al. 2015.

| Functional Group | Species | Multistanza Groups (Months) |
|---|---|---|
| Red Drum | *Sciaenops ocellata* | (0-3), (3-8), (8-18), (18-36), (36+) |
| Sea Trout | *Cynoscion arenarius* | (0-3), (3-18), (18+) |
| | *Cynoscion nebulosus* | |
| | *Cynoscion nothus* | |
| Mullet | *Mugil cephalus* | (0-6), (6-18), (18+) |
| | *Mugil curema* | |
| Mackerel | *Auxis rochei* | (0-3), (3+) |
| | *Scomber japonicas* | |
| | *Scomber colias* | |
| | *Scomber scombrus* | |
| | *Scomberomorus cavalla* | |
| | *Scomberomorus maculatus* | |
| | *Scomberomorus regalis* | |
| Ladyfish | *Elops saurus* | (0-10), (10+) |
| Grouper | *Epinephelus morio* | (0), (1-3), (3+) |
| | *Epinephelus spp.* | |
| | *Epinephelus adscensionis* | |
| | *Epinephelus drummondhayi* | |

| | | |
|---|---|---|
| | *Epinephelus flavolimbatus* | |
| | *Epinephelus guttatus* | |
| | *Epinephelus itajara* | |
| | *Epinephelus nigritus* | |
| | *Epinephelus niveatus* | |
| | *Mycteroperca bonaci* | |
| | *Mycteroperca interstitialis* | |
| | *Mycteroperca microlepis* | |
| | *Mycteroperca phenax* | |
| | *Mycteroperca venenosa* | |
| Menhaden | *Brevoortia patronus* | Juvenile and Adult |
| | *Brevoortia gunteri* | |
| | *Brevoortia smithi* | |
| Red snapper | *Lutjanus campechanus* | (0-6), (6-24), (Older) |

Table 2. A list of all functional groups without age classes included in the model and the associated species within each group. This list is largely based off the original model of Suprenand et al. 2015. The benthic invertebrate functional group includes the organisms larger than 1 cm such as other crab species, star fish, and bivalves.

| Functional Group | Species | Functional Group | Species | Functional Group | Species |
|---|---|---|---|---|---|
| Jacks | *Caranx hippos* | Rays | *Rhinoptera bonasus* | Atlantic croaker | *Micropogonias undulatus* |
| | *Caranx spp.* | | *Hypanus sabina* | Large Coastal Sharks | *Carcharhinus leucas* |
| | *Caranx crysos* | | *Hypanus americana* | | *Carcharhinus limbatus* |
| | *Hemicranx amblyrhynchus* | Pompano | *Trachinotus carolinus* | | *Carcharhinus isodon* |
| | *Seriola spp.* | | *Rachycentron canadum* | | *Isurus oxyruchus* |
| | *Seriola dumerili* | | *Alectic ciliaris* | | *Carcharhinus plumbeus* |
| | *Seriola fasciata* | | *Trachinotus falcatus* | | *Sphyrna mokarran* |
| | *Seriol zonata* | Lobster | *Homarus* | | *Carcharhinus brevipinna* |
| | *Seriola rivoliana* | | *Munidopsis spp.* | | *Galeocerdo cuvier* |
| Bay Anchovy | *Anchoa mitchilli* | | *Munida spp.* | Benthic Invertebrates | |
| Pin Fish | *Lagodon rhomboids* | | *Munida flinti* | Macrozooplankton | |
| | *Diplodus holbrooki* | | *Munida forceps* | Microzooplankton | |
| Small fish | *Dorosoma petenense* | | *Munida iris* | Meiofauna | *Kinorhyncha* *Nematoda* *Copepoda* *Ostracoda* |
| Silver Perch | *Bairdiella chrysoura* | | *Munida irrasa* | Macrofauna | *Polychaete* *Isopoda* *Amphipoda* |
| Scaled Sardine | *Harengula jaguana* | | *Munida longipes* | Attached Microalgae | |
| Catfish | *Bagre marinus* | | *Munida pusilla* | Sea Grass | |
| | *Ariopsis felis* | | *Munida robusta* | Phytoplankton | |
| | *Ictalurus furcatus* | | *Munida simplex* | Detritus | |

| | | |
|---|---|---|
| | *Ictalurus punctatus* | *Munida valida* |
| Caridean Shrimp | | *Nephropsis spp.* |
| Shrimp | *Farfantepenaeus notialis* | *Nephtropsis aculeata* |
| | *Farfantepenaeus subtilis* | *Nephtropsis rosea* |
| | *Liptopenaeus schmitti* | *Polycheles typhlops* |
| Stone Crab | *Menippe mercenaria* | *Panulirus argus* |
| | *Menippe adina* | *Sclaridae spp.* |
| | *Menippea spp.* | *Scyllarides aequinoctialis* |
| Blue Crab | *Callinectes spp.* | *Scyllarides delfosi* |
| | *Callinectes sapidus* | *Scyllarides depressus* |
| | *Callinectes similis* | *Scyllarides nodifer* |
| | *Callinectes ornatus* | *Scllarus spp.* |
| Pigfish | *Orthopristis chrysopterus* | *Scyllarus americanus* |
| | | *Scyllarus chacei* |

### 2.1.1 Addition of Meiofauna and Macrofauna

In the original model (Suprenand et al. 2015), organisms living on/in the sediment included blue crabs, stone crabs, benthic invertebrates, and infauna. For the present study, the original infauna functional group (biomass: 20 t/km$^2$) was further divided into the meiofauna and macrofauna functional groups. This was done because meiofauna (between .042 and 0.3 mm) and macrofauna (> .3 mm) responded differently to the DWH spill (Baguley et al. 2015; Washburn et al. 2016). This distinction may also have an effect on ecosystem service valuation because certain fish species and age groups preferentially feed on different groups. For example, meiofauna are an important food source for the juvenile stages of many fish species (Mullaney and Gale, 1996; De Morais and Bodiou, 1984).

The proportion of meiofauna to macrofauna changes with depth, and the model encompasses a large depth range, therefore a realistic separation of the two groups had to be established (Thiel, 1979). Starting proportions of meiofauna and macrofauna were based on Thiel (1979) but were changed to the following during model balancing, 12 t/km$^2$ for meiofauna and 11.5 t/km$^2$ for macrofauna. The ratio values for production/biomass (P/B) and consumption/biomass (Q/B) for macrofauna and meiofauna functional groups were taken from Arreguín-Sánchez et al. (2002). Macrofauna values are based on polycheate rates; (P/B of 4 $yr^{-1}$ and a Q/B of 21 $yr^{-1}$). Meiofauna values are a P/B of 8 $yr^{-1}$ and a Q/B of 53 $yr^{-1}$. Following the aforementioned additions, the diet matrix was updated, pre-balance diagnostic tests were run (Link, 2010), and the model was balanced (Table SM1). Refer to supplemental material for the full explanation.

### 2.1.3 Oil Forcing Functions

We created oil response curves to estimate (c), which is a linear scaler on growth efficiency (g) in equation 3. Thus, equation 3 determines the change in group biomass between time steps (dB/dt) while incorporating changes in feeding efficiency from sub-lethal oil impacts. For the remaining representations, *i* refers to the prey, j refers to the predator, *B* is biomass, f() is a functional relationship used to predict consumption rates, M is natural mortality, F is fishing mortality, I is immigration rate out of the ecosystem, and *E* is emigration. A similar approach was used by Ainsworth et al. (2011). Within equation 3, c was calculated from individual dose response models where Z is baseline total mortality from Ecopath basic input, ϴ is a scaling factor on total mortality because of oil exposure, K is the total biomass exposed to oil , and Bh is the total biomass in the habitat area (eq. 4).

(3) Linear scalar on growth efficiency equation (Ainsworth et al., 2011)

$$\frac{dB_i}{dt} = cg_i \sum_{j=1}^{n} f\left(B_j, B_i\right) - \sum_{j=1}^{n} f\left(B_i, B_j\right) + I_i - B_i\left(M_i + F_i + E_i\right) \quad \text{eq. 3}$$

(4) Link to individual dose response models.

$$c = 1 - ([(\theta - Z)/Z] * (K/Bh)) \quad \text{eq.4}$$

This function was entered as a modifier within Ecosim for each month the spill persisted, April – October. However, because the original model included the entire northern Gulf of Mexico (NGOM) the percent of each functional group effected by the spill needed to be calculated (K/Bh eq. 4, Table 3) and then multiplied by percent mortality from the dose response models to generate an accurate forcing function for each group (eq. 4).

Table 3. Calculated model area for each depth range and the area of the surface slick. Visual representation in Figure 1.

| Depth Range (m) | Total Area ($km^2$) | Area Oiled ($km^2$) |
|---|---|---|
| Shoreline | 25,584 (km) | 2113 (km) |
| 0 – 10 | 73,571 | 10,104 |
| 10 – 20 | 57,791 | 7,339 |
| 20 – 50 | 126,029 | 18,574 |
| 50 – 200 | 126,506 | 14,891 |
| 200 – 2000 | 114,831 | 24,981 |
| Carbon Sequestration Depth | | |
| 200 –1000 | 226,205 | 40,486 |

**2.1.3.1 Fish Forcing Functions**

The appropriate fish dose-response model was chosen based on the work of Dornberger et al. (2016), who looked at the impact of the Deepwater Horizon oil spill on the frequency of fish lesions, a proxy for mortality rate. The 'hockey stick' model implied that below a certain oil concentration there were no lethal effects on the population. For our work the following parameters from Dornberger et al. (2016) were input into equation 5, oil threshold ($oil_{thresh}$) = 2.942 ppb and slope ($m$) = 0.1051 $yr^{-1}$. The $oil_{thresh}$ is the oil concentration level above which population-level effects increase log-linearly (Horness et al. 1998; Johnson et al. 2002). In equation 5, ϴ is a scaling factor on total mortality, Z is baseline total mortality from Ecopath basic input, and $m$ is the rate of change in the population response. *Oil* was determined by examining the predicted water column oil concentrations by depth, from the simulations reported in Perlin et al. (2020) (Figure SM1) and the depth ranges of fish groups (Table SM2). In summary, change in biomass over time was determined from eq. 3, c was calculated from eq. 4, and ϴ was calculated from eq. 5 for the fish functional groups.

$$(5)\ \Theta = \begin{cases} Z & if[Oil] < [Oil]_{thresh} \\ Z + m * log[Oil/Oil_{thresh}] & otherwise \end{cases} \qquad \text{eq. 5}$$

**2.1.3.2 Invertebrate Forcing Functions**

Dose response models were used to determine the impact of the Deepwater Horizon oil spill on invertebrates. Affects for shrimp groups came from Echols et al. (2016) and stone crabs and blue crabs was calculated based on DWHNRDA (2016). To determine the impact on meiofauna and macrofauna, the dose response model of Balthis et al. (2017) was used. Effects on macro and micro zooplankton groups came from (Almeda et al. 2013). Full details available in the supplemental material.

**2.1.3.3 Primary Producer Forcing Functions**

The results of studying the impact of oil on phytoplankton have been mixed. Therefore, for our estimates, we combined the phytoplankton findings of Hu et al. (2011) and the toxicity findings of Garr et al. (2014) when we generated the forcing function for phytoplankton. For sea grasses, 95% mortality was assumed within the model in an area of 21.13 $km^2$, which is 8.3% of the entire model shoreline area (Table 4) based on Silliman et al. (2012) and Nixon et al. (2016). The impact on attached microalgae was calculated by using the same exposure response as the phytoplankton, but with the area and exposure time of Seagrass. Full details available in the supplemental material.

Table 4. Values used to calculate the mortality modifier entered under search rate in equation 4 (change in mortality = (Θ - Z)/Z, fraction of population impacted = K/Bh, search rate modifier = c). Overall change = change in mortality*fraction of population impacted (K/Bh). Values below are based on the annual average, but monthly averages were calculated for the model. When a range of oil values was found the average was used. In group name the numbers represent age in months.

| Group name | Change in Mortality | Fraction of Population Impacted | Overall Change | Search Rate Modifier |
|---|---|---|---|---|
| Red Drum (0-3) | 0.23 | 0.13 | 0.030 | 0.9699 |
| Red Drum (3-8) | 0.13 | 0.13 | 0.017 | 0.9825 |
| Red Drum (8-18) | 0.42 | 0.13 | 0.056 | 0.9445 |
| Red Drum (18-36) | 0.77 | 0.13 | 0.10 | 0.8982 |
| Red Drum (36+) | 3.07 | 0.13 | 0.41 | 0.5928 |
| Sea Trout (0-3) | 0.08 | 0.14 | 0.011 | 0.9893 |
| Sea Trout (3-18) | 0.33 | 0.14 | 0.046 | 0.9544 |
| Sea Trout (18+) | 0.65 | 0.14 | 0.091 | 0.9087 |
| Mullet (0-6) | 0.08 | 0.13 | 0.010 | 0.9900 |
| Mullet (6-18) | 0.15 | 0.13 | 0.020 | 0.9796 |
| Mullet (18+) | 0.46 | 0.13 | 0.061 | 0.9389 |
| Mackrel (0-3) | 0.12 | 0.15 | 0.018 | 0.9823 |
| Mackrel (3+) | 0.39 | 0.15 | 0.059 | 0.9409 |
| Ladyfish (0-10) | 0.16 | 0.14 | 0.023 | 0.9772 |
| Ladyfish (10+) | 0.29 | 0.14 | 0.040 | 0.9601 |
| Grouper (0) | 0.23 | 0.13 | 0.030 | 0.9699 |
| Grouper (1-3) | 0.76 | 0.13 | 0.10 | 0.8996 |
| Grouper (3+) | 1.03 | 0.14 | 0.14 | 0.8567 |
| Jacks | 0.58 | 0.14 | 0.080 | 0.9206 |
| Bay Anchovy | 0.18 | 0.14 | 0.025 | 0.9755 |
| Pin Fish | 0.23 | 0.13 | 0.030 | 0.9699 |
| Small fish | 0.27 | 0.13 | 0.035 | 0.9646 |
| Silver Perch | 0.32 | 0.13 | 0.043 | 0.9570 |
| Scaled Sardine | 0.25 | 0.14 | 0.033 | 0.9665 |
| Menhaden Juvenile | 0.18 | 0.14 | 0.025 | 0.9748 |
| Menhaden Adult | 0.24 | 0.14 | 0.034 | 0.9664 |
| Catfish | 0.46 | 0.14 | 0.064 | 0.9361 |
| Caridan Shrimp | 0.20 | 0.14 | 0.029 | 0.9715 |
| Shrimp | 0.20 | 0.14 | 0.027 | 0.9726 |
| Stone Crab | 0.20 | 0.14 | 0.028 | 0.9716 |
| Blue Crab | 0.20 | 0.13 | 0.027 | 0.9734 |
| Pigfish | 0.57 | 0.13 | 0.075 | 0.9247 |
| Rays | 1.52 | 0.14 | 0.21 | 0.7870 |
| Pompano | 0.47 | 0.15 | 0.068 | 0.9319 |
| Lobster | 0.20 | 0.14 | 0.030 | 0.9712 |

| Group name | Change in Mortality | Fraction of Population Impacted | Overall Change | Search Rate Modifier |
|---|---|---|---|---|
| Red Snapper (0-6) | 0.15 | 0.13 | 0.020 | 0.9799 |
| Red Snapper (6-24) | 0.23 | 0.13 | 0.031 | 0.9695 |
| Red Snapper older | 0.77 | 0.13 | 0.10 | 0.8982 |
| Atlantic croaker | 0.30 | 0.13 | 0.040 | 0.9598 |
| Large Coastal Sharks | 1.73 | 0.14 | 0.25 | 0.7554 |
| Benthic Invertebrates | 0.2 | 0.006 | 0.005 | 0.9987 |
| Zooplankton | 0.5 | 0.19 | 0.095 | 0.9046 |
| Phytoplankton | 0.5 | 0.19 | 0.095 | 0.9046 |
| Seagrass | 0.95 | 0.08 | 0.922 | 0.9215 |
| Attached Microalgae | 0.5 | 0.08 | 0.959 | 0.9587 |
| Macro/Meio-fauna | 0.80 | 0.006 | 0.005 | 0.99 |

### 2.2 Observational Data and Ecosim Tuning

The predictability of the model was improved by tuning the model to observational data collected for catch and biomass values throughout the entire Northern GoM from 2004-2014. Fish and shrimp data were obtained from SEAMAP's public database which provides catch per unit effort (CPUE) information for the entire Northern GoM. Bottom line catch information was standardized across surveys by calculating CPUE based on the number of hook hours at each station. Trawl data was standardized across surveys by calculating CPUE based on trawl distance. The data was averaged for each year to correct for differences in the number of surveys and stations sampled. Commercial and recreational landings data were obtained from NOAA's public landings statistics (NMFS, 2016) from 2004 to 2014 and used to tune fisheries yield outputs from the model. Discards that were entered within Ecopath were added to the observational totals. The comparison of modeled to observed DWH effects for fisheries data was performed on biomass values averaged for three years pre- and post- DWH, from 2007 to 2009 and 2010 to 2012 to keep the analysis balanced.

Vulnerabilities were determined by running an optimization routine to minimize discrepancies versus observational data. Vulnerabilities are the degree to which a large increase in predator biomass will lead to predation mortality for a given prey. This was accomplished by vulnerability search where each interaction was selected individually to achieve the best fit against observational data. A separate vulnerability search was run and applied for the no oil and the oil simulations. To further match the model outputs to the observational data, a production anomaly optimization (Christensen et al. 2005) was run and applied to phytoplankton production. Manual calibration of the model was also done in Ecopath to further improve overall model fit. However, in the course of tuning the model, the changes led to decreased ecotrophic efficiency EE for 6-18 mullet (0.04), 18+ mullet (0.02), catfish (0.04), and large coastal shark (0.02).

**2.3 EWE Simulations**

Two simulations starting with initial conditions in 2004, and predicting forward to 2014, were run: (1) no oil and (2) oil. In both simulations fishing was simulated by using the same fleets as Suprenand et al. (2015), which was kept constant based on initial landings entered in Ecopath. Under no oil conditions the baseline tuned Ecosim model was run. To simulate the DWH blowout conditions the oil forcing function values from Table 4 were applied as a modifier to search rate for consumers. In general functions were entered for 2010 (year six in the model) from April – October for all functional groups. The functions were applied at different times for the following groups; from April – June of year seven for meiofauna and macrofauna, from April – June for Zooplankton, from April – September for phytoplankton, and from May – August for seagrass and attached microalgae. The absolute biomass and yield values from both simulations were compared in order to measure potential impact. Ecosim outputs results for absolute biomass are in metric t/ $km^2$. These results were multiplied by the habitat area for each functional group to measure changes in metric tons. The final results are presented as percent change in 2010 and 2011 biomass. This was calculated by subtracting oil scenario values from no oil scenario values for the same year.

### 2.4 Ecosystem Services

To test for changes in ecosystem services a service was assigned following the Millennium Ecosystem Assessment (MEA, 2005) framework. Six functional groups were chosen for further analysis concerning changes in ecosystem services: shrimp, blue crabs, stone crabs, grouper, red snapper, and detritus. For commercial fisheries the model yield outputs by functional group and year from 2008 – 2012 were multiplied by the habitat area in which each group was found, resulting in a value of metric tons. This assumes that fisheries are operating in the entire habitat area of the functional group. Metric tons were then multiplied by the ex-vessel price to get monetary change. The use of ex-vessel prices is important in assessing fisheries management and economic impact (Sumaila et al. 2007). The approach of valuing a fishery using ex-vessel prices, where the focus is primarily on modeled bio-physical changes to the fishery, has been examined with ocean acidification (Cooley and Doney, 2009), ecosystem based management in large marine ecosystems (Christensen et al. 2009), commercial fisheries losses because of closures due to DWH (McCrea-Strub et al. 2011), and commercial fisheries losses estimated up to seven years after DWH (Sumaila et al. 2012).

To determine the change in ex-vessel value of the commercial fisheries yield, the inflation adjusted ex-vessel prices from NMFS (2016) were applied to the model outputs for 2010, 2011, 2012, 2013, 2014. For example, the modeled amount of red snapper catch in pounds was multiplied by the NMFS (2016) price of $3.13 for 2010, $3.20 for 2011, $3.34 for 2012, $3.89 for 2013, and $4.04 for 2014 model output. Dollar values were rounded to two significant digits.

At the ocean surface, atmospheric carbon is taken up by phytoplankton through photosynthesis. When the phytoplankton die, their remains and the carbon they have incorporated sink to the seafloor. When this incorporated carbon cannot return to the atmosphere for at least 100 years or when it reaches depths greater than 1000 m it is considered to be sequestered (Guidi et al. 2015). Once the organic remains reach the seafloor it is called phytodetritus which adds to the organic remains of other organisms collectively called detritus. We used the detrital biomass predicted by the model to calculate how much of the carbon stored within became buried through carbon sequestration. It is important to note that detrital biomass includes not only phytoplankton but 20% from every trophic interaction. This method only takes into account the particulate organic carbon (POC) sequestration therefore the results are only estimating changes in POC sequestration. We determined how much of the atmospheric carbon was sequestered in the deep sea through this process. Guidi et al. (2015) calculated carbon sequestration values for the 56 biogeochemical provinces (Longhurst, 1995), taking into account the amount that is remineralized and never reaches the seafloor. Two sequestration units were provided, (1) sequestration at 2000 m and (2) sequestration at the top of the permanent pycnocline (Guidi et al. 2015). Because our model only extends to 2000 m, we calculated sequestration at the top of the permanent pycnocline, which starts at 200 m and extends to 1000 m (Melvin et al. 2016). Therefore, when we calculated the change in sequestration from the model output, we only considered the area of the model found below 200 m. The Gulf of Mexico is not counted among the 56 biogeochemical provinces considered in Guidi et al. (2015), therefore, we compared two estimates of carbon sequestration, the values for the Gulf Stream (1.81 tg C/yr) and the global value (0.72 pg C/yr). This represents 0.00024% and 0.095% of the total atmospheric carbon, based on a total of 760 Gt (Mcleod et al. 2011).

## 3. Results

### 3.1 Model Fit

Model fit analysis was performed for catch and biomass from 2004-2014. Statistical analysis of catch model fit found MEF was above zero for catfish and Atlantic croaker but below zero for all other catch groups (Table 5). Catch r was at or above 0.5 for catfish, red snapper, and Atlantic croaker (Table 5). The Spearman correlation was at or above 0.5 and significant for pompano and Atlantic croaker (Table 5). Catch RI was close to one and error (RMSE, AE, and AAE) was close to zero for all groups (Table 5). Statistical analysis of relative biomass model fit found MEF was above zero for mullet, ladyfish, grouper, jacks, shrimp, blue crab, red snapper, and Atlantic croaker (Table 6). Biomass r was at or above 0.5 for jacks, pinfish, menhaden, shrimp, blue crab, and Atlantic croaker (Table 6). The Spearman correlation was at or above 0.5 and significant for pinfish and red snapper (Table 6). Biomass RI was close to one for all except pompano and error (RMSE, AE, and AAE) was close to zero for red drum, sea trout, jacks, catfish, and blue crab (Table 6). Graphical representation of the comparison of predicted catch and predicted relative biomass to time series data can be found in the supplemental material (Figures SM2-SM8).

Table 5. Model skill metrics for catch data. r = correlation coefficient. RMSE = root mean squared error. RI = reliability index. AE = average error, AAE = average absolute error. MEF = modeling efficiency. * = P-value below 0.05.

| Metric | r | RMSE | RI | AE | AAE | MEF | Pearson | Spearman | Kendall |
|---|---|---|---|---|---|---|---|---|---|
| Red Drum 18-36 | -0.3135 | 0.0016 | 1.0536 | 0.0002 | 0.0011 | -0.3646 | -0.3135 | -0.2091 | -0.1273 |
| Sea Trout 18+ | -0.2460 | 0.0024 | 1.0511 | -0.0015 | 0.0021 | -1.2831 | -0.2460 | -0.4636 | -0.2727 |
| Mullet 18+ | -0.2548 | 0.0003 | 1.0198 | 0.0001 | 0.0002 | -0.2668 | -0.2548 | -0.2364 | -0.0909 |
| Mackrel 3+ | 0.3447 | 0.0005 | 1.0049 | 0.0002 | 0.0004 | -0.6193 | 0.3447 | 0.1909 | 0.1636 |
| Ladyfish 10+ | 0.0627 | 0.0000 | 1.0231 | 0.0000 | 0.0000 | -0.0396 | 0.0629 | -0.2091 | -0.1636 |
| Grouper 3+ | -0.7638 | 0.0009 | 1.0562 | -0.0002 | 0.0006 | -0.3222 | -0.7638* | -0.6818* | -0.5636* |
| Jacks | 0.3437 | 0.0004 | 1.0058 | -0.0001 | 0.0003 | -0.0214 | 0.3437 | 0.3455 | 0.3091 |
| Bay Anchovy | 0.6255 | 0.0000 | 1.0001 | 0.0000 | 0.0000 | 0.4858 | 0.9991* | 1* | 1* |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Pin Fish | 0.9995 | 0.0000 | 1.0000 | 0.0000 | 0.0000 | 0.9990 | 0.9995* | 1* | 1* |
| Silver Perch | 0.1128 | 0.0000 | 1.0358 | 0.0000 | 0.0000 | -0.1704 | 0.1105 | 0.0547 | 0.0734 |
| Scaled Sardine | -0.2777 | 0.0006 | 1.0043 | 0.0002 | 0.0004 | -5.1610 | -0.2777 | -0.4273 | -0.3455 |
| Menhaden | 0.9544 | 0.0001 | 1.0000 | 0.0000 | 0.0000 | 0.9174 | 1* | 1* | 1* |
| Catfish | 0.4894 | 0.0003 | 1.0040 | -0.0001 | 0.0002 | 0.0225 | 0.4894 | 0.4182 | 0.2364 |
| Shrimp | -0.2571 | 0.0246 | 1.0146 | 0.0192 | 0.0213 | -3.8975 | -0.2571 | -0.4182 | -0.2727 |
| Blue Crab | 0.2618 | 0.0065 | 1.0158 | 0.0039 | 0.0056 | -0.4360 | 0.2618 | -0.2727 | -0.2364 |
| Pigfish | -0.1043 | 0.0000 | 1.0052 | 0.0000 | 0.0000 | -0.3337 | -0.1042 | -0.1182 | 0.0182 |
| Pompano | 0.7488 | 0.0003 | 1.1864 | -0.0002 | 0.0002 | -2.7815 | 0.7488* | 0.6818* | 0.5273* |
| Lobster | 0.2020 | 0.0004 | 1.0094 | -0.0002 | 0.0003 | -0.1750 | 0.2020 | 0.2546 | 0.1636 |
| Red Snapper older | 0.4864 | 0.0010 | 1.0191 | 0.0005 | 0.0009 | -0.1029 | 0.4864 | 0.5636 | 0.4546 |
| Atlantic Croaker | 0.7193 | 0.0001 | 1.0162 | -0.0001 | 0.0001 | 0.0796 | 0.7193* | 0.7364* | 0.5636* |

Table 6. Model skill metrics for relative biomass data. r = correlation coefficient. RMSE = root mean squared error. RI = reliability index. AE = average error, AAE = average absolute error. MEF = modeling efficiency. * = P-value below 0.05.

| Metric | r | RMSE | RI | AE | AAE | MEF | Pearson | Spearman | Kendall |
|---|---|---|---|---|---|---|---|---|---|
| Red Drum 18-36 | -0.2937 | 0.3345 | 1.4492 | -0.3153 | 0.3153 | -8.4387 | -0.2937 | -0.2329 | -0.1667 |
| Sea Trout 18+ | 0.2527 | 0.0533 | 1.0144 | -0.0065 | 0.0430 | -0.0256 | 0.2527 | 0.2466 | 0.1667 |
| Mullet 18+ | 0.1137 | 3.2481 | 1.0571 | 3.0238 | 4.4290 | 0.1621 | -0.1244 | -0.3000 | -0.2000 |
| Ladyfish 10+ | -0.3090 | 0.8365 | 1.0539 | 0.4603 | 0.9038 | 0.0542 | 0.1843 | -0.5798 | -0.4140 |
| Grouper 3+ | -0.0720 | 0.8140 | 1.0898 | -0.0444 | 0.6051 | 0.0214 | 0.2271 | 0.4667 | 0.3889 |
| Jacks | 0.4946 | 0.0187 | 1.0417 | -0.0010 | 0.0124 | 0.0838 | 0.5350 | 0.4438 | 0.2697 |
| Bay Anchovy | -0.4411 | 0.6078 | 1.0735 | -0.0685 | 0.4775 | -0.1760 | -0.4411 | -0.0455 | -0.0182 |
| Pin Fish | 0.6625 | 5.0661 | 1.1454 | -1.7467 | 3.6910 | -0.0838 | 0.6625* | 0.7909* | 0.6364* |
| Silver Perch | 0.0907 | 0.7395 | 1.3816 | -0.2883 | 0.5299 | -0.1447 | 0.6625 | 0.7909 | 0.6364 |
| Menhaden | 0.5857 | 4.0567 | 1.0951 | -1.1944 | 2.4033 | -0.0176 | 0.5857 | 0.3455 | 0.2727 |
| Catfish | 0.2305 | 0.0966 | 1.0631 | -0.0238 | 0.0765 | -0.0155 | 0.2305 | 0.1185 | 0.1101 |
| Shrimp | 0.4967 | 1.3381 | 1.0666 | -0.2722 | 0.6721 | 0.0206 | 0.4967 | 0.0636 | -0.0182 |
| Blue Crab | 0.4872 | 0.0979 | 1.0275 | -0.0174 | 0.0652 | 0.0582 | 0.4872 | 0.4091 | 0.3091 |
| Pigfish | 0.3344 | 0.3186 | 1.1868 | -0.1058 | 0.2498 | -0.0913 | 0.3344 | 0.2909 | 0.1636 |
| Pompano | 0.2728 | 1.3968 | 1.8460 | -0.7100 | 0.8239 | -0.1830 | 0.1732 | 0.4603 | 0.1972 |
| Red Snapper older | -0.1266 | 0.4762 | 1.0415 | 0.0750 | 0.4095 | 0.0217 | 0.4417 | 0.7500* | 0.5556 |
| Atlantic Croaker | 0.4988 | 0.4879 | 1.0103 | 0.0343 | 0.4398 | 0.1166 | 0.4988 | 0.2636 | 0.2364 |

### 3.2 Senarios

When the absolute biomass output from no oil conditions was compared to the oil simulation, biomass decreased in the DWH blowout simulation for 16 and increased for 33 groups in 2010 (Table 7). The percent changes in biomass were larger in 2011 than in 2010 (Table 7). The highest percent change in 2010 was seen in Atlantic croaker with a decrease of

23.78% compared to a 44839% increase in mullet (18+) in 2011 (Table 7). The mullet functional group had high error values when compared to observational data (Table 6).

Table 7. Change in absolute biomass between scenarios (spill output-normal output) across the entire model area in the Northern Gulf of Mexico for 2010 and 2011. Percent values represent percent change. In group name, the numbers represent age in months.

| Group name | 2010 Biomass Change (Metric Tons) | 2011 Biomass Change (Metric Tons) |
|---|---|---|
| Atlantic croaker | -94593.36 (-23.78%) | -3571284.68 (-91.86%) |
| Attached Microalgae | 0.99 (0.01%) | 5.03 (0.07%) |
| Bay Anchovy | -4055.98 (-1.29%) | 38908.04 (13.03%) |
| Benthic Invertebrates (Entire) | 36020.41 (0.25%) | -107438.92 (-0.75%) |
| Benthic Invertebrates (Offshore) | 16166.81 (0.25%) | -48221.11 (-0.75%) |
| Blue Crab | 275.07 (0.90%) | 30617.46 (2338.59%) |
| Caridan Shrimp | 64157.63 (1.68%) | 2065459.90 (109.16%) |
| Catfish | 4862.80 (20.63%) | -44507.70 (-57.15%) |
| Detritus (Entire) | -128426.42 (-0.20%) | -96457.09 (-0.15%) |
| Detritus (Offshore) | -57640.80 (-0.20%) | -43292.21 (-0.15%) |
| Grouper (0) | 16.47 (2.28%) | -551977.68 (-100%) |
| Grouper (1-3) | 262.12 (9.46%) | -432077.10 (-99.19%) |
| Grouper (3+) | 1397.26 (0.49%) | 165795.73 (113.49%) |
| Jacks | 111.51 (1.20%) | -5822537.80 (-99.83%) |
| juv Menhaden | 7616.27 (13.92%) | 31265.35 (71.60%) |
| Ladyfish (0-10) | -591.86 (-2.95%) | 16878.40 (288.45%) |
| Ladyfish (10+) | 118.66 (0.07%) | -72100.00 (-26.55%) |
| Large Coastal Sharks | 8953.07 (0.16%) | -78864965.57 (-93.12%) |
| Lobster | -1439.81 (-0.26) | -5516931.88 (-90.29%) |
| Mackerel (0-3) | -0.29 (-0.38%) | -12042.99 (-99.26%) |
| Mackerel (3+) | -3279.88 (-2.53%) | -192145.73 (-56.30%) |
| Macrofauna (Entire) | 13666.28 (0.19%) | 10725.59 (0.14%) |
| Macrofauna (Offshore) | 6133.75 (0.19%) | 4813.90 (0.14%) |
| Macrozooplankton (Entire) | 670837.41 (10.74%) | 436667.54 (8.10%) |
| Macrozooplankton (Offshore) | 301087.60 (10.74%) | 195986.65 (8.10%) |
| Meiofauna (Entire) | 39260.05 (0.51%) | 38863.48 (0.52%) |
| Meiofauna (Offshore) | 17620.83 (0.51%) | 17442.84 (0.52%) |
| Menhaden | 126434.81 (17.91%) | 806010.23 (1293022%) |
| Microzooplankton (Entire) | -279178.91 (-5.65%) | -213488.65 (-4.66%) |
| Microzooplankton (Offshore) | -125302.06 (-5.65%) | -95818.72 (-4.66%) |
| Mullet (0-6) | -152.86 (-2.36%) | -5419.00 (-48.44%) |
| Mullet (18+) | -6055.21 (-0.60%) | 977430.28 (44839.65%) |
| Mullet (6-18) | -10418.69 (-5.65%) | -84166.37 (-28.92%) |
| Phytoplankton (Entire) | 8748.86 (0.05%) | 5924.09 (0.04%) |
| Phytoplankton (Offshore) | 3926.69 (0.05%) | 2658.87 (0.04%) |

| Group name | 2010 Biomass Change (Metric Tons) | 2011 Biomass Change (Metric Tons) |
|---|---|---|
| Pigfish | -1332.11 (-3.61%) | 9946.148 (34.97%) |
| Pin Fish | 7169.72 (1.38%) | 177914.06 (47.04%) |
| Pompano | 4762.64 (9.81%) | -1657211.28 (-97.07%) |
| Rays | -2987.53 (-0.31%) | 830826.17 (512.36%) |
| Red Drum (0-3) | 0.93 (1.57%) | 43.96 (350.46%) |
| Red Drum (18-36) | 12374.92 (8.41%) | 151522.71 (877.42%) |
| Red Drum (3-18) | 194.83 (3.81%) | -5589.50 (-51.87%) |
| Red Drum (36+) | 12479.34 (1.11%) | 1019184.55 (485.43%) |
| Red Drum (8-18) | 1207.99 (3.26%) | -20006.48 (-31.81%) |
| Red Snapper (0-6) | 56.35 (4.23%) | -3062831.48 (-99.96%) |
| Red Snapper (6-24) | 3748.98 (5.10%) | -3302749.22 (-97.39%) |
| Red Snapper older | 9820.93 (4.46%) | -2626608.13 (-91.18%) |
| Scaled Sardine | -12090.52 (-0.75%) | 1646153.92 (1146.85%) |
| Sea Grass | 2.64 (0.01%) | 0.79 (0.002%) |
| Sea Trout (0-3) | 1.97 (7.81%) | -664356.81 (-100%) |
| Sea Trout (18+) | 4218.22 (10.54%) | -30079.98 (-38.91) |
| Sea Trout (3-18) | 745.21 (11.90%) | 8270.92 (18537.73%) |
| Shrimp | 47864.59 (11.09%) | 477539.68 (1356.33%) |
| Silver Perch | 660.88 (1.72%) | -1003832.91 (-95.75%) |
| Small fish | -2411.14 (-0.59%) | 425515.03 (3056.08%) |
| Stone Crab | -32753.25 (-12.34%) | -315431.37 (-55.80%) |

### 3.3 Model Fit to observed DWH effects

Percent change in observed biomass pre (2007 – 2009) and post the DWH (2010 – 2012) varied by functional group (Figure 3). Percent change was larger within the observational data than the model outputs. Directional changes were in agreement for 62% of the functional groups that had observational time series data available pre and post the DWH (Figure 3). These groups included blue crabs, red snapper older, Atlantic croaker, pinfish, shrimp, menhaden, catfish, grouper 3+, and bay anchovy.

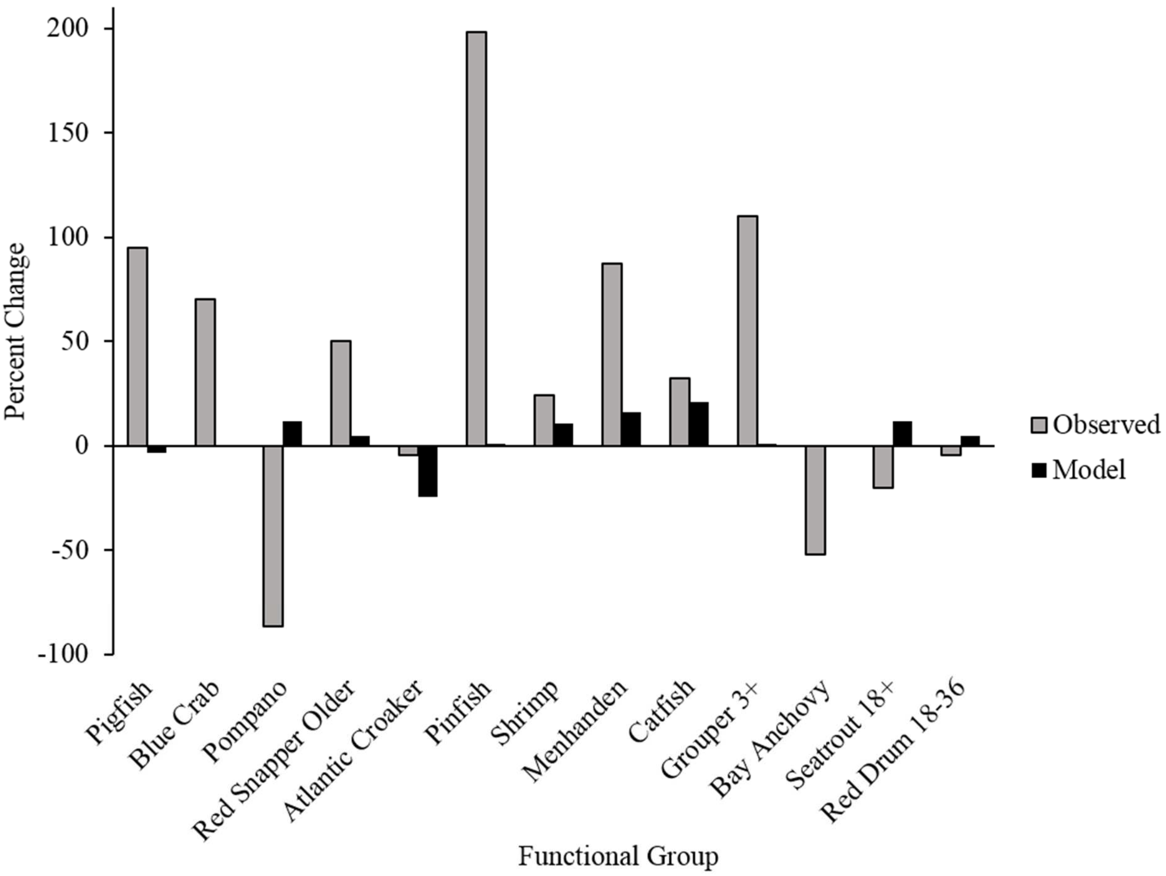


Figure 3. Percent change in SEAMAP absolute biomass and model outputs based on the average of (2010 – 2012) minus the average of (2007 – 2009).

### 3.4 Ecosystem Services

Monetary valuation was performed for yield outputs of grouper, red snapper, shrimp, stone crab, and blue crabs. The yields for all groups except stone crab were higher in the spill scenario (Table 8). The change in yield for each resulted in estimated monetary changes ranging from $18 million to -$15 million in 2010 (Table 9).

Table 8. Yield outputs from the no-oil (N) and oil scenarios (O). The pre-spill years of 2008 and 2009 are included to show the natural model trends in yield. Yields are in Metric tons.

| | Functional Group | | | | |
|---|---|---|---|---|---|
| Simulation | Grouper | Red Snapper | Shrimp | Stone Crab | Blue Crab |

| | | | | | |
|---|---|---|---|---|---|
| 2008 | 589 | 4750 | 37155 | 11465 | 3305 |
| 2009 | 609 | 4874 | 37006 | 11439 | 3284 |
| 2010 (O) | 619 | 5281 | 44200 | 10471 | 3455 |
| 2010 (N) | 616 | 5033 | 39789 | 11944 | 3424 |
| 2011 (O) | 669 | 5991 | 47256 | 11240 | 3573 |
| 2011 (N) | 660 | 5652 | 42761 | 12910 | 3560 |
| 2012 (O) | 685 | 5486 | 41440 | 10346 | 3328 |
| 2012 (N) | 680 | 5188 | 37621 | 11958 | 3310 |
| 2013 (O) | 665 | 5429 | 44299 | 10457 | 3448 |
| 2013 (N) | 664 | 5142 | 39803 | 12026 | 3412 |
| 2014 (O) | 702 | 6253 | 48666 | 11397 | 3615 |
| 2014 (N) | 696 | 5878 | 43964 | 13129 | 3598 |

Table 9. Difference in ex-vessel value in commercial fisheries from no-oil to oil simulations. Monetary estimates are the difference from oil simulation outputs – no oil simulation outputs.

| Functional Group | 2010 | 2011 | 2012 | 2013 | 2014 |
|---|---|---|---|---|---|
| Grouper (all relevant species) | $18,000 | $57,000 | $35,000 | -$300,000 | $3,600 |
| Red Snapper | $1,700,000 | $2,400,000 | $2,200,000 | -$7,700,000 | $2,500,000 |
| Shrimp (all species) | $18,000,000 | $20,000,000 | $16,000,000 | -$83,000,000 | $25,000,000 |
| Stone Crab | -$15,000,000 | -$16,000,000 | -$16,000,000 | -$30,000,000 | -$23,000,000 |
| Blue Crab | $68,000 | $26,000 | $38,000 | -$3,100,000 | $100,000 |

To determine how POC sequestration could have been altered following the DWH blow out, the change in the amount of detritus in the offshore environment (200 – 2000 m) was measured from the model outputs. When compared to the no oil simulation, this resulted in a detrital decrease of 57,640.80 metric tons in 2010, and a decrease of 43,292.21 metric tons in 2011. When carbon sequestration percentages were applied to the model outputs, sequestration in 2010 decreased by 0.15% which equated to 33.89 metric tons based on the average global sequestration rate (Guidi et al. 2015), and 0.09 metric tons based on the Gulf Stream sequestration rate (Guidi et al. 2015), in 2010. In 2011, sequestration increased by 0.06% equating to 12.98 metric tons (Global average) and 0.03 metric tons (Gulf Stream rate).

## 4. Discussion

It is always important in environmental assessment to be able to understand how an event affects people. One way to do that is to translate biophysical impacts into ecosystem service impacts. In the case of Natural Resource Damage Assessments (NRDA), there is a need to monetize due to legal obligations for damage assessment. Therefore, methods exist to transform sampling data to lost economic value (NAS, 2012). Here, an approach is presented that quantifies how offshore ecosystem services were affected by the DWH. Importantly, the methods here allow for valuation of indirect benefits, such as the maintained availability of prey items on which exploited species depend.

### 4.1 Overall Model Fit

Overall, the model had an ideal RI, consistently predicting catch and relative biomass with an average multiplicative factor of one (Tables 5 and 6). Pompano relative biomass was the only exception (Table 6). The other functional groups differed in their objective MEF, the tendency to vary with the observational data (correlation), and their prediction accuracy (error) (Tables 5 and 6). The model performed best overall when predicting catfish catch, Atlantic croaker catch, jacks relative biomass, blue crabs relative biomass, and Atlantic croaker relative biomass. However, in addition, the model had an above average MEF when predicting pompano catch, mullet relative biomass, ladyfish relative biomass, grouper relative biomass, shrimp relative biomass, and red snapper relative biomass (Tables 5 and 6). Therefore, the model produced good reliable predictions for 15% of the catch groups and 47% of the relative biomass groups with the time series observational data available.

Tuning of the model to match observational trends resulted in low EE values for 4 functional groups. The EE value for LC shark was low (0.02) but it may be justifiable as there are few predators on large sharks. However, low EE values for mullet [6 – 18 mullet (0.04) and 18+ mullet (0.02)] and catfish (0.04) reflect that there is predation on these groups not identified in the available diet information. Further revisions to the model should work to identify additional potential predators, although benthic diet data is limiting. Thus, the sensitivity of mullet and catfish to top-down trophic effects may be conservatively estimated. This model is based on a simplified Gulf of Mexico food web, which could explain the differences seen with the observational data and low EE values, not all diet interactions are represented. In particular, the absence of sea birds and dolphins who are important predators of many fish groups.

**4.2 Differences between simulations**

The DWH simulation showed an overall positive impact on functional group biomass percent change. The most negatively impacted group was the Atlantic croaker in 2010 with a decrease of 23.78% (Table 7). In 2011, the values showed a greater percent change when compared to the no oil simulation. It is surprising that some functional groups that previously showed a slight decline in biomass in 2010 increased dramatically in 2011 (Table 7). This increase in biomass was because of a decrease in predation pressure on these groups in the oil spill simulation, fishing effort was not changed in the simulation. As seen in Table 7 mackrel, jacks, red snapper older, LC sharks, red drum, sea trout, and catfish decreased in biomass in the 2011 oil simulation.

**4.3 Model Fit to observed DWH effects**

Model predictions agreed with observational data with regard to directional changes but underestimated magnitude of change. Directional trends were in agreement for 63% of the functional groups that had observational data available. These groups included blue crabs, red snapper older, Atlantic croaker, pinfish, shrimp, menhaden, catfish, grouper 3+, bay anchovy, meiofauna, and macrofauna (Figure 3). Baguley et al. (2015) found that meiofauna abundance increased in offshore areas approximately 5 months after the spill, with increases ranging from 104 – 197%. In the DWH scenario, meiofauna biomass increased immediately after the spill and reaches 0.75% above no oil conditions in 2010. Washburn et al. (2016) found that macrofauna abundance decreased 30 – 85% in the highly impacted zone of the DWH spill. In the model, macrofauna biomass decreased 0.12% 6 months after the spill then increased the following year. When model outputs are compared with survey data, the majority of the functional groups should match the biomass trends direction, but not necessarily magnitude (Kaplan and Marshall, 2016). This is the case with the DWH model, 63% of the functional groups match the directional changes in the observational data after the DWH oil spill (Figure 3).

The model does not take into account other sources of disturbance such as red tide, hypoxia, harmful algal blooms, and hurricanes which would have compounded the impacts from the DWH oil spill. In addition, changes were made to the observational sampling effort. There were more stations sampled with SEAMAP from 2007 – 2009 then from 2010 – 2012. SEAMAP adopted new sampling methods in 2010 that could have increased the capture rate. In addition to other sources of disturbance, differences in observed versus modeled outputs maybe attributed to model discrepancies because of model structure and parameter error. The discrepancies include: one general fisheries dose response model, area of impact calculation, and no fisheries closures included.

Simulated impact of oil toxicity on fish functional groups was based on one general dose-response model, which has been informed by the current knowledge regarding oil spill impacts on fish populations. The intensity of the toxic affect depends on the fish species, the life stage, the oil concentration, and the oil composition (Mosbech, 2002; McCay et al. 2004; and Incardona et al. 2011). In addition, oil exposure is not always associated with an immediate lethal outcome (for example see, Heintz et al. 2000; Incardona et al. 2013; and Incardona et al. 2014). Toxic effects of oil exposure in fish include cardiac toxicity (Incardona et al. 2013; Incardona et al. 2014: Incardona & Scholz, 2015; Morris et al. 2015a; Morris et al. 2015b), reduced growth (Ortell et al. 2015), reduced immune function (Ortell et al. 2015), and reduced swim performance (Mager et al. 2014; Morris et al. 2015b). For the model, it was presumed that the effects of oil exposure likely led to death by indirectly impacting their survival rates (Refer to Moles and Norcross, 1998; Meador et al. 2006).

The area of the model spans the entire Northern GoM. To account for the small portion of this area impacted by the DWH spill, the proportion of the population affected was accounted for when calculating the forcing function. The area of impact in the water column was calculated using the area of the surface oil slick. However, oil entrained at depth likely spread differently below the surface (Paris et al. 2012, Le Hénaff et al. 2012). Nevertheless, using the surface slick may be appropriate because the organic flocculent and hydrocarbons mixed near the surface provided a conduit for benthic deposition (Schwing et al. 2015). In considering the surface slick, the mass accumulation at the bottom was also considered. A comparison of the two approaches would be informative.

**4.4 Ecosystem Service Changes**

While some oil spill models focus on the physical and chemical aspects of a spill, others focus on the biological/ecological aspects (Okey and Pauly, 1998; French-McCay, 2004; McCay, 2003; Afenyo et al. 2017; Carroll et al. 2018; Ainsworth et al. 2018). Fewer still have examined changes in ecosystem services following the DWH (Washburn et al. 2018). The model presented here is the first EWE model to simulate the effects of the DWH oil spill while valuing ecosystem services.

Of the ecosystem services valued, the greatest impact was seen within the commercial stone crab industry with an estimated ex-vessel loss of $15 million in 2010. This value is lower than the potential minimum loss of $247 million based on fisheries closures and the visual extent of the oil spill using ex-vessel price information (McCrea-Strub et al. 2011). It is also lower than a seven-year loss projection of $1.6 billion in commercial fisheries total revenue losses (Sumaila et al. 2012). It is not surprising that these other estimates are higher than the current model because they consider economic loss resulting from fisheries closures and total revenue. Following the Deepwater Horizon blowout, fisheries closures were implemented to aid in the recovery. These closures were not implemented in the current version of the model and could have affected biomass changes seen in the EWE simulation for the oil spill scenario. In addition, the current model underestimated changes in biomass in the DWH simulation thereby underestimating catch.

The loss in the ability of the system to sequester carbon brings forth interesting social impacts. One way to illuminate the loss of this service is by applying the social cost of $CO_2$ to the change in sequestration. The social cost of $CO_2$ is a monetary estimate of the damages associated with the increasing carbon emissions (IWGSCC, 2015). The value includes changes in human health, property damage from increased flood risk, net agricultural productivity, and the value of ecosystem services because of climate change (IWGSCC, 2015). The IWGSCC (2015) value of $36 per metric ton of $CO_2$ (3% discount rate) was applied to the model output. Dollar costs were rounded to two significant digits and given the estimated decrease in sequestration in 2010 of 33.89 metric tons (Global average) and 0.09 metric tons (Gulf Stream rate), this is equivalent to a social cost loss of $1200 and $3 respectively. There was an increase in sequestration in 2011 equating to 12.98 metric tons (Global average) and 0.03 metric tons (Gulf Stream rate), this is equivalent to a social cost gain of $470 and $1 respectively.

**4.5 Comparisons to Published Oil Spill Models**

Model simulations of a North Cape oil spill on the south coast of Rhode Island simulated biological effects using direct mortality and lost production over a one-hundred day period (McCay, 2003). The North Cape oil spill simulation predicted up to a 40% loss in average sensitivity species and up to 90% loss in sensitive species for demersal fish and invertebrates (McCay, 2003). Percent loss was lower in the 2010 EWE DWH simulation with the greatest percent loss seen within Atlantic croaker (23%) (Table 7). An oil spill model of the Northeast Artic Cod fishery found a maximum of a 12% decrease in adult cod biomass within 90 days using direct mortality (Carroll et al. 2018). This decrease is closer to the decreases seen in the 2010 EWE DWH simulation but there are no directly comparable functional groups to Artic cod (Table 7). An Atlantis model of the DWH oil spill simulated oil effects with direct mortality and growth reductions (Ainsworth et al. 2018). The Atlantis model predicted in areas most heavily impacted that the biomass of large reef fish decreased by 25-50% and large demersal fish decreased by 40-70% with the largest decreases occurring 7-16 months after the spill. That estimate employed the same Perlin et al. (2020) oil model concentrations and the same "hockey stick" model from Dornberger et al. (2016). The EWE 2010 DWH simulation predicted up to a 3% decrease in reef fish (pigfish) and up to a 24% decrease in demersal fish (Atlantic croaker) (Table 7). The Atlantis model also predicted a general reduction in 2011 of catch from 20-40%

(Ainsworth et al. 2018). Overall catch had a percent change increase of 5% in the EWE DWH simulation from 2010 to 2011 but percent change decreased by 264% from 2011 to 2012.

The EWE DWH model predictions of biomass loss were lower than other published oil spill models, and often led to increased biomass instead. The most likely reason for this is that the oil effects were applied only as a forcing function on consumer search rate and not through direct mortality or a combination of the two. This approach only takes into account the sublethal effects of the DWH oil spill occurring on growth rate. Similar approaches were used to simulate climate change impacts (Ainsworth et al. 2011; Suprenand and Ainsworth 2017). Other approaches to simulating mortality effects include direct mortality which is often achieved in EWE by using a pseudo fishery (McCay, 2003; Ainsworth et al. 2018; Carroll et al. 2018; DiLeone and Ainsworth, 2019). Thus, population impacts estimated here may be conservative. Representing both lethal and sublethal effects may provide more accurate predictions because both occur in an oil spill and future model revisions should compare the two.

**4.6 Improvements to the Current Biophysical Model**

The EWE model predictions did not correspond to directional trends and magnitude changes in observational data for all functional groups. One way to improve this is to develop specific dose-response curves for each functional group instead of using one dose response value for all fish groups. This will only be possible once more data on the response of individual fish species becomes available. Second, seasonal effects can be added. For example, research has suggested that changes within zooplankton are variable and depend on the time of the year (Carassou et al. 2014). The same is true for phytoplankton and sub sequentially benthic fauna whose main food sources are the phytoplankton that fall from the surface. Improvements can also be made to the calculations for the area of the water column effects, which could be done by developing a geographic information system (GIS) layer showing oiling area by depth and implementing improved oil model results.  The marine oil snow sedimentation and flocculent accumulation (MOSSFA, Daly et al. 2016) event was not included.  Instead, the model only estimated changes in detritus as a result of normal ecosystems process.

## 5. Conclusion

The approach presented here is an important step towards understanding and valuing changes to ecosystem services.  Despite discrepancies between observed and predicted results, the model and the methods employed provide valuable tools that can applied to any EWE oil spill model.  This approach can be applied to different perturbations and different environments.  Ecopath is the preferred tool for fisheries in the European Union (Fretzer, 2016).  Here, it is demonstrated that Ecopath is a valuable tool to resource managers and decision makers because it can estimate changes in ecosystem services.

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