The Effect of the DeepWater Horizon Oil Spill on Human Wellbeing in the Gulf of Mexico

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Table of Contents

i. Abstract

It's always important in environmental assessment to be able to understand how an event has an effect on people. The only way to do that is to translate biophysical impacts to ecosystem service impacts. This approach was taken in order to determine how the Deepwater Horizon (DWH) blowout impacted ecosystem services in the Gulf of Mexico. An *Ecopath with Ecosim* model was developed for the Northern Gulf of Mexico that incorporated three ecosystem services: commercial fisheries, recreational fisheries, and carbon sequestration. The model predicted an overall decrease in all three services investigated. Changes in commercial fisheries and carbon sequestration were valued by linking the model outputs to monetary valuation models. With regards to commercial fisheries the change in fisheries yield resulted in monetary changes ranging from $65 to -$5,091,109 in one year. Investigation of carbon sequestration predicted up to an $876,583 loss in the ability of the Northern Gulf of Mexico offshore environment to sequester carbon. This project has provided the first estimates of ecosystem services in an offshore environment and evaluated their changes as a result of DWH accident.

ii Acknowledgements

Many people helped in completing this project. At Texas A&M University-Corpus Christi (TAMUCC), Ph.D. Candidate Melissa Rohal had a major role in constructing, calibrating, and running the model, and in writing this report. This report will be a chapter in Melissa's dissertation. Several other TAMUCC students helped to obtain biomass and community structure data, this included undergraduate student Tiffany Hawkins, and graduate students Meagan Hardegree, Amanda Gordon, and Travis Washburn. Elani Morgan (TAMUCC) helped with data management. Brach Lupher (TAMUCC) helped with GIS analysis to determine the Gulf of Mexico area calculations by depth and oil overlap. The model originated from Cameron Ainsworth (University of South Florida) and Paul Supernanad (Mote Marine Laboratory). Claire Paris-Limouzy and Natalie Perlin (University of Miami) provided information on Macondo oil concentrations by water depth.

This report fulfills all the original objectives of the proposal.

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1. Introduction

The increasing natural and anthropogenic pressures on offshore marine ecosystems highlights the need to understand and verify the tradeoffs among ecosystem services in order to establish appropriate management strategies (NRC, 2012). Ecosystem services are the direct and indirect contributions from ecosystems that support, sustain, and enrich human life (Yoskowitz, 2014; Carollo, 2013). The Millennium Ecosystem Assessment (MEA, 2005) created a framework to classify ecosystem services into four categories: provisioning, regulating, cultural, and supporting. Work to date on offshore ecosystem services has focused on high visibility services such as tourism and commercial and recreational fisheries (for examples see White et al., 2012; Worm et al., 2006), identifying potential goods or services (Werner et al., 2014; Armstrong et al., 2010), and assessing stakeholder value (Yoskowitz et al., 2014). The importance of offshore functions and processes, including, oil and gas production is not well understood, with the exception of the potential benefits of platforms providing habitat (Helvey, 2002; Page et al., 2006).

It is now recognized that informed management decisions and policy need measures linking human actions to changes in ecosystem functions and inevitably to changes in human well-being (NRC, 2012). Human well-being is a complex concept that is comprised of several components or domains. According to The Millennium Ecosystem Assessment (MEA, 2005) there are five constituents of human well-being: security, basic material for a good life, health, good social relations, and freedom of choice and action. In 2012 the U.S. Environmental Protection Agency (EPA) proposed a human well-being index for the U.S. based upon eight domains: social cohesion, education, connection to nature, health, living standards, leisure time, safety and security, and cultural fulfillment. More recently, the National Oceanic and Atmospheric Administration (NOAA) released a technical memorandum (Dillard et al., 2013) describing the indicators necessary to monitor well-being in Gulf of Mexico (GoM) coastal communities; these indicators are: social connectedness, economic security, basic needs, health, access to social services, education, safety, governance- management and planning, and environmental condition. A better understanding of the links between offshore environments, ecosystem services, and human well-being will aid policy makers and managers in their decision-making. For these strategies to be effective there is the need to better understand the dynamics of complex systems, such as the deep Gulf of Mexico (NRC, 2013).

The Deepwater Horizon (DWH) event highlighted the need to identify and quantify the ecosystem services provided by the GoM offshore ecosystem to fully understand the impacts on human well-being. The goal of this project was to begin to explicitly connect the structure, function, and processes of the offshore environment to human well-being in a manner that is scalable and transferable, while considering the potential changes in the system as a result of the DWH event in the GoM. This was achieved by: 1) building an ecosystem model considering the “system” as a whole, rather than a particular bio-physical feature or habitat (e.g., banks, reefs, etc.), 2) running simulations to test how biomass has changed because of the DWH blowout, 3) improving the model’s predictability to reflect observational data, and 4) incorporating ecosystem services into the model to determine how services have changed following the DWH blowout.

2. Methods

To provide understanding of the linkages between offshore environmental impacts and human well-being following the Deepwater Horizon (DWH) blow out, an *Ecopath with Ecosim* (*EWE*) model of the Northern Gulf of Mexico was built incorporating ecosystem services. *EWE* utilizes a trophic flows model based on the mass-balance fluxes of biomass, which has been combined with approaches for the analysis of flows between the elements of ecosystems (Christensen et al., 2005). The foundation of the Ecopath models is formed by two equations that represent production (Production = Catches + Predation Mortality +Net Migration + Biomass Accumulation + Other Mortality) and consumption (Consumption = Production + Respiration +Unassimilated Food) (Christensen et al., 2005). A trophic flow approach enables consideration of the whole ecosystem from phytoplankton, to detritus, to benthos, to fish (Christensen et al., 2005). Within *EWE* there are three main linked routines, *Ecopath*, *Ecosim*, and *Ecospace*. *Ecopath* is a static mass-balance picture of the ecosystem, *Ecosim* allows for the representation of temporal dynamics, and *Ecospace* is a spatial and temporal dynamic module (Christensen et al., 2005). The *Ecopath* and *Ecosim* routines were used in the development of our model.

2.1 Model Simulation

Our model was built by expanding upon the spatial-temporal *EWE* model of the northern Gulf of Mexico developed by Suprenand et al. (2015). The domain of the model ranges from 24-31°N latitude to 80-98°W longitude with depths ranging from 0-2000 m (Figure 1). The focus of the model is based on offshore ecosystem services but, nearshore and offshore are linked therefore, shallower depths were include to consider the system as a whole. We added Meiofauna and Macrofauna functional groups, oil forcing functions, and ecosystem service categories. Two temporal simulations starting with initial conditions in 2004 and predicting forward to 2014 were run, (1) normal conditions and (2) DWH blowout. To improve the predictive power of the model it was compared to observational data when possible.

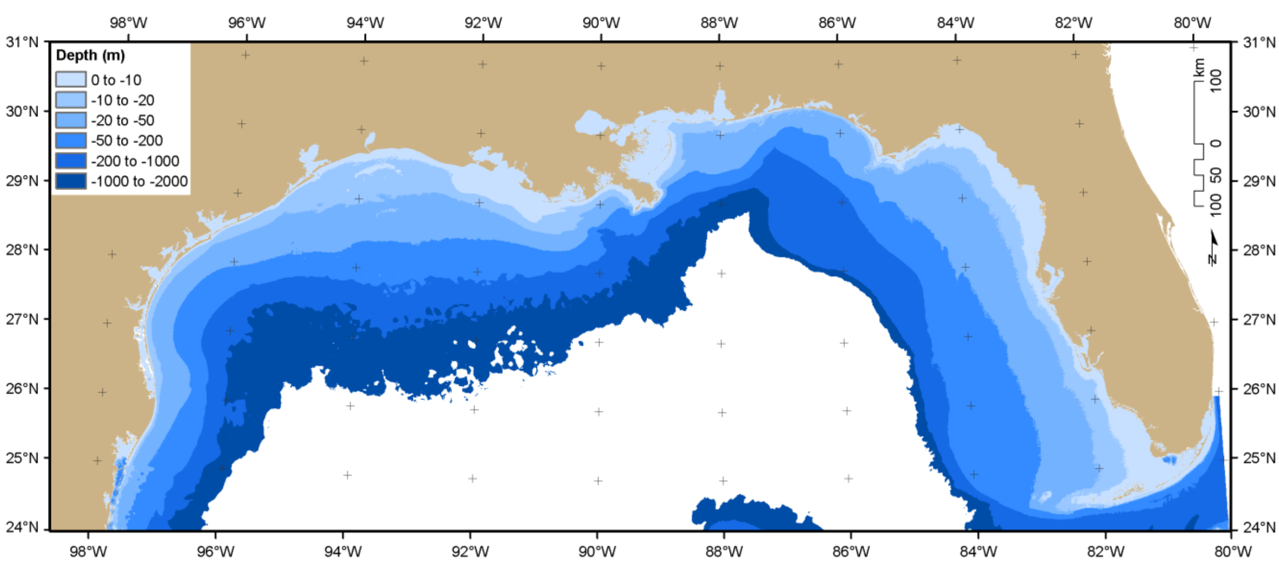

Figure 1. Map of the area modeled within *Ecopath with Ecosim*.

2.1.1 Addition of Meiofauna and Macrofauna

In the model by Suprenand et al. (2015) the organisms living on/in the sediment included blue crabs, stone crabs, benthic invertebrates, and infauna. For our study the original infauna functional group (20 t/km^2) was further divided into the meiofauna and macrofauna functional groups. Figure 3 from Thiel (1979) shows the relationship between macrofauna and meiofauna with depth, which was then used to determine the proportion of infauna belonging to the macrofauna and meiofauna. Because the majority of the model area is comprised of depths ≤ 1000, we use the starting points of the graph to determine a value of 3.4 (number/m^2 log$_{10}$) for macrofauna and 6.05 (number/m^2 log$_{10}$) for meiofauna, based on proportions. Therefore, 56% of the infauna belong to the Meiofauna functional group, resulting in a biomass of 11.2 t/km^2, and 44% of the infauna belong to the Macrofauna functional group, resulting in a biomass of 8.8 t/km^2. However, to during the process of balancing the model these values were changed to 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 were taken from Arreguin-Sanchez et al. (2002). Macrofauna values are based off polycheate rates; specifically; a P/B of 4 and a Q/B of 21. The meiofauna values are a P/B of 8 and a Q/B of 53.

Following the division of the infauna functional group, the diet matrix was updated to include trophodynamic connections unique to macrofauna and meiofauna. Functional groups that previously feed on zoobenthos and/or detritus are assumed to ingest meiofauna and macrofauna, at least inadvertently. In some cases the consumption of detritus, macrofauna, and meiofauna is added to reflect diet information available on Fishbase. For example, pigfish feed on zoobenthos and detritus (Fishbase, 2016), but this was not captured in the original diet matrix.

2.1.2 Ecopath Balancing

The model was balanced to correct high biomass accumulation (BA) values, low ecotrophic efficiency (EE) values, and production/consumption (P/Q) values outside of the 0.1-0.3 range. In the original model by Suprenand et al. (2015) biomass accumulation was estimated for Red Drum (0-3), Red Drum (3-8), Red Drum (8-18), Red Drum (18-36), Red Drum (36+), Seatrout (18+), Mullet (0-6), Shrimp, Blue Crab, Red Snapper (older), and Atlantic Croaker. Functional groups in which the BA values were to high included the following: Red Drum (0-3) - 1.78, Red Drum (3-8) - 2.61, Sea Trout (0-3) - 1.20, Mullet (0-6) - 2.80, and Mackerel (0-3) - 1.01. To lower the BA of the Red Drum functional group (0-3) pinfish were included as a predator because they are natural predators of red drum larvae (Fuiman, 1994; Rooker et al, 1998). As this initially put too much strain on the Red Drum multistanza group, the biomass of the Red Drum (36+) functional group is increased by 0.3 t km^{-2}, which results in a biomass increase of all red drum functional groups. These changes result in a BA/year of 0.7 for Red Drum (0-3). The Red Drum (3-8) functional group BA was lowered to a BA/year of 0.97 by adding predation from large coastal sharks and increasing the predation from Jacks, Grouper 3+, and Grouper 1-3. The BA/year for Sea Trout was reduced by adding predation from Sea Trout 18+. In the original model 18+ Sea Trout were already feeding on the 3-18 Sea Trout, and therefore it is likely that a small portion of older Sea Trout fed on younger Sea Trout, which results in a BA of 0. The Mullet (0-6) BA/year was corrected to 0.45 by increasing predation from Red Drum (ages 8-18, 18-36, and36+), Sea Trout (18+), Mackerel (0-3) and Ladyfish. Lastly, Mackerel (0-3) was corrected by increasing predation of Sea Trout (3-18/18+), resulting in a BA/year of 0 even with a very small increase in predation.

In general, Ecotrophic efficiency (EE) values were too low for Ladyfish (0-10) - 0.03, Ladyfish (10+) - 0.09, and Red snapper (6-24) - 28. To increase EE values for the entire Ladyfish multistanza groups, we further appended trophodynamic connections in the diet matrix. For example, it is suggested that juvenile ladyfish may be preyed on by zooplankton (Florida Museum of Natural History, 2016) and larger fish; therefore, macrozooplankton are included as predators of ladyfish (0-3), and large coastal sharks are included as predators of ladyfish (10+). This results in new EE values for ladyfish: (0-10) = 0.72, and ladyfish (10+) = 0.41. The EE of red snapper was increased by adding large coastal sharks as predators, resulting in a new EE of 0.55. This increase is logical, as the large coastal sharks were predators of the other age classes in the original model by Suprenand et al. (2015).

Finally, to accomplish a balanced Ecopath model P/Q ratios for Red drum (0-3), Red drum (36+), Mullet (0-6), Small Fish, and Large Coastal Sharks are appended to fall within the acceptable range of 0.1-0.3. As consumption can only be changed for the leading stanza, which subsequently changes the consumption of the other age groups within the stanza, we focused on changing the lead stanza in the Red Drum and Mullet functional groups ,by adjusting the production values (z). To get a value within the acceptable range, the z value for Red Drum (0-3) and (36+) was increased from 2 to 3.5 and 0.15 to 0.25, respectively. The acceptable range for Red drum (0-3) was based on Chagaris (2007), where z was increased to 6 in order to balance the model. Similarly, Mullet (0-6) was increased from 3 to 5.1, as in Chagaris (2007), the balanced model resulted in z values of mullet (0-6)- 6, mullet (6-18)-3, and mullet (18+)- 1.0. Correction to the small fish ratio was made by implementing the P/B and Q/B values from Gascuel et al. (2008), as this functional group includes invertivore species such as silversides and threadfin shad. These values are 1.070 for P/Band 9.60 for Q/B. The final ratio for large coastal sharks was appended by using the production value of 0.405from Gascuel et al. (2008). The consumption value was not used because it led to the model being largely unbalanced.

2.1.3 Oil Forcing Functions

Oil forcing functions were generated by translating the effect of oil concentrations on search rate, a proxy for mortality, through the use of dose response models following the methods of Okey and Ainsworth (unpublished). For all functional groups, the percent mortality change and the percent of the population affected were calculated. The two where then multiplied to determine the overall impact in the model area. These values were entered as a negative modifier on consumer search rate within Ecosim, Table 1. This allows the model to be predictive and have a monthly oil exposure as opposed to adding mortality directly as a mortality forcing.

Table 1. Values used to calculate the mortality modifier entered under search rate. Overall change = change in mortality*fraction of population impacted. 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.

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	0.9822
Mackrel (3+)	0.39	0.15	0.059	0.9408
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	0.8562
Jacks	0.58	0.14	0.080	0.9204
Bay Anchovy	0.18	0.14	0.025	0.9754
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	0.9317
Lobster	0.20	0.14	0.030	0.9712
Red Snapper 0-6	0.15	0.13	0.020	0.9799

Group name	Change in Mortality	Fraction of Population Impacted	Overall Change	Search Rate Modifier
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	0.7547
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

2.1.3.1 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 spill on the frequency of fish lesions, a proxy for mortality rate. The results of their work indicated a ‘hockey stick’ model was best for mortality. The ‘hockey stick’ model implies that below a certain oil concentration there are no lethal effects on the population. Because the expansion of the Northern Gulf of Mexico model is based on the impacts of the Deepwater Horizon blow out the parameters estimated by Dornberger et al. (2016) were used to determine the impact on the fish functional groups in the model. In equation 1 below, the parameters from Dorenberger et al. (2016) that were used are oil threshold = 2.942 and *m* = 0.1051. The oil threshold is the oil concentration level above which population-level effects increase log-linearly (Horness et al. 1998; Johnson et al. 2002). In equation 1, *m* is the rate of change in the population response. The Z parameter is the natural mortality which is equivalent to P/B for each functional group in the Ecopath model. The oil concentration parameter for each functional group was determined by examining the predicted water column oil concentrations by depth, the depth ranges of fish groups, and the spatial extent of the model area that was covered by the surface oil slick.

$$Z^* = \begin{cases} Z & if\ [Oil] < [Oil]_{thresh} \\ Z + m * \log[Oil/Oil_{thresh}] & otherwise \end{cases} \qquad \text{Eq. 1}$$

From Okey and Ainsworth (unpublished)

Oil concentrations based on total PAH were acquired from the predicted water column oil concentrations of Paris et al. (2012). Their model is based off far-field oil plume modeling and provides oil concentration by depth and days after the spill, Figure 2. In order to determine what oil concentrations the fish groups are exposed, we estimated their ecologically relative depth ranges (Table 2). For example, the depth ranges of juvenile fish are generally restricted to a shallower depth ranges (for example see, Frias-Torres et al., 2007). In aggregated (multi-species) fish functional groups, the full depth range over which all species live is used. The fish depth ranges were then compared to the oil concentration data to determine the total oil each group was exposed. When the fish range did not encompass the entire depth range in the oil model the percent of the depth the fish inhabits is calculated, then entered into in equation 1 as the oil parameter. For example, the max depth of bay anchovies is 70 m but the oil concentration is from 50-200m. Therefore, 70 divided by 150 (200-50) equals 0.46, or 46% percent of the total oil concentration at that depth was applied to the oil exposure for the group.

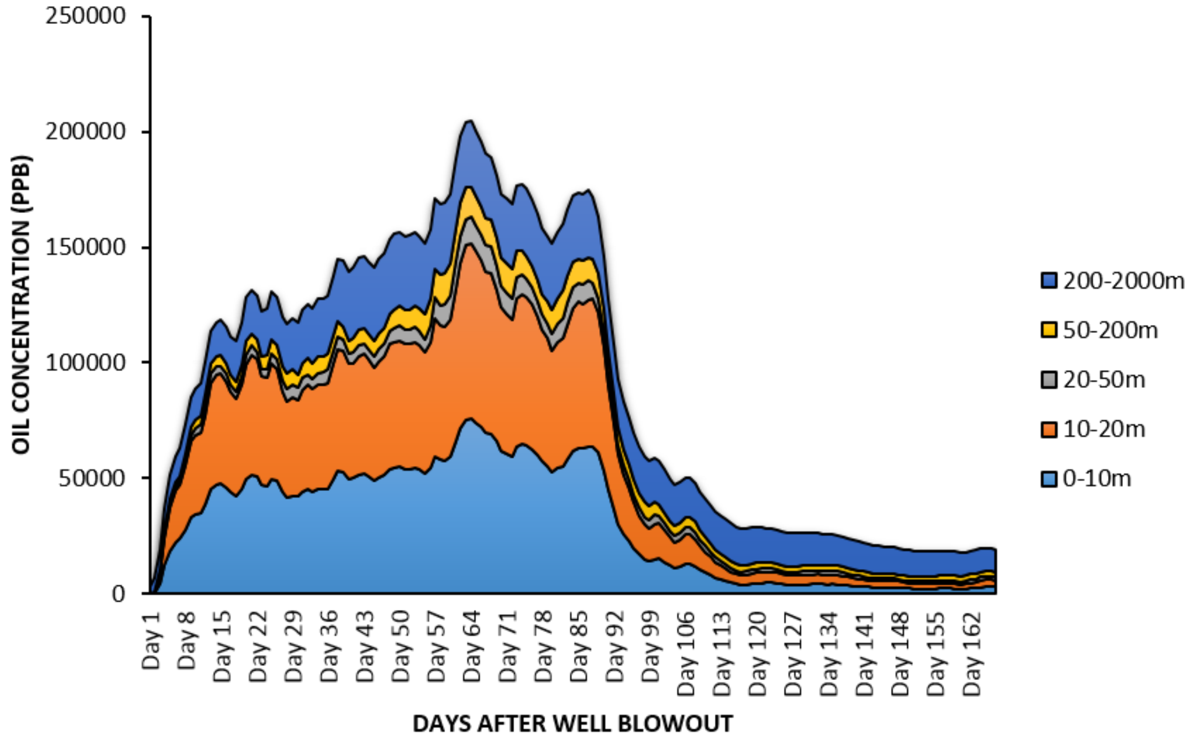

Figure 2. Modeled oil concentration in the water column based on the data of Paris et al. (2012).

Table 2. Depth ranges of fish functional groups.

Group name	Species Common Name	Min Depth (m)	Max Depth (m)	Source
Red Drum	*Sciaenops ocellata*	2	200	Powers et al. (2012)
Sea Trout	*Cynoscion arenarius*	1	30	Mcdonald et al. (2009)
	Cynoscion nebulosus	10		Fishbase (2016)
	Cynoscion nothus	2	18	Fishbase (2016)
Mullet	*Mugil cephalus*	0	120	Fishbase (2016)
	Mugil curema	1	30	Fishbase (2016)
Mackerel	*Auxis rochei*	10		Fishbase (2016)
	Scomber japonicas	0	300	Fishbase (2016)
	Scomber scombrus	0	1000	Fishbase (2016)
	Scomberomorus cavalla	5	140	Fishbase (2016)
	Scomberomorus maculatus	10	35	Fishbase (2016)
	Scomberomorus regalis	1	20	Fishbase (2016)
Ladyfish	*Elops saurus*	0	50	Gulf Coast Research Laboratory (2016)
Grouper (0, 1-3, 3+)	*Epinephelus morio*	5	330	Fishbase (2016)
	Epinephelus adscensionis	1	120	Fishbase (2016)
	Epinephelus drummondhayi	60	120	Fishbase (2016)
	Epinephelus flavolimbatus	90	360	Fishbase (2016)
	Epinephelus guttatus	100		Fishbase (2016)
	Epinephelus itajara	0	100	Fishbase (2016)
	Epinephelus nigritus	55	525	Fishbase (2016)
	Epinephelus niveatus	100	200	Fishbase (2016)
	Mycteroperca bonaci		250	Fishbase (2016)
	Mycteroperca interstitialis	2	35	Fishbase (2016)
	Mycteroperca microlepis	30	160	Fishbase (2016)
	Mycteroperca phenax	30	100	Fishbase (2016)
	Mycteroperca venenosa	2	137	Fishbase (2016)
Jacks	*Caranx hippos*	1	350	Fishbase (2016)
	Caranx crysos	0	100	Fishbase (2016)
	Hemicranx amblyrhynchus	0	50	Fishbase (2016)
	Seriola dumerili	1	360	Fishbase (2016)
	Seriola fasciata	55	130	Fishbase (2016)
	Seriola rivoliana	5	245	Fishbase (2016)
Bay Anchovy	*Anchoa mitchilli*	1	70	Fishbase (2016)
Pin Fish	*Lagodon rhomboids*	1	17	Nelson (2002)
Small fish	*Dorosoma petenense*	0	15	Fishbase (2016)
Silver Perch	*Bairdiella chrysoura*	0	20	No information found depth range estimated
Scaled Sardine	*Harengula jaguana*	1	22	Fishbase (2016)
Menhaden	*Brevoortia patronus*	0	50	Fishbase (2016)
	Brevoortia gunteri	0	50	Fishbase (2016)
	Brevoortia smithi	0	50	Fishbase (2016)

Group name	Species Common Name	Min Depth (m)	Max Depth (m)	Source
Catfish	*Bagre marinus*	0	50	Fishbase (2016)
	Ictalurus furcatus	50		Fishbase (2016)
	Ictalurus punctatus	15		Fishbase (2016)
Caridean Shrimp		0	850	King and Butler (1985)
	Farfantepenaeus notialis	48	329	Encyclopedia of Life (2016)
Shrimp	*Farfantepenaeus subtilis*	4.5	174	Encyclopedia of Life (2016)
	Liptopenaeus schmitti	4	45	Encyclopedia of Life (2016)
Stone Crab	*Menippe mercenaria*	4	790	Munro et al. (2015)
Blue Crab	*Callinectes spp.*	0	36	Animal Diversity (2016)
Pigfish	*Orthopristis chrysopterus*	0	20	No information found depth range estimated
	Rhinoptera bonasus	0	25	Fishbase (2016)
Rays	*Dasyatus sabina*	0	22	Fishbase (2016)
	Dasyatis americana	0	53	Fishbase (2016)
	Trachinotus carolinus	0	70	Fishbase (2016)
Pompano	*Rachycentron canadum*	0	1200	Fishbase (2016)
	Alectic ciliaris	60	100	Fishbase (2016)
	Trachinotus falcatus	0	36	Fishbase (2016)
	Homarus	2	470	Encyclopedia of Life (2016)
	Munida flinti	100	183	Encyclopedia of Life (2016)
	Munida forceps	105	580	Encyclopedia of Life (2016)
	Munida iris	28	604	Encyclopedia of Life (2016)
	Munida irrasa	17	425	Encyclopedia of Life (2016)
	Munida pusilla	18	159	Encyclopedia of Life (2016)
Lobster	*Munida simplex*	52	1170	Encyclopedia of Life (2016)
	Munida Valida	9	850	Encyclopedia of Life (2016)
	Nephtropsis aculeata	16	662	Encyclopedia of Life (2016)
	Nephtropsis rosea	27	1097	Encyclopedia of Life (2016)
	Polycheles typhlops	64	2971	Encyclopedia of Life (2016)
	Panulirus argus	1	393	Encyclopedia of Life (2016)
	Scyllarus americanus	4	145.5	Encyclopedia of Life (2016)
	Scyllarus chacei		342	Encyclopedia of Life (2016)
Red snapper	*Lutjanus campechanus*	10	190	Fishbase (2016)
Atlantic croaker	*Micropogonias undulatus*	0	12	Comyns and Lyczkowski-Shultz (2004)
	Carcharhinus leucas	1	152	Fishbase (2016)
	Carcharhinus limbatus	0	100	Fishbase (2016)
	Carcharhinus isodon		10	Fishbase (2016)
LC sharks	*Isurus oxyruchus*	0	740	Fishbase (2016)
	Carcharhinus plumbeus	0	500	Fishbase (2016)
	Sphyrna mokarran	1	300	Fishbase (2016)
	Carcharhinus brevipinna	0	100	Fishbase (2016)
	Galeocerdo cuvier	350	800	Fishbase (2016)

To determine what percentage of the entire model population was impacted, the spatial extent of the fish depth ranges and the area impacted by the oil spill were calculated within ArcGIS 10.0. To accomplish this a map was created based on the spatial parameters of Suprenand et al. (2015) projected using Albers Equal Area GOM. Bathymetry contours from Texas A&M University's Gulf of Mexico Coastal and Ocean Observing System (GCOOS, 2016) were added to the map and used to calculate the area within the modeled depth ranges of Paris et al. 2012. The oil spill surface layer from ERMA Deepwater Gulf Response was added to calculate the area of each depth range exposed to oil (ERMA, 2016), Table 3.

Table 3. Calculated model area for each depth range and the area of the surface slick. Calculations made within ArcGIS. Shoreline is linear not area.

Depth Range (m)	Total Area (Km^2)	Area Oiled (Km^2)
Shoreline	25584.407 (Km)	From Nixon et al. (2016)
0-10	73570.5399	10103.80337
10-20	57791.068	7338.94256
20-50	126028.823	18573.87889
50-200	126506.487	14890.64532
200-1000	226204.862	40485.95439
100-2000	273828.019	46728.2064
200-2000	114830.548	24980.78794

2.1.3.2 Invertebrate Forcing Functions

Dose response models were used to determine the impact of the Deepwater Horizon Oil spill on invertebrates. According to figure 1 in, Echols et al. (2016), 20% mortality of the mysid shrimp (*Americamysis bahai*) occurs at TPAH concentrations of 60 ug/l (60 ppb). This relationship was used to estimate the impact on mortality for both shrimp functional groups in the model. Caridean shrimp and other shrimp species are found between 0-850 m and 4-329 m respectively (Fishbase, 2016), Table 2. These depth ranges were exposed to oil concentrations of 80248 ppb and 74516 ppb, according to the oil model of Paris et al. 2012. Because both concentrations are above the 20% mortality mark, a 20% increase in mortality was assumed for the entire affected population. Affected population was determined using the same method for fish forcing functions mentioned above.

The response of stone crabs and blue crabs was calculated based on the LC20 values presented in DWHNRDA (2016). The report showed 20% mortality occurring at TPAH50 concentrations of 56.8-105.1 ug/l (56.8-105.1 ppb). Based on the depth ranges for stone crabs and blue crabs (Table 2) they were exposed to 79655 ppb and 60992 ppb respectively. Therefore, 20% mortality was assumed to occur in the percentage of the population exposed to the oil. The same dose response was used for lobsters and benthic invertebrates because to our knowledge no toxicity study relating to the DWH has been published for these functional groups.

To determine the impact on meiofauna and macrofauna, the in progress dose response model of Balthis et al. (unpublished) was used. A high risk of impact (>80%) was found to occur at concentrations greater than 24 ppm (24000 ppb) for macrofauna and 25 ppm (25000 ppb) for meiofauna. The data collected by Montagna et al. (2013) showed the average of the sum of 40 toxic Polycyclic Aromatic Hydrocarbons (PAH40) to be 215,919 ppb across their study area in 2010, a value greater than the concentration in the dose response model, resulting in an 80% chance of impact. Therefore, 80% increase in mortality was assumed in the habitat area where oiling occurred. Data collected in 2011 shows very little decrease in sediment PAH40 so the forcing function was carried over into 2011. According to DWHNRDA (2016) chemistry and floc data up to 1810 km^2 of the deep seafloor were impacted, with evidence suggesting a larger area. Meiofauna and macrofauna are found at all depths therefore the 2113 km of oiled shoreline (Nixon et al. 2016) was also added to the amount of habitat area impacted. For the purpose of the model, we used 1 km for the width of the shoreline.

Almeda et al. (2013) found that mesozooplankton had a LC50 of 32.4 µl/L (32400 ppb) for crude oil PAHs. This relationship was applied to both the macro and micro zooplankton groups in the model. The DWHNRDA (2016) calculated the surface PAH50 oil concentration to be between 1010-13700 µg/g (1010000-13700000 ppb) and the area of the surface oil to be 112115 km^2. Therefore, a 50% increase in mortality was applied to the fraction of the population exposed to the oil slick. Evidence suggests that the impact was short lived and changes were no longer significant by July 2010 (Carassou et al. 2014). Therefore the forcing function was only entered from April-June 2010.

2.1.3.2 Primary Producers

The results of studying the impact of oil on phytoplankton have been mixed. Studies have shown an increase in phytoplankton growth, an inhibition of photosynthesis, and the occurrence of blooms following a spill (for examples see, Hu et al. 2011). With regards to the DWH blowout satellite imagery shows a short lived phytoplankton bloom occurring in August 2010 (> 1.0 mg/m3 of Chlorophyll-a) and evidence suggests it may have been linked to the spill (Hu et al. 2011). In contrast, Prouty et al. (2016) found that there was a reduction in primary production and carbon export to the deep sea at least 6-18 months following the blow out. Therefore, for the purpose of our project we combined the findings of Hu et al. (2001) and the toxicity findings of Garr et al. (2014) when generating the forcing function. When the surface oil slick was present the dose response model of Garr et al. (2014) was used to generate an increased mortality. In their study they found that after 96 hours of exposure the growth of two species of microalgae was inhibited by 50% at PAH of 0.106/0.143 mg/L. For our model we assumed growth inhibition to be equal to mortality. The average PAH concentration of 0.1245 mg/l (124.5 ppb) for the two species was used as the threshold for 50% mortality. The 50% mortality was applied to the fraction of the phytoplankton population exposed to the surface oil slick until August when the slick disappeared. From August-September an increase in biomass was simulated by decreasing the amount of time it took predators to search for phytoplankton.

Silliman et al. (2012) found that sea grasses located within 10m of oiled shorelines experienced nearly complete loss (approx. 95%) of above ground cover. According to Nixon et al. (2016) 2113 km of shoreline were oiled. The area of impacted habitat was calculated to be

21.13 km^2. Therefore, 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. Shoreline area within the model was calculated in ARCGIS as mentioned above. The time of impact was set from May 2010 until August 2010 when the surface oil slick disappeared. The impact on attached microalgae was calculated by using the same exposure response as for phytoplankton but with the area and time frame of seagrass.

2.2 Observational Data

The temporal predictability of the model was improved by comparing the model outputs to observational data collected for catch and relative biomass values throughout the entire northern GoM. The fish data was obtained from SEAMAP's public database which provides catch information for the entire Northern Gulf of Mexico. This abundance information was standardized across surveys by calculating catch per unit effort (CPUE). To calculate biomass from CPUE the average biomass for a species was used based on information from Fishbase (2016). For our purposes, the average yearly biomass from 2004-2014 was then used to generate a time series that was entered as a csv file and loaded into Ecosim for tuning of the model. Only data sets that encompassed the entire range of the Gulf of Mexico were included. However, data localized to the blowout location was also obtained. Commercial and recreational landings data was obtained from NOAA's public landings statistics (NMFS, 2016) from 2004-2014.

2.2.1 Ecosim Tuning

Vulnerabilities effect whether the simulation treats the functional group as a top-down (high value) or bottom up (low value) control. Instead of using the default vulnerabilities of 2 they were determined by running a vulnerability search on the compiled time series. A number of combinations of predator and prey categories were tested. We achieved the lowest sum of squares (SS) by running 45 categories across predators (SS=329).

Model predictability was improved by comparing observational data to the model output. In general, the model overestimated catch values. This was corrected by entering the 2004 landings data as the initial fishery's landings entry within *Ecopath*. When only an incomplete time series was available then the average across all years excluding 2010 was used for the initial fishery's entry. In some cases catch data was still too high because of high discards and deaths due to red tides, part of Suprenand et al.'s original (2015) model. Because the focus of the model is on the DWH blow out the red tide fishery was removed. Discards were corrected to reflect no more than 6 times the target catch for the dirtiest fishery like shrimp trawls and no more than 2 times the target catch for all other fisheries. These changes resulted in a good fit to observational data (Figures 3 and 4).

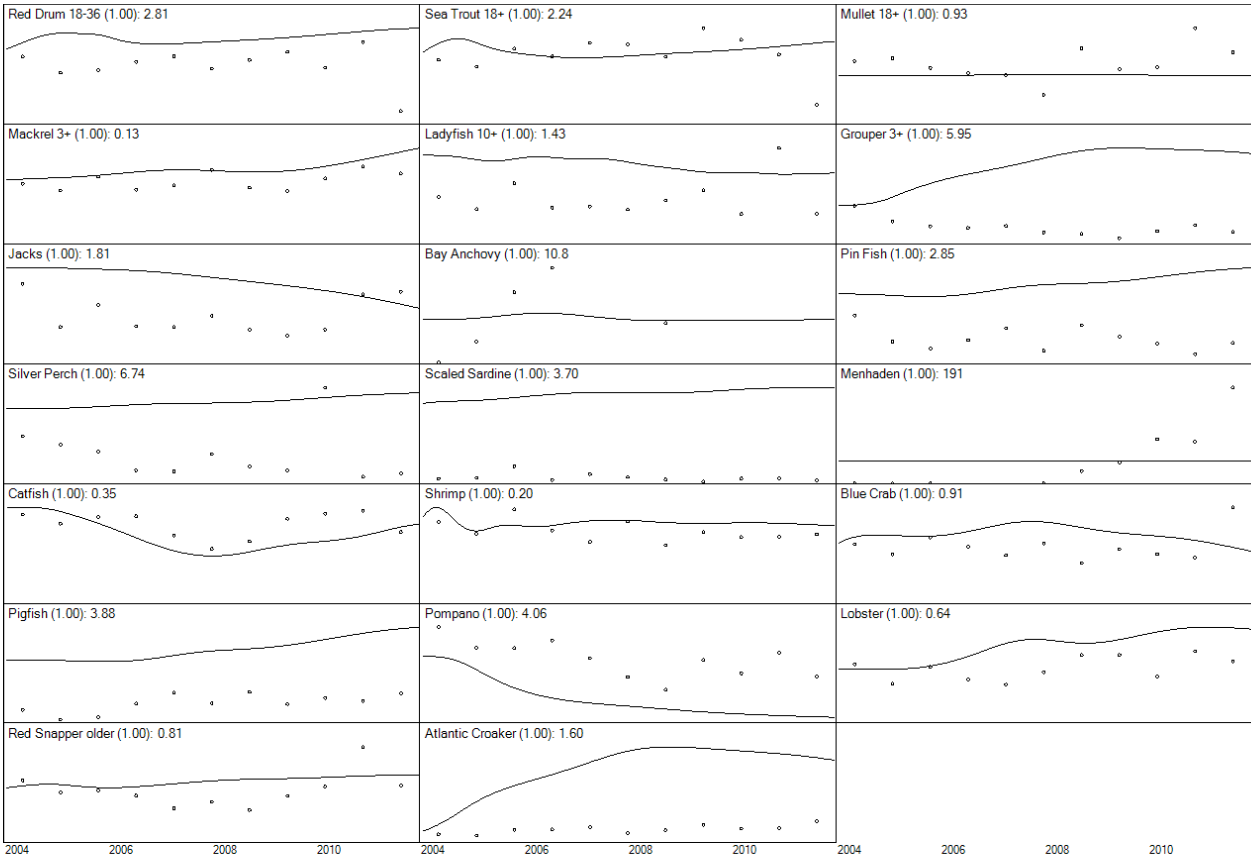

Figure 3. Predicted catch (solid line) compared to time series data (dots) from 2004-2014.

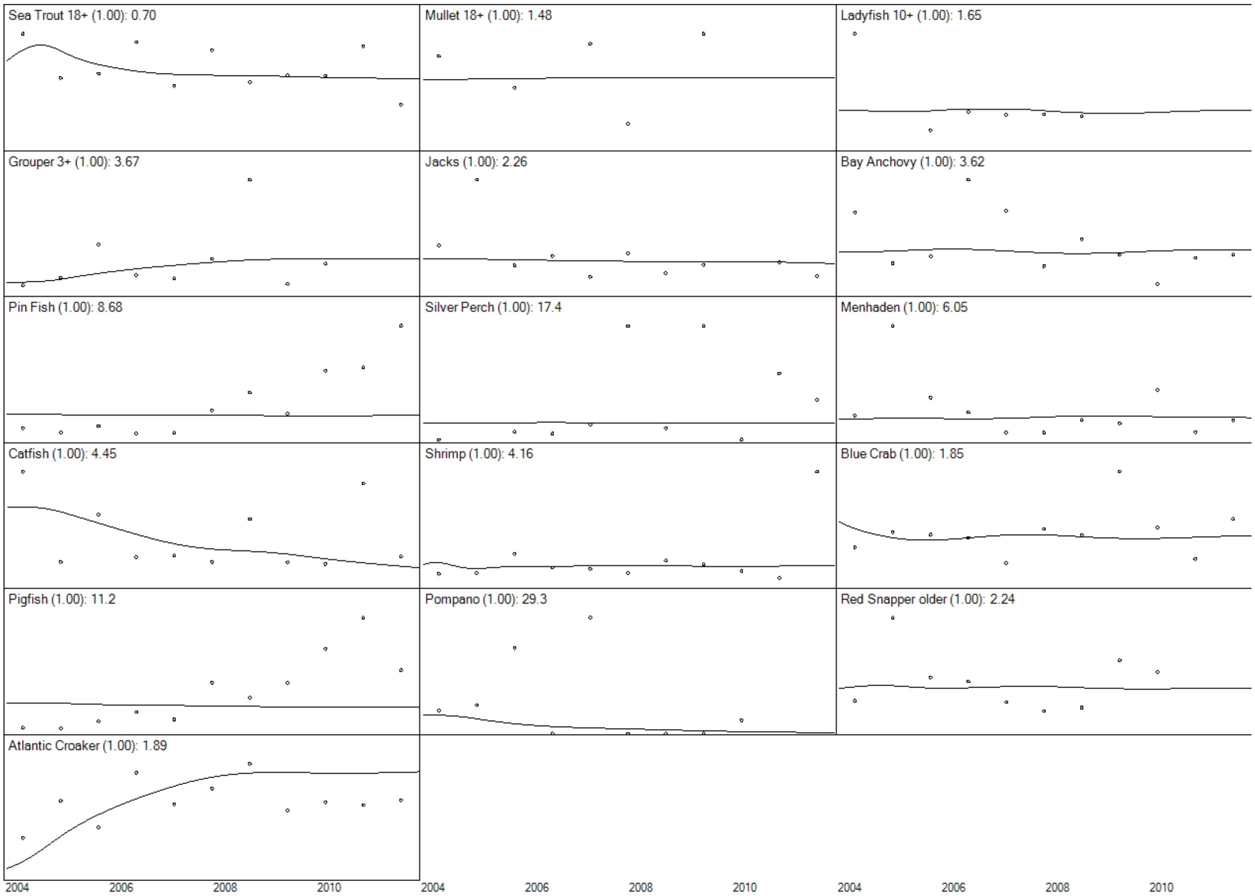

Figure 4. Predicted relative biomass (solid line) compared to time series data (dots) from 2004-2014.

2.3 Ecosystem Services

The model by Suprenand et al. 2015 contains 48 functional groups. In order to test the impact on human well-being that changes in these groups would have, ecosystem service categories were assigned to each group (Table 4). Categories were 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. The ecosystem services for commercial fisheries and carbon sequestration were valued by assigning monetary values to the modeled outputs. 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 is found, resulting in a value of metric tons. This value was then multiplied by the average percent of catch that is attributed to recreational and commercial fisheries, calculated from values provided by the National Marine Fisheries Service (NMFS) from 2004-2014 excluding 2010. In order to determine the monetary value of the commercial fisheries yield the inflation adjusted ex-vessel prices from NMFS (2012) were applied to the model outputs for 2010, 2011, and 2012.

Carbon sequestration was evaluated by first determining how much of the atmospheric carbon is sequestered in the deep sea. 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 2000m 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. As in Melvin et al. (2016) we used 200m as the depth for the top of the permanent pycnocline and 1000m for the bottom of the permanent. The Gulf of Mexico is not counted among the 56 biogeochemical provinces therefore we compared two estimates of carbon sequestration, the values for the Gulf Stream (1.81 Tg C yr^{-1}) and the global value (0.72 Pg C yr^{-1}). This represents 0.00024% and 0.095% of the total atmospheric carbon, based on a total of 760 Gt (Mcleod et al. 2011). Once the change in carbon sequestration was calculated by comparing normal model outputs to spill outputs the social cost of CO_2 was used to place a dollar value on the changes in 2010 and 2011. The IWGSCC (2015) 3% average value of $36 per metric ton of CO_2 was applied to the model output.

Table 4. Functional groups in the model and assigned ecosystem services.

Group name	Ecosystem Service Category	Specific Service	Evaluation Method
0-3 Red Drum	Cultural	Recreation and ecotourism	Monetary Valuation
3-8 Red Drum	Provisioning	Food	Monetary Valuation
8-18 Red Drum	Provisioning	Food	Monetary Valuation
18-36 Red Drum	Provisioning	Food	Monetary Valuation
36+ Red Drum	Provisioning	Food	Monetary Valuation
0-3 Sea Trout	Provisioning	Food	Monetary Valuation
3-18 Sea Trout	Provisioning	Food	Monetary Valuation
18+ Sea Trout	Provisioning	Food	Monetary Valuation
0-6 Mullet	Provisioning	Recreation and ecotourism	Monetary Valuation
6-18 Mullet	Provisioning	Food	Monetary Valuation
18+ Mullet	Provisioning	Food	Monetary Valuation
Mackerel 0-3	Provisioning	Food	Monetary Valuation
Mackerel 3+	Provisioning	Food	Monetary Valuation
Ladyfish 0-10	Cultural	Recreation and ecotourism	Monetary Valuation
Ladyfish 10+	Cultural	Recreation and ecotourism	Monetary Valuation
Grouper 0	Cultural	Recreation and ecotourism	Monetary Valuation
Grouper 1-3	Provisioning/ Cultural	Food /Scuba Diving	Monetary Valuation
Grouper 3+	Provisioning/ Cultural	Food /Scuba Diving	Monetary Valuation
Jacks	Provisioning	Food	Monetary Valuation
Bay Anchovy	Cultural	Recreation and ecotourism	Monetary Valuation
Pin Fish	Cultural	Recreation and ecotourism	Monetary Valuation
Small fish	Cultural	Recreation and ecotourism	Monetary Valuation
Silver Perch	Cultural	Recreation and ecotourism	Monetary Valuation
Scaled Sardine	Cultural	Recreation and ecotourism	Monetary Valuation
juv Menhaden	Cultural	Recreation and ecotourism	Monetary Valuation
Menhaden	Cultural	Recreation and ecotourism	Monetary Valuation
Catfish	Provisioning	Food	Monetary Valuation
Caridan Shrimp	Provisioning	Food	Monetary Valuation
Shrimp	Provisioning	Food	Monetary Valuation
Stone Crab	Cultural		
Blue Crab	Provisioning	Food	Monetary Valuation
Pigfish	Provisioning	Food	Monetary Valuation
Rays	Cultural	Scuba Diving/Snorkeling	
Pompano	Provisioning	Food	Monetary Valuation
Lobster	Provisioning	Food	Monetary Valuation
red snapper 0-6	Cultural	Recreational Fisheries	Monetary Valuation
red snapper 6-24	Provisioning	Food	Monetary Valuation
red snapper older	Provisioning	Food	Monetary Valuation
Atlantic croaker	Provisioning	Food	Monetary Valuation
Large Coastal Sharks	Cultural	Recreation and ecotourism	Monetary Valuation

Group name	Ecosystem Service Category	Specific Service	Evaluation Method
Benthic Invertebrates	Supporting/Cultural	Recreation and ecotourism /Food web	
Macro Zooplankton	Supporting	Nutrient Cycling	
Micro Zoolplankton	Supporting	Food web	
Attached Microalgae	Supporting	Food web	
Sea Grass	Supporting/Cultural	Habitat	
Phytoplankton	Supporting	Primary Production	
Meiofauna	Supporting	Food web/Nutrient Cycling	
Macrofauna	Regulating	Purification and Waste Treatment	
Detritus	Regulating/Supporting	Climate Regulation/ Nutrient Cycling	Carbon Market

3. Results

3.1 Model Simulations

Two temporal simulations starting with initial conditions in 2004 and predicting forward to 2014 were run, (1) normal conditions and (2) DWH blowout. Ecosim gives results for absolute biomass in metric tons/ km^2. These results were multiplied by the habitat area for each functional group to measure changes in metric tons. In general, when the absolute biomass output from normal conditions was compared to the DWH blowout, biomass decreased in the DWH blowout simulation (Table 5). The percent changes in biomass were larger in 2011 than in 2010. The highest percent change in 2010 was seen in Macro Zooplankton with a decrease of 17.61% compared to a 67892% increase in Mullet 18+ in 2011.

Table 5. 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.

Group name	2010 Biomass Change (Metric Tons)	2011 Biomass Change (Metric Tons)
0-3 Red Drum	-0.43 (1.37%)	14.63 (81.65%)
3-8 Red Drum	-359.79 (9.50%)	-4783.30 (57.06%)
8-18 Red Drum	-776.27 (2.78%)	-55824.44 (67.92%)
18-36 Red Drum	-835.09 (0.78%)	85545.19 (400.80%)
36+ Red Drum	5622.55 (0.62%)	672122.31 (265.32%)
0-3 Sea Trout	-3.07 (7.91%)	-791060.82 (100%)
3-18 Sea Trout	-229.07 (3.86%)	4925.18 (16297.22%)
18+ Sea Trout	-409.41 (0.74%)	-938.14 (1.72%)
0-6 Mullet	-577.08 (8.09%)	-2282.20 (22.94%)
6-18 Mullet	-14580.60 (5.88%)	-18894.22 (7.34%)
18+ Mullet	-14766.21 (1.25%)	1157040.62 (67891.89%)
Mackerel 0-3	-4.93 (8.37%)	-12437.94 (99.53%)
Mackerel 3+	-3356.93 (3%)	-590449.53 (85.24%)
Ladyfish 0-10	-1203.44 (6.24%)	15206.54 (240.65%)
Ladyfish 10+	-10424.42 (5.72%)	-74119.90 (31.22%)
Grouper 0	-22.72 (3.76%)	-500735.60 (99.89%)
Grouper 1-3	-69.37 (2.45%)	-341104.09 (99.04%)
Grouper 3+	56.19 (0.01%)	471199.00 (314.08%)
Jacks	-72.67 (0.69%)	-4944482.95 (99.79%)
Bay Anchovy	-17008.17 (5.63%)	-59246.07 (17.19%)
Pin Fish	-6618.79 (1.31%)	-51351.09 (9.49%)
Small fish	-12187.73 (3.45%)	315869.25 (4675.09%)
Silver Perch	-1201.76 (2.71%)	-646053.86 (93.68%)
Scaled Sardine	-57043.78 (3.75%)	1395850.02 (1035.74%)
juv Menhaden	-1536.52 (1.98%)	77715.83 (905.65%)
Menhaden	-17558.15 (1.74%)	1067553.45 (2647.95%)
Catfish	-458.46 (3.23%)	-53690.34 (81.25%)
Caridan Shrimp	-54327.39 (2.22%)	566067.81 (28.53%)
Shrimp	-13188.08 (3%)	397707.72 (1181.58%)
Stone Crab	-1507.04 (2.36%)	-554592.74 (91.56%)
Blue Crab	-444.00 (2.11%)	18563.95 (1538.01%)
Pigfish	-280.25 (0.83%)	4642.98 (16.62%)
Rays	3224.59 (0.3%)	901364.84 (621.53%)
Pompano	-869.70 (1.77%)	-3217374.73 (98.79%)
Lobster	-6567.99 (1%)	-6402665.90 (91.17%)
red snapper 0-6	-109.44 (8.83%)	-3086529.92 (99.96%)
red snapper 6-24	-3519.38 (4.16%)	-2704452.75 (97.16%)
red snapper older	-2411.03 (1.10%)	-3479127.13 (94.29%)

Group name	2010 Biomass Change (Metric Tons)	2011 Biomass Change (Metric Tons)
Atlantic croaker	-3873.59 (0.5%)	-3117778.44 (79.91%)
Large Coastal Sharks	-25515.28 (0.39%)	-78172528.83 (92.22%)
Benthic Invertebrates (Entire)	-708932.17 (4.74%)	-907038.32 (6.32%)
Benthic Invertebrates (Offshore)	-318185.42 (4.74%)	-407100.12 (6.32%)
Macro Zooplankton (Entire)	-834912.69 (17.61%)	-76687.96 (1.73%)
Micro Zooplankton (Entire)	453834.58 (8.67%)	637988.31 (10.88%)
Macro Zooplankton (Offshore)	-374728.44 (17.61%)	-34419.36 (1.73%)
Micro Zooplankton (Offshore)	203691.63 (8.67%)	286344.15 (10.88%)
Attached Microalgae	-40.78 (0.54%)	65.29 (0.86%)
Sea Grass	-2656.25 (5.93%)	-610.52 (1.36%)
Phytoplankton (Entire)	-154148.32 (1.04%)	-424331.89 (2.94%)
Phytoplankton (Offshore)	-69185.39 (1.04%)	-190450.13 (2.94%)
Meiofauna (Entire)	-78459.09 (1.10%)	-42579.00 (0.59%)
Macrofauna (Entire)	-148992.96 (1.81%)	-139835.33 (1.91%)
Meiofauna (Offshore)	-35214.28 (1.10%)	-19110.46 (0.59%)
Macrofauna (Offshore)	-66871.54 (1.81%)	-62761.38 (1.91%)
Detritus (Entire)	-1078855.18 (1.79%)	-693222.05 (1.15%)
Detritus (Offshore)	-484215.56 (1.79%)	-311134.35 (1.15%)

3.2 Observational Data

Trends in observational data following the DWH blowout varied by area and functional group. In general, the biomass increased across the Northern Gulf of Mexico and decreased nearshore around Louisiana (Table 6). When compared to observational fisheries data across the entire Gulf of Mexico trends in the model outputs were in agreement for Red Drum, Sea Trout, Blue Crab, Pigfish, Pompano, and stingrays. Biomass increased for Red Drum and Stingrays. While biomass decreased for Sea Trout, Blue Crab, Pigfish, and Pompano.

Table 6. Percent change in population based on observational fisheries data collected in 2009 and 2010. All data was corrected for effort and averaged across the number of sampling events.

Functional Group	Observed Change 2010	Area	Source	Unit of Change
Red Drum	68.1%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Sand Sea Trout	-31.9%	Northern Gulf	SEAMAP	Calculated Biomass
Sand Sea Trout	23.5%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Menhaden	127.4%	Northern Gulf	SEAMAP	Calculated Biomass
Gulf Menhaden	-51.8	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Catfish	162.2%	Northern Gulf	SEAMAP	Calculated Biomass
Gafftopsail Catfish	-31.7%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Shrimp	82.4%	Northern Gulf	SEAMAP	Biomass
Brown Shrimp	38.6%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
White Shrimp	-10.8	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Blue Crab	-10.1%	Northern Gulf	SEAMAP	Calculated Biomass
Blue Crab	-44.8%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Pigfish	-28.9%	Northern Gulf	SEAMAP	Calculated Biomass
Pigfish	-7.6%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Atlantic Croaker	27.1%	Northern Gulf	SEAMAP	Calculated Biomass
Atlantic Croaker	-30.1	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Grouper	309.7%	Northern Gulf	SEAMAP	Calculated Biomass
Pompano	-0.5%	Northern Gulf	SEAMAP	Calculated Biomass
Florida Pompano	161.7%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Red Snapper	14.6%	Northern Gulf	SEAMAP	Calculated Biomass
Pinfish	56%	Northern Gulf	SEAMAP	Calculated Biomass
Pinfish	10.1%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass
Atlantic Stingray	107.8%	Louisiana nearshore	Louisiana Department of Wildlife	Calculated Biomass

When compared to publication data similar trends where seen within the phytoplankton, meiofauna, and macrofauna. Parsons et al. (2015) observed an 85% lower abundance in phytoplankton after the spill when compared to baseline data. This decreasing trend was also seen in the model. Baguley et al. (2015) found that meiofauna abundance increased in offshore areas impacted by the DWH while diversity and richness decreased. The DWH scenario output shows the meiofauna functional group starting to increase 6 months after the spill and eventually reaching a higher biomass than under normal conditions (Figure 5). However, there is an offset on when the increase is seen in the model (December) and when it was observed (Sept-October). Washburn et al. (2016) found that macrofauna diversity and abundance was lower in areas impacted by the DWH. This trend was captured in the model where macrofauna biomass was lower than the normal conditions in 2010 and 2011 (Figure 6). Detritus was generally lower than normal conditions with a two month increase above normal levels (Figure 7). This is supported by observational data showing a reduction in primary production and in carbon export to the deep sea (Prouty et al. 2016).

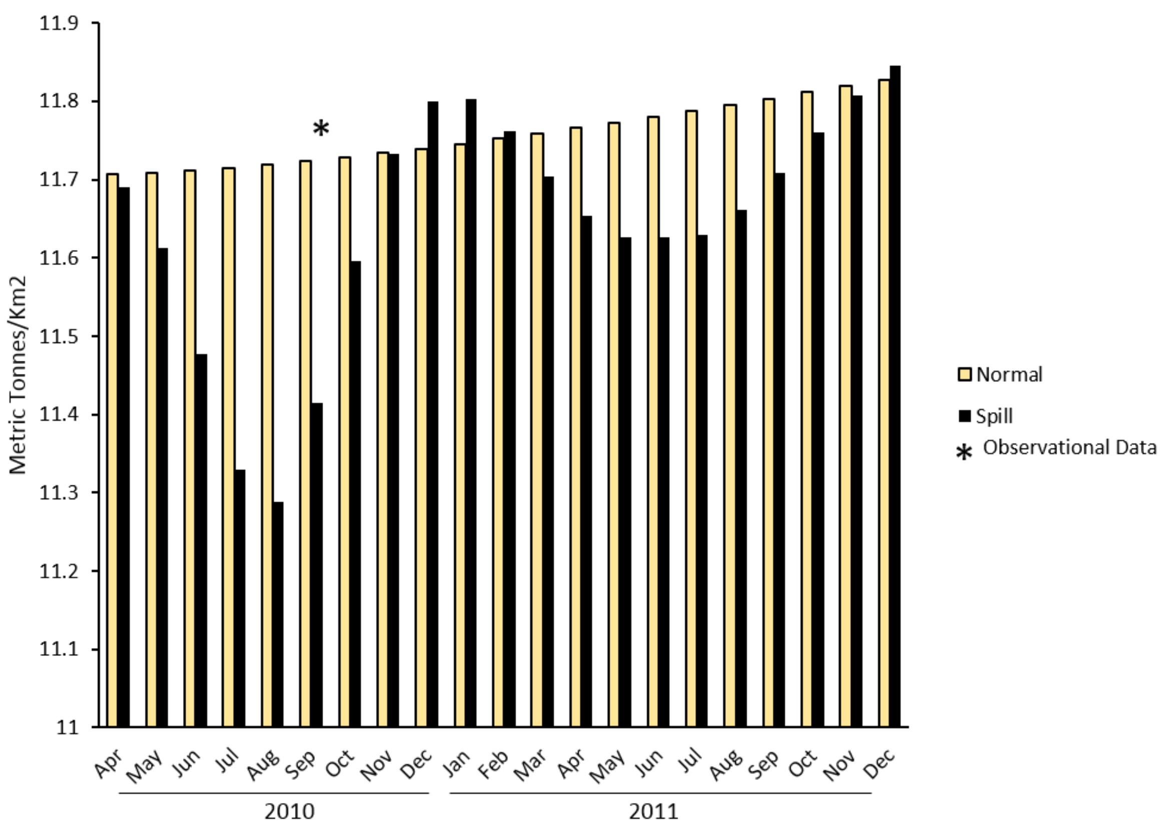

Figure 5. Differences in meiofauna biomass between the two model scenarios: (1) normal conditions and (2) DWH blowout.

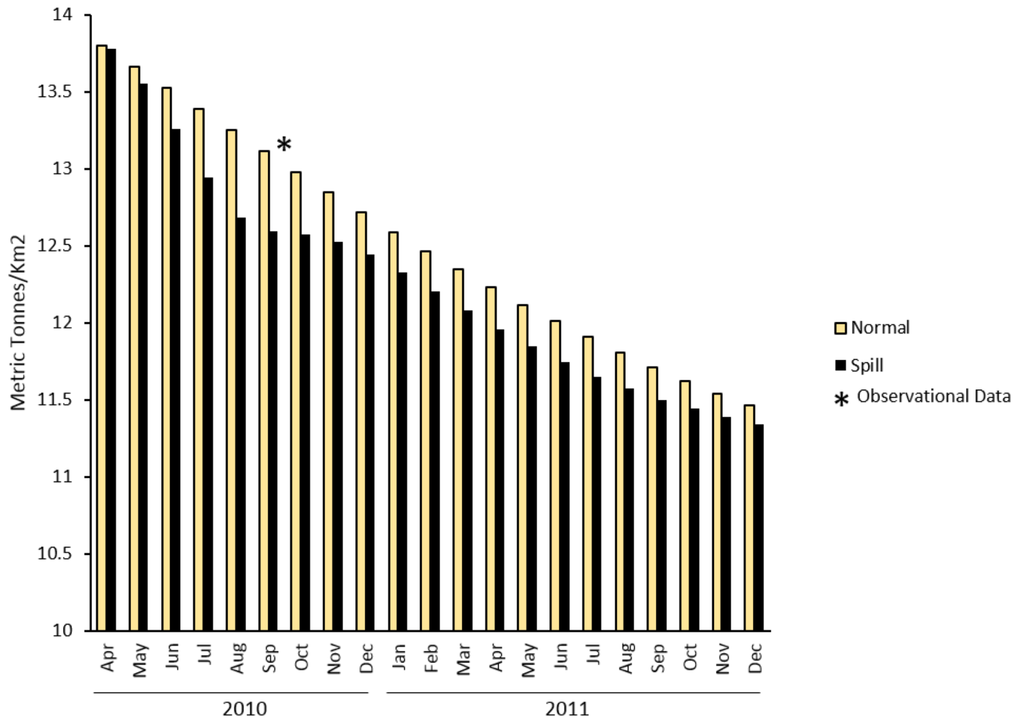

Figure 6. Differences in macrofauna biomass between the two model scenarios: (1) normal conditions and (2) DWH blowout.

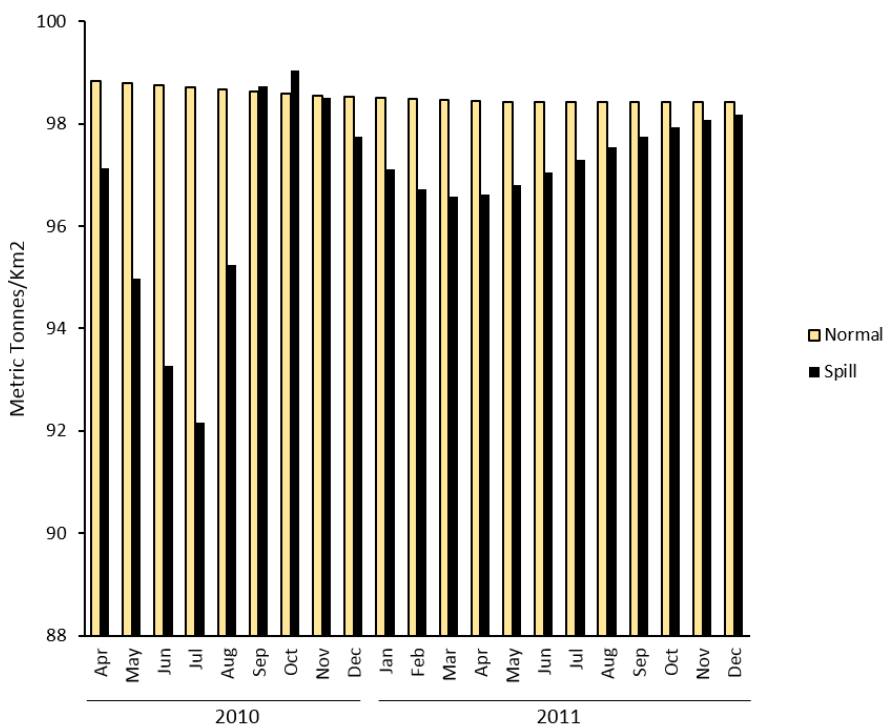

Figure 7. Differences in detritus between the two model scenarios: (1) normal conditions and (2) DWH blowout.

3.3 Ecosystem Services

The functional groups within the model were assigned to one of the four ecosystem service categories. All four of the categories ended up being represented within the model; cultural, provisioning, regulating, and supporting (Table 4). Six functional groups were chosen for further analysis concerning changes in ecosystem services. To examine the changes in provisional services, specifically commercial fisheries, the following functional groups were selected Shrimp, Blue Crabs, Stone Crabs, Grouper, and Red Snapper. The yields for all groups except grouper were lower in the spill scenario (Table 7). The change in yield for each resulted in a monetary changes ranging from $65 to -$5,091,109 (Table 8).

Table 7. Yield outputs from the normal (NS) and spill scenarios (S). Yields are in Metric tons.

Functional Group	2008	2009	2010 (S)	2010 (NS)	2011 (S)	2011 (NS)	2012 (S)	2012 (NS)
Red Drum	8.82	8.71	8.62	8.69	8.63	8.77	8.80	8.86
Grouper	7.59	8.24	8.83	8.83	9.23	9.28	9.53	9.52
Red Snapper	2508.89	2668.46	2668.80	2765.45	2598.43	2681.73	2499.37	2574.80
Shrimp	38850.36	39994.31	39283.54	40498.97	39755.78	40148.47	38791.35	39461.45
Stone Crab	3707.11	3185.61	2806.34	2874.15	2301.53	2536.96	1842.97	2129.63
Blue Crab	2209.69	2317.67	2300.03	2349.71	2212.49	2302.55	2171.32	2232.28

Table 8. Changes in ex-vessel value in commercial fisheries attributed to the Deepwater Horizon blowout from modeled output.

Functional Group	2010	2011	2012
Shrimp (All Species)	-$5,091,109	-$1,722,779	-$2,747,800
Blue Crab	-$109,538	-$174,726	-$130,343
Stone Crab	-$683,122	-$2,320,116	-$2,913,381
Grouper	$0	-$310	$65
Red Snapper	-$666,923	-$587,607	-$563,727

To determine how carbon sequestration could have been altered following the DWH blow out the change in the amount of detritus in the offshore environment (200-2000m) was measured from the model outputs. When compared to normal conditions this resulted in a detrital decrease of 257,023.01 metric tons in 2010 and a decrease of 214.89 metric tons in 2011. When carbon sequestration percentages were applied to the model outputs sequestration decreased by 1.15% (Global Average: 24349.55 metric tons, Gulf Stream: 61.21 metric tons) in 2010 and decreased by 0.00096% (Global Average: 20.36 metric tons, Gulf Stream: 0.051 metric tons) in 2011. Based on the 2015 social cost of CO_2 (IWGSCC, 2015) this is equivalent to losses ranging from $1.84 to $876,583 (Table 9).

Table 9. Modeled monetary changes in carbon sequestration attributed to the Deepwater Horizon blowout.

Sequestration Value Used	2010 (NS)	2010 (S)	2011 (NS)	2011 (S)
Gulf Stream	$190,908	$188,705	$191,150	$191,148
Loss	$2,204		$2	
Global Mean	$75,941,427	$75,064,843	$76,037,476	$76,036,743
Loss	$876,584		$733	

4. Discussion

It's always important in environmental assessment to be able to understand how an event has an effect on people. The only way to do that is to translate biophysical impacts to ecosystem service impacts. In the case of NRDA there is a need to monetize because of legal obligations to pay fines or replace damaged resources. Therefore, we have to have ways to go from sampling to describing the results in a way that is descriptive and clearly shows the values there were lost. What we provide here is a first approach to solving this problem. To our knowledge this was the first time an attempt has been made to quantify how offshore ecosystem services were affected by the DWH. The results show an overall negative impact on ecosystem services.

4.1 Differences Between Observed and Predicted Results

When compared to observational data the model outputs were in agreement for 10 functional groups: Red Drum, Sea Trout, Blue Crab, Pigfish, Pompano, Stingrays, Meiofauna, Macrofauna, and Detritus. The differences found among the other groups are likely attributed to model parameters that can be improved upon. The parameters 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, a limitation of 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; Incardona et al., 2011; and Mckenna et al., 2013). 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., 2014: Incardona & Scholz, 2015; Incardona et al., 2013; 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). However, 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 Gulf of Mexico. To account for the small portion of this area impacted by the blowout, the percent of the population affected was accounted for when calculating the forcing function. While this method is valid it does not necessarily provide the most realistic model predictions. The area of impact in the water column was calculated using the area of the surface oil slick. While a good first step, in reality the oil spread differently below the surface (see for example, Paris et al., 2012).

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. Including an additional model scenario showing the predicted impact of closures would provide results closer to the observational data as well as provide an indication of the closure efficiency.

4.2 Deepwater Horizon

Despite discrepancies between observed and predicted results, the *EWE* model is a valuable tool. Fretzer (2016) demonstrated that Ecopath was the number one tool for environmental management in the European Union. To our knowledge, this model is the first attempt to show how offshore ecosystem services have been affected by the Deepwater Horizon blowout. With regards to ecosystem functioning the functional group that was most impacted in 2010 was the Macro Zooplankton while the least impacted group was Grouper 3+. Within the model the largest area of impact was the ocean surface covered by the surface slick. Therefore, it stands to reason that the relatively immobile organisms inhabiting this area, for example the Macro Zooplankton, would be the greatest impacted. In 2011 the values showed a greater percent change when compared to the normal simulation. The functional group that experienced the greatest impact was Mullet 18+ while Meiofauna showed the smallest impact. Of the ecosystem services valued the greatest impact was seen within the commercial shrimping industry with an ex-vessel loss of $5,091,109 in 2010. The loss of this ecosystem service not only impacted the environment but the fishermen whose lively hoods depend on the shrimping industry. This is a valuable tool when deciding where support is needed and how recovery funds should be dispersed.

In conclusion, this is an important first step towards understanding and valuing changes to ecosystem services. This approach can be applied to different situations and different environments. It is a valuable tool to resource managers and decision makers because it shows changes in ecosystem services in an easily understandable way. Not only did it identify what changes occurred in offshore ecosystem services as a result of the DWH but it also provided a magnitude of change. The model is not an exact match to observations but it is still an important tool. As George Box said "All models are wrong but some are useful," and this model is very useful and an important step forward.

5. From Exploratory to Future Research

5.1 Where We Were

To date, work on offshore ecosystem services has mainly focused on high level reports that either state the need for a better understanding of ecosystem services (NRC, 2012; NRC, 2013) or work to identify potential goods and services (Werner et al., 2014; Armstrong et al., 2010). There has been little to no information with regards to the actual value or the magnitude of values. This means we have no idea on how or to what magnitude the DWH affected offshore services. Therefore, we have no starting point of analysis.

5.2 Where We Are

This project has provided the first estimates of ecosystem services in offshore environments and evaluated their changes as a result of the DWH blowout. By fostering a collaboration between scientists and students in the biophysical and economic disciplines improvements have been made to how we value offshore ecosystem services. Using this approach it was possible to identify the possible impact of the DWH blowout on offshore services and provide a magnitude of impact in a way that is understandable to decision makers.

5.3 Where We Want To Go

5.3.1 Improvements to the Current Biophysical Model

The *EWE* model predictions did not correspond to trends in observational data for all functional groups. However, improvements can be made to the model so that it is more realistic. The first of which would be to work with toxicology experts to develop a specific dose response for each of the functional groups instead of using one dose response value for all fish groups. Second, seasonal affects 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 source are the phytoplankton that fall from the surface. Improvements can also be made to the area of impact calculations. Calculations for water column impacts were made based on surface oil area. Collaborating with Claire Paris and her lab can aid in the development of a GIS layer showing oiling area by depth.

There is another aspect of *EWE* that was not utilized for this report. *EWE* includes a spatial analysis tool called Ecospace that builds upon the food web and time dynamics of Ecopath and Ecosim. This tool allows for the consideration of habitat effects and organism movement and generates a spatial representation of the changes. The creator of the original model, Paul Suprenand, expanded upon for this project is currently working on creating a dynamic Ecospace model in response to oil activity in the artic. Collaboration with Paul Suprenand could lead to a more detailed and visual explanation of the changes to ecosystem services following the DWH blowout.

5.3.2 Further Project Expansion

Future work will require improving model structure, expanding the model to fill data gaps, and linking the model to economic analysis. These activities are very labor intensive and will require much more effort over the years. There is a whole movement into thinking about benefit relevant indicators (BRIs) that more explicitly links bio-physical structure, function, and processes to human well-being. Additional work would look at those connections, of which many BRIs could be bio-physical, because of the inherent difficulty in monetizing ecosystem services. In addition, we need to continue to develop the biophysical science behind ecosystem services whether it be monetized or non-monetized.

On the economic side only four ecosystem services were investigated in the current project. Those represented in the model are considered low hanging fruit. Further works needs to be done in order to determine evaluation and valuation techniques for the remaining services. For example, waste regulation is an important offshore service that was not considered and often utilized in our society. In the past the deep ocean was often used for waste disposal and continues to be. It is of particular interest as a way of storing excess CO^2. Of the services explored in the current model improvements can be made to the associated economic models. For example, a collaboration with NOAA to use their NMFS economic impact model would lead to a more detailed and accurate calculation of ecosystem services values not only for commercial fisheries but recreational as well. In addition, management actions were not adequately addressed in the model. For example, fisheries closures because of the DWH blowout were not account for in the current model. Fisheries closures would affect both the biophysical and economic model.

The biophysical *EWE* model didn't capture everything. For example bacteria, and biogeochemical cycles were left out of model due to the limitations of *EWE*. Therefore an additional component could be to compare the effectiveness of different model types in predicting changes in offshore ecosystem services as a result of the DWH blowout. There are a number of dynamic system models that could be tested but there is often a tradeoff between complexity and usability. For example, Atlantis represents all ecosystem components from nutrients to predators and it is linked to an oceanographic model but it is complex and not user friendly. As a result fewer people utilize it. In contrast EWE is user friendly and a commonly used tool for fisheries studies.

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